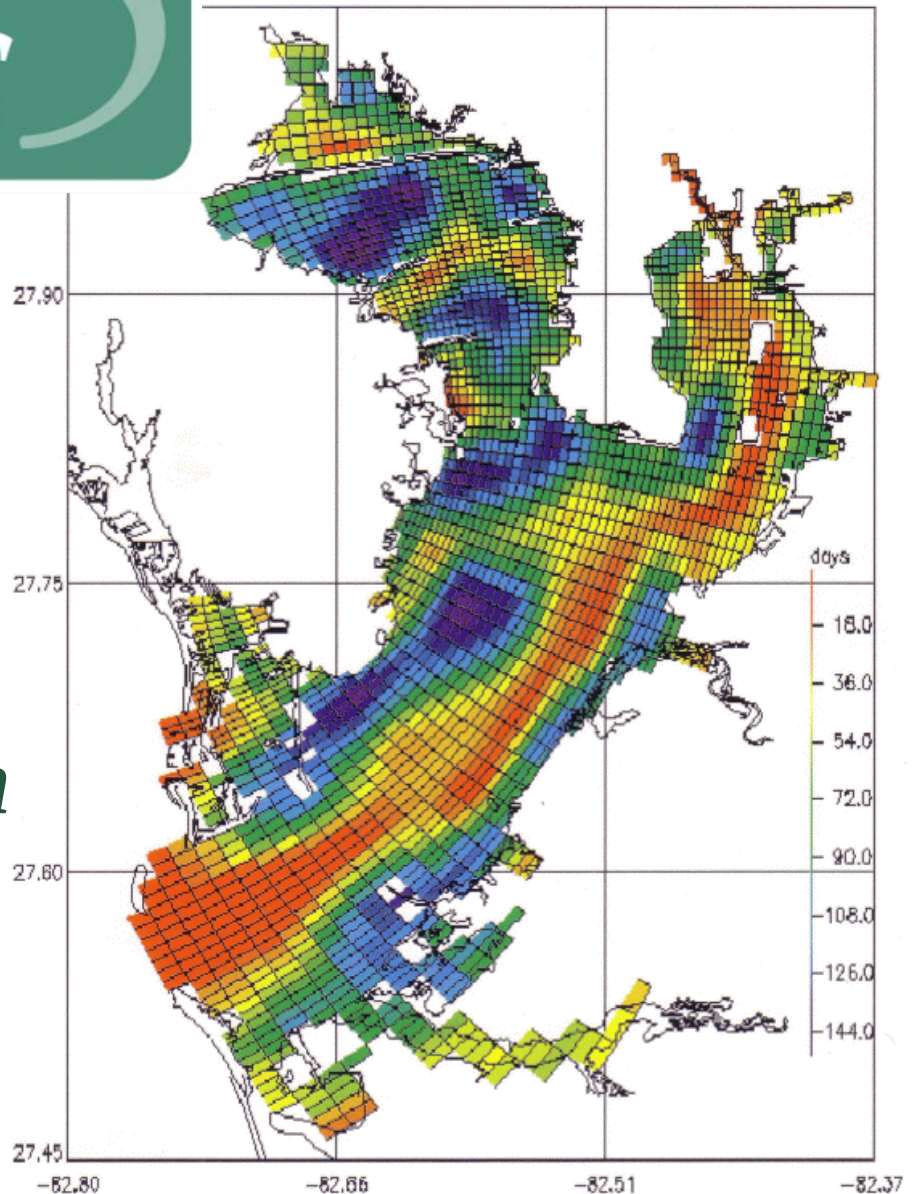




*Bay
Area
Scientific
Information
Symposium*



**Using Our Knowledge
to Shape Our Future**

Tampa Bay Estuary Program
Tampa Bay Regional Planning Council

**PROCEEDINGS
THE FIFTH TAMPA BAY AREA SCIENTIFIC INFORMATION SYMPOSIUM
BASIS 5**

**October 20 - 23, 2009
St. Petersburg, Florida**

**S.T. Cooper
Editor**

FOREWORD

These Proceedings contain presentations given at the fifth Tampa Bay Scientific Information Symposium, held October 20 - 23, 2009 in St. Petersburg, Florida. Since its inception in 1987, the BASIS conference series has provided a forum for sharing state-of-the-art research on Tampa Bay. The theme of BASIS 5, Using Knowledge to Shape the Future, is organized around the important components of the estuary: Geology & Sediments, Circulation, Water Quality & Primary Productivity, Water Quality & Ecosystem Implications, Seagrass Resources & Management, Ecosystem Restoration & Management, Biota, and Tidal Tributaries, as well as Environmental Regulations & Protections, Archaeology & Anthropology and Science Communication. More than 200 scientists, resource managers and students from the Tampa Bay area participated in the symposium, which included 75 papers and posters.

ACKNOWLEDGEMENTS

We gratefully acknowledge the input and direction provided by the Steering Committee for BASIS 5, which resulted in the well-balanced and thorough representation of Bay research. We also sincerely thank Dick Eckenrod, past Executive Director of the Tampa Bay Estuary Program, for providing opening remarks which reflected on the first BASIS and the progress that has been made in restoring Tampa Bay, quite an accomplishment during a time when the population within the watershed more than doubled.

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BASIS 5 OPENING REMARKS

R. Eckenrod

The theme for this year's symposium is "Using Our Knowledge to Shape Our Future". Over the next three and one-half days prominent scientists and managers will be presenting recent advances in the science and management of Tampa Bay, highlighting the progress made in our understanding of the bay since the original BASIS in May 1982.

The organizers of the first BASIS knew that it would not be the last review and compilation of scientific information about Tampa Bay. Rather, it would be the start of a process to comprehend Tampa Bay as a single ecological unit, to better understand the linkages among its many physical, chemical, and biological components, and to apply that knowledge to attain a better managed and more productive estuary. Indeed the vision of the first BASIS has been realized in the three BASIS symposia that followed it: BASIS 2 emphasizing a watershed approach to bay management; and BASIS 3 and BASIS 4 reporting on how recently acquired knowledge was being applied to improve management of the estuary. All of the symposia have served the community well by bringing together scientists and managers to share the results of their work and their future plans to achieve a healthier, more productive bay.

Is there anyone here today who has attended all four of the previous BASIS symposia? You have an especially useful perspective on what these four symposia have contributed to the advancement of science and management of the bay.

Who in the audience is attending BASIS for the first time? You will be impressed over the next three and one-half days by how far the scientific community in the Tampa Bay region has advanced since the first BASIS 27 years ago.

The benefits of these symposia are manifest in a variety of ways, not the least of which has been more comprehensive and integrated management of the Tampa Bay ecosystem. Indeed the theme of BASIS 1 was a striving for holistic management of the bay ecosystem. Among other things, that meant developing a better understanding between water quality and fish and wildlife habitats in the estuary. It meant understanding how much light and the quality of light required for seagrasses to thrive. It led to developing mathematical relationships between light attenuation and algal concentrations and between chlorophyll and nitrogen loading. Those improved statistical relationships provided a foundation for science-based goals that partners in bay restoration could embrace with great confidence that their investment in bay restoration would yield tangible results. And, in fact, since 1982, the year of the first BASIS, seagrass coverage in Tampa Bay has expanded by 12 square miles, a 35 percent increase, making Tampa Bay one of the few estuaries in the world that has shown such a remarkable recovery. Those are the kinds of results that are possible when scientists and resource managers like you come together in conferences like BASIS to provide scientifically-sound and credible guidance to community leaders.

It is the kind of holistic thinking incubated in BASIS 1 that led to the concept of restoring more natural proportions of wetland habitat types, a balance which, once achieved, would allow the ecosystem to reach an optimum productivity and sustainability. Following that restoring the balance paradigm,

SWFWMD, other agencies, and private groups engaged in habitat restoration are continuing to make steady progress toward a comprehensive restoration of Tampa Bay habitats.

In 1982, organizers of the first BASIS reported a population of 1.3 million in the greater Tampa region. By 2006 the population in the four-county region; Pinellas, Hillsborough, Pasco, and Manatee had more than doubled to 2.8 million. The recovery of seagrasses and the steady progress in restoring most other bay habitats is all the more remarkable when considering that it was achieved in the face of that burgeoning population pressure.

Bay area communities, backed by a strong scientific community and firm but reasonable regulation have demonstrated what many well-informed people might have written off as impossible: that an intensely developed urban estuary like Tampa Bay could be substantially restored.

The recovery of Tampa Bay is testimony to the resilience of estuaries. While people frequently refer to the sensitivity of estuaries, these dynamic environments with continually changing depths and salinities are among the hardiest of ecosystems. They can take a knock-down punch, and then get up off the mat.

But an estuary cannot withstand unabated insults in the form of pollution loads and habitat destruction and be expected to thrive. Estuaries must be given a chance to exercise their recuperative powers. The Tampa Bay scientific community through conferences such as BASIS and participation in various management programs has defined what conditions are necessary to allow recovery to occur, both through natural and controlled processes.

I encourage those of you who are attending BASIS for the first time and you BASIS veterans to seek out each other during the breaks and learn from one another, not only about your particular area of research, but how research in your field might be enhanced through collaboration. This is a great opportunity to make some new acquaintances and hatch new ideas for collaborative research. Research projects that are inspired over the next three and one-half days might very well be ones that you are reporting on at the next BASIS symposium. Do what you can to continue the tradition of holistic research and management that is the legacy of the BASIS series.

From time to time, the Tampa Bay Estuary Program and other institutions and programs revisit research priorities that are important in advancing their programs' mission. And they look to scientists and managers like you for advice in establishing those priorities. The scientific information and approaches to resource management that will be exchanged over the next three and one-half days will better equip you to provide well-informed advice for the benefit of those programs.

What have we learned since the first BASIS held 27 years ago about Tampa Bay, and how have we applied that knowledge to better manage the bay? To the 27 year-olds in the audience, 27 years may seem like a long time. But it's less than a human generation. How far have we come in that short 27 years? I submit we've come a long way. But you need not take my word for it. The presenters you'll be hearing over the next three and one-half days will narrate what has truly been a remarkable story of the rebirth of a bay.

Since the first BASIS, one to three decades of additional monitoring of water and sediment quality, primary production, aerial extent of estuarine habitats, seagrass transect studies, populations of benthic macroinvertebrates, birds, fishes, and manatees has been performed. Those data together with new tools such as 3-D hydrodynamic models and satellite imagery have greatly enhanced our ability to comprehend the Tampa Bay ecosystem. Over the next three and one-half days you'll have a chance to make your own assessment of how well the scientific community has responded to the challenges put forth in BASIS 1.

Great strides have been made in understanding natural process and solving problems that weren't even on the radar screen at the time of BASIS 1. Consider for example atmospheric deposition of nitrogen to the bay. It simply wasn't being discussed as a significant part of the nutrient budget for Tampa Bay. Yet today we have a basic understanding of the magnitude and distribution of wet and dry deposition of nitrogen species in the bay and its watershed. And reductions in NOx emissions are now a crucial part of the nitrogen management strategy for the bay to help sustain seagrass recovery.

Sea level rise - a very timely issue today - was only being discussed in terms of geologic time at BASIS 1. But now designers of coastal habitat restoration projects at SWFWMD and elsewhere are factoring the accelerating pace of sea level rise into their projects, recognizing that today's salterns and high marshes will be tomorrow's mangrove swamps.

I have no doubt that in another 27 short years some of you will have plumbed new depths of understanding of the Tampa Bay ecosystem and will be reporting at BASIS 10 on topics we cannot fathom today.

I want to take a minute to thank some of the many people who have worked hard to organize BASIS 5, beginning with Ed Sherwood, the Program Scientist at the Estuary Program, Suzanne Cooper and Wren Krahll at the Tampa Bay Regional Planning Council, and many of the other staffers at the Estuary Program and Planning Council.

Special thanks are certainly in order for the session organizers including:

- Gregg Brooks chairing the session on the Geology and Sediments of Tampa Bay
- Richard Boler and Roger Johansson, Co-Chairs of the Water Quality and Primary Production session;
- Robin Lewis, Roger Johansson, and Walt Avery for the Seagrass Resources and Management session;
- Brandt Henningsen and Doug Robison for the Ecosystem Restoration and Management session;
- Ann Hodgson and Bob McMichael for the Tampa Bay Biota session;
- Justin Krebs for the session on Tidal Tributary Studies;
- Suzanne Cooper for Integrating Watershed Management Initiatives;
- Rhonda Evans and Clark Hull for Environmental Regulations and Protections;
- Ed Sherwood for the General Call session;
- Ernie Estevez for the session on Climate Change and Future Challenges
- Bill Burger for the Archaeology and Anthropology session; and
- Nanette O'Hara for the session on Science Communication.

And our sincere appreciation goes to all of the presenters who will be sharing the results of their research and management efforts to improve Tampa Bay.

Finally, thanks to all of you for attending BASIS 5. Let's commit today to making the next 27 years of scientific inquiry and management of Tampa Bay as productive as the last 27 years.

HIGH-RESOLUTION GEOLOGIC INVESTIGATIONS IN TAMPA BAY: SOURCE, DISTRIBUTION AND ACCUMULATION RATES OF FINE- GRAINED, ORGANIC-RICH SEDIMENTS IN SAFETY HARBOR, FLORIDA

G.R. Brooks & R.A. Larson

ABSTRACT

While, with few exceptions, the geology of Tampa Bay has likely changed little since the last BASIS, our ability to detect geologic events at much higher resolution, sub-decadal to sub-annual temporal scales, and mm-cm spatial scales, has increased dramatically. Consequently, we can more accurately link causes/effects, and determine controls/processes of geological events preserved in the sedimentary record.

A case in point is the recently completed Safety Harbor Sediment Project. Objectives included identifying the source, accumulation rates, and distribution patterns of fine-grained, organic-rich sediments entering Safety Harbor. A multidisciplinary approach consisting of sedimentology/geochronology, bottom type classification/mapping, geochemistry, and biology, was undertaken to address these objectives. The sedimentology/geochronology was based on a dense sampling grid of 60 core sites. It was found that organic-rich, fine-grained sediments have been accumulating in deep, open portions of Safety Harbor proper and in isolated, sheltered depressions along the western shore. In most cases, these sediments have been accumulating for thousands of years (i.e. since being flooded by the last sea-level rise), but rates of accumulation have increased steadily since the early 1900s, with a dramatic increase (~3x) since ~1960. Sedimentological data point to the Lake Tarpon Outfall Canal (LTOC) as the primary sediment source, but geochemical analyses indicate a marine source for the organic fraction. The LTOC was likely not only the source for land-derived, mud-sized sediments, but nutrients as well. These nutrients in turn enhanced the production of marine organic detritus that was deposited along with the land-derived muds because they both react the same hydrodynamically. Thus fine-grained, organic-rich sediments entering Safety Harbor originate from multiple sources, the identification of which would have been unlikely without utilizing a multidisciplinary approach.

INTRODUCTION

It is generally perceived that the input/accumulation of fine-grained, organic-rich sediments in coastal systems can create negative impacts for environmental, recreational and aesthetic reasons. This project was designed to better understand the distribution and origin of fine-grained, organic-rich sediments accumulating in Safety Harbor (upper Tampa Bay), Florida. A four-pronged approach was developed, consisting of: 1) sediment and seafloor mapping; 2) sedimentology and geochronology; 3) geochemistry; and, 4) ecology. This report focuses on the sedimentology and geochronology.

Geologic Setting—Safety Harbor occupies the northwest corner of Old Tampa Bay, which represents the northwestern lobe of the large (<1,000 km²), multi-lobed Tampa Bay estuarine system of interconnected bays and lagoons (Figure 1). Unconsolidated sediments throughout Tampa Bay, including the study area, can be differentiated into 11 distinct sedimentary units (termed “facies”) representing the transition from non-marine to estuarine conditions as a result of flooding during the most recent sea-level rise (Table 1). This transition began in the study area ~3,000 – 6,000 years ago (kya) (Brooks, in press). Although most facies are found in all regions throughout the bay, the 3-D facies architecture (i.e., stacking patterns) varies considerably from region to region. Each region exhibits a unique facies architecture suggesting that Tampa Bay as a whole has had a complex sedimentological history with different regions developing as independent basins.

Surface sediments throughout Tampa Bay are composed of 4 of the 11 facies identified (Table 1; Figure 1). Carbonate sands and gravels occupy most of the lower to mid bay reflecting a strong marine influence. Clean, medium-fine, quartz sands occupy almost the entirety of old Tampa Bay, the eastern portion of Hillsborough Bay and portions of mid and lower Tampa Bay, likely reflecting a combination of relict (remnant), re-worked, and modern fluvial sand deposition. Muddy sands contain a significant fine-grained component, and occupy deeper portions of the open bay, and partially restricted regions around the bay periphery. Organic sandy mud occupies the bathymetric depressions in western Hillsborough Bay and poorly flushed regions in upper Old Tampa Bay, including much of Safety Harbor. Baywide surface sediment distribution patterns have been attributed to: 1) sediment source and supply rate, 2) bathymetry, a function of the antecedent topography, and 3) the winnowing effect of wind-generated waves that prohibits modern sediment deposition in more open portions of the Bay (Brooks, in press).

METHODS

A total of 60 sites were chosen for sediment coring (Figure 1) based on a combination of factors including: 1) adequate coverage of the study area; 2) areas of interest designated by the local population; and, 3) seismic profiling, side scan sonar, and QTC bottom classification data (see Locker et al. this volume), that helped to identify the aerial distribution of fine-grained, organic-rich sediments.

Vibracores and/or push cores were collected at all sites using 3" diameter x 20' long aluminum pipes. Vibracores were collected according to the method described by Lanesky et al. (1979) using a 5.5 hp gasoline engine coupled to a standard cement vibrator. Cores were collected aboard 20' and 31' vessels. Navigation was by WAAS GPS, estimated at sub 3 m accuracy.

Based upon the initial results of coring at the original 60 sites, five sites were chosen for short-lived radioisotope (^{210}Pb , ^{137}Cs , ^7Be) geochronology (Figure 1). Site selection focused on areas of interest, as well as fine-grained sediment distribution. Push cores were collected by SCUBA at each of the five sites using 4" diameter x 3' long acrylic tubes. Cores were kept vertical throughout the entire process.

All cores were transported to the Eckerd College Marine Geology Laboratory for analyses. Aluminum cores were split longitudinally, photographed, and described visually by standard methods. Based upon visual description, each core was subsampled at specific intervals for texture and compositional analysis. All lithologies throughout the core were sampled, but sampling focused on near surface sediments since they represent the most recent deposition.

All sediment samples from aluminum cores were analyzed in the laboratory for grain size, % carbonate content, and % total organic matter (TOM). Grain size was determined by initially wet sieving the sample through a 63 μm screen. The sand-size ($>63 \mu\text{m}$) fraction was then analyzed by settling tube (Gibbs 1974), and the fine-size ($<63 \mu\text{m}$) fraction was determined by pipette (Folk 1965). Carbonate content was determined by the acid leaching method according to the procedure described in (Milliman 1974). Total organic matter (TOM) was determined by loss on ignition (LOI) at 550° C for at least 2.5 hours (Dean 1974).

All acrylic cores were extruded carefully from the bottom to the surface and sampled at precisely 1 cm intervals. Samples were weighed immediately to provide the "wet weight" of the sample in order to determine pore water content (required for bulk density). Once dried, each sample was ground to a fine powder and analyzed for short-lived radioisotope (^{210}Pb , ^{137}Cs , ^7Be) activities by gamma emission on a Canberra Gamma Ray Photon Detector. Short-lived radioisotope (^{210}Pb , ^{137}Cs , ^7Be) data are used to determine annual to decadal-scale timing of events and sediment accumulation rates over the past 100 (approximate) years. ^{210}Pb , with a half-life of 22.8 years is

ideal where ecosystems have changed within the last century. ^{137}Cs , with a half-life of 30.3 years, is a thermonuclear by-product. Its presence is directly related to the atmospheric testing of nuclear devices, which was begun in the early 1950s and reached a global peak in the early - mid 1960s (Olsson 1986). Therefore, ^{137}Cs can be used directly as a dating tool, as well as provide a calibration for ^{210}Pb chronology (Noller 2000). ^7Be , with a half-life of ~54 days, is most useful for measuring sediment deposition on less than an annual time scale (Holmes 2001).

In order to assign specific ages to sedimentary layers down core, ^{210}Pb data must be run through a model. For the objectives of this project we chose the CRS (Constant Rate of Supply) Model (Appleby and Oldfield 1978), which is subject to the following assumptions/limitations:

- 1) The unsupported (excess) ^{210}Pb is supplied at a constant rate to sediments through time.
- 2) The initial ^{210}Pb concentration in the sediments is variable.
- 3) The influx rate of sediment is variable.

Activity values vs. depth down core are plotted for each core, and results of the CRS Model applied to assign a year and mass accumulation rate ($\text{g}/\text{cm}^2/\text{yr}$) to each individual sample. Mass accumulation rates (MAR) are preferred over linear accumulation rates (LAR) (e.g., cm/year) because they correct for differential sediment compaction down core, thereby enabling a direct comparison of accumulation rates throughout the core (i.e., over the last ~100 years). MAR are calculated as follows:

$$\text{MAR } (\text{g}/\text{cm}^2/\text{yr}) = \text{Dry Bulk Density } (\text{g}/\text{cm}^3) \times \text{LAR } (\text{cm}/\text{yr})$$

Where: Dry Bulk Density (g/cm^3) = dry wt/sample volume
Sample volume = 1 cm x area of 3" dia circle

RESULTS

Lithology—Quartz sands are by far the dominant sediment type throughout the study area. The quartz sand facies is defined as >80% sand-sized sediments consisting dominantly of the mineral “quartz” (Table 1). Quartz sands represent a terrestrial origin, thereby indicating sediment input from the surrounding land sources, or relict (remnant) sediments deposited prior to flooding by the most recent sea-level rise. Carbonate (i.e., marine-derived) sediments are rare (generally <10%), indicating minimal marine sediment input. TOM ranges from 0.1% to slightly more than 15%. Most, however, consist of <4% organic matter.

Stratigraphically, sediments represent a transition from non-marine to marine conditions during the recent sea-level rise, which began in this region ~3,000 – 6,000 years ago (kya). Therefore, sediments recovered in cores have recorded both natural conditions prior to most intense human activities, as well as recent anthropogenic impacts.

Geochronology—Core 55 was collected in ~0.8 m water depths along the northeast margin of Safety Harbor proper (Figure 1) due to reports of recent shoaling and accumulation of organic-rich mud. Virtually the entire core consisted of >90% quartz sand, <1% TOM, was strongly bioturbated and showed no systematic down-core decrease in ^{210}Pb activities. Thus, Core #55 was deemed unreliable for valid short-lived radioisotope geochronology, but there is no evidence that this site has experienced the recent accumulation of fine-grained and/or organic-rich sediments.

Core 16 was collected at the mouth of the Lake Tarpon Outfall Canal (LTOC) in ~2 m water depth (Figure 1). Virtually the entire core consisted of muddy sand, with 60-80% quartz sand-size material, and organic content (TOM) of 2-4% (Figure 2). A gradational contact at ~10 cm represents a recent (up core) increase in fine-grained sediments and higher organic content, to

~50% mud and >4% TOM, respectively. As will be discussed below, this transition corresponds to ~2000. Results of short-lived radioisotope analysis show excess ^{210}Pb to a depth of ~55 cm down core representing ~100 years before present (ybp). The first occurrence of ^{137}Cs was detected at ~35 cm down core, thereby theoretically representing the 1950s – 1960s. No ^7Be was detected indicating that there has been no measurable deposition in this area within the past year. The ^{210}Pb and ^{137}Cs data appear to be excellent quality and ideal candidates for CRS modeling. The first occurrence of ^{137}Cs corresponds to 1960, which supports the model results (Figure 3). Mass accumulation rates (MAR) were calculated for each data point providing a value for each date (year) calculated by the CRS model. From the 1920s to ~1960, MAR remain relatively stable at ~0.3-0.5 g/cm²/yr. Beginning at ~1960 there is a significant increase in MAR reaching peaks in 1975 and 1980, both representing a 2-3x increase over pre-1960 rates. From the early 1980s to the present, MAR have been relatively stable at ~0.5-0.7 g/cm²/yr, with the exception of a peak to 0.9 g/cm²/yr in the early 2000s.

Core 30 was collected in northwest Safety Harbor proper near the mouth of the LTOC (Figure 1) in 3.1 m water depth. The upper 2.45 m of the core consisted of a sandy mud, with >60% mud-sized sediments and >4% TOM (Figure 2). There are no visible contacts in the upper 2.45 m that would indicate an abrupt change in depositional patterns. However, there is a slight fining-upward, accompanied by a subtle increase in %TOM over the upper 50 cm, which, as will be discussed later, corresponds to the last ~60 years. A sharp contact at 2.45 m separates the surficial sandy mud unit from an underlying organic sandy mud unit, which likely represents the transition from non-marine to estuarine conditions during flooding by the most recent sea-level rise. Results of short-lived radioisotope analysis show excess ^{210}Pb to a depth of ~64 cm representing ~100 ybp (Figure 3). The first occurrence of ^{137}Cs occurs at ~44 cm. Results of the CRS model show the first occurrence of ^{137}Cs corresponds to the early 1950s, which is a little earlier than expected, but reasonable as nuclear testing started in the 1950s and has been reported in sediments of this age. MAR show a steady increase from the early 1900s to the present, from ~0.1 g/cm²/yr to almost 0.5 g/cm²/yr representing an ~5x increase over this time period. A major peak in the early 1950s to 0.6 g/cm²/yr, and minor peaks corresponding to the early 1920s, 1974, 1983, and 2005 likely represent relatively short-lived (annual-scale) pulses of sediment input.

Core 33 was collected from the lower mid-portion of Safety Harbor proper (Figure 1), in 5.0 m water depth. The upper 22 cm consisted of muddy sand with >80% sand-size material and >4% TOM (Figure 2). Gradational contacts were visible at 22 cm and 15 cm representing a subtle up-core increase in % mud and % TOM. Contacts at 22 cm and 15 cm correspond to ~1962 and ~1978 respectively. From 22 cm to the base of the core at 0.73 m sediments consist of shelly, muddy sand. The geochronology for this site was conducted in 2002 as part of the USGS Tampa Bay Integrated Science Study (P. Swarzenski, pers. comm. 2007). Consequently, those data will be integrated into this discussion. Excess ^{210}Pb was detected to a depth of 34 cm representing ~100 ybp (Figure 3). Excess ^{210}Pb detected below 34 cm is interpreted to represent a groundwater signal, which is not uncommon in this type of geologic setting. ^{137}Cs and ^7Be were not reported in the 2002 study and, therefore, are not available. MAR show a steady increase from the early 1920s to 1985 from ~1.0 to >2.5 g/cm²/yr. From 1985 to the present, the model shows an abrupt decrease to pre-1920s rates, which have remained consistent at 0.75 g/cm²/yr. Significant spikes in MAR at ~1955, ~1974, and ~1985 likely represent short-lived (annual-scale) pulses of sediment input. Note that these spikes correlate closely with those identified in Core 30. Also note that Core 30 showed a significant spike at ~2005, but since this core was collected in 2002 that peak is undetectable here.

Core 9 was collected near the west shore of Safety Harbor adjacent to Phillipi Pt. (Figure 1), in 1.3 m water depth. Lithologically, from 1.81 m (core base) to 1.75 m the sediments consist of blue-green clay (Figure 2), which is interpreted to represent carbonate platform material exposed

prior to the most recent rise in sea level. Overlying this blue-green clay unit, and separated by a sharp contact, is a muddy sand unit that extends to 0.79 m down core. The muddy sand represents marine deposition following flooding. A gradational contact separates the muddy sand from an overlying sandy mud unit that extends to 0.05 m down core. The sandy mud consists of ~20% to >75% mud size sediment, with ~4 to >12% TOM. Overlying this, and separated by a sharp contact corresponding to the mid-to-late 1990s, is a 5 cm-thick surficial layer of organic-rich mud with ~80% mud size sediments and ~14% TOM. This core shows the most distinct change in sediment distribution patterns over the past ~100 years. ^{137}Cs and ^7Be were not detected, the latter of which suggests no active deposition within the past year. The lack of ^{137}Cs is problematic in that there is no independent validation of the CRS model. MAR show a steady ~5x increase from ~0.20 g/cm²/yr during the early 1900s, peaking at ~1.00 g/cm²/yr in the mid 1980s, then a steady decrease to ~0.40 g/cm²/yr to the present (Figure 3).

Core 57 was collected along the western shore of the study area just north of the Safety Harbor Marina in ~2 m water depth (Figure 1). Site 57 is in the lee of a substantial sandbar that runs along the coast from Phillipi Pt. southward to the Courtney Campbell Causeway. Lithologically, the basal sedimentary unit extending from 0.95 m to the core base at 1.72 m is an organic mud consisting of 100% mud-size material and >12% TOM (Figure 2). The overlying unit from 0.20 m to 0.95 m is a sandy mud consisting of ~20% sand-sized material, the remainder being mud, and >12% TOM. The surficial 20 cm is separated by a gradational contact from the underlying unit. Surficial sediments are more like the basal unit with 100% mud and ~16% TOM. The gradational contact at 20 cm corresponds to ~1983.

The geochronology of Core 57 is problematic as excess ^{210}Pb was detected to the core base at ~62 cm (Figure 3). However, the relatively low activity at 62 cm suggests the signal may be close to background. Therefore we will assume that the core base at 62 cm corresponds to ~100 ybp. ^{137}Cs was detected throughout the core with a peak at ~0.40 m, but activities are extremely low. CRS modeling is not ideal because ^{210}Pb did not reach background and the ^{137}Cs signal is poorly defined. However, the ^{137}Cs peak at ~0.40 m correlates with the early 1950s, which is reasonable. The nature of the ^{210}Pb and ^{137}Cs data may reflect vertical mixing either by physical disturbance or bioturbation, or possibly could represent re-sedimentation. MAR reflect a slight but steady increase from the early 1900s to the present from ~0.02 - 0.1 g/cm²/yr.

DISCUSSION

Organic-rich, fine-grained sediments in the study area belong to the Organic Sandy Mud Facies (Brooks, In Press), and are the dominant sediment type in 13 (~22%) of the 60 core sites. The major depocenter is the relatively deep, low energy environment of central Safety Harbor proper extending into the mouth of the LTOC (Figure 1). This is likely a function of bathymetry (e.g., low-energy depressions), and proximity to source. Additionally, there are localized “hot spots” along the western shore adjacent to Phillipi Pt. and the Safety Harbor Marina (Figure 1). These regions are relatively isolated, occupying sheltered depressions, and in the case of the sites adjacent to the marina, are in the lee of energy-dampening sandbar(s). They may also represent localized sources of input. Essentially the entire eastern and northern shores, as well as portions of the western shore, are characterized by sand-sized sediments with no evidence of recent fine-grained and/or organic-rich sediment accumulation.

The major depocenter in central Safety Harbor proper is relatively deep for this portion of the Bay, and energy levels are relatively low, which promotes the deposition of both fine-grained sediments and organic material, regardless of their source. The extension of this pattern into the mouth of the LTOC, which is shallower, is consistent with a LTOC source for fine-grained, land-derived sediment input. However, results of geochemical and ecological analyses point toward a marine, rather than terrestrial, origin for the organic matter (see Peebles et al. this volume).

Consequently, the LTOC likely provides nutrients for marine organisms to flourish, and once dead the organic matter is deposited in low-energy, bathymetric depressions along with the fine-grained terrestrial sediments because they react the same from a hydrodynamic standpoint.

The central Safety Harbor depocenter was penetrated by seven cores including geochronology cores 16, 30 and 33 (on the periphery) (Figure 1). Each consists almost entirely of organic-rich sediments, often exceeding 2 m in thickness, with no distinct breaks in sediment type. Based on accumulation rates it is likely that this area has been accumulating fine-grained, organic-rich sediments since estuarine conditions were originally established several thousand years ago. Average linear accumulation rates (LAR) for the past hundred years range from ~34 cm/100 years on the periphery of the depocenter (site 33) to 54 cm/100 years at the mouth of the LTOC (site 16), to 59 cm/100 years in the middle of the depocenter (site 30). These rates are slightly higher than other fine-grained sediment sinks in Tampa Bay and Charlotte Harbor (both 28 - 44 cm/100 years), but consistent with other estuarine systems with minimal fluvial input (Nichols and Biggs 1985).

The isolated deposits of organic-rich sediments adjacent to Phillipi Pt. and in the Safety Harbor Marina area both show a relatively recent, distinct transition toward an increase in fine-grained, organic-rich sediment input and deposition. Once again, deposition is promoted by low-energy conditions resulting from a combination of bathymetry (i.e., low-energy depressions) and sheltering by sandbar(s). The precise sediment source is difficult to predict based upon sedimentology alone, but the pattern of deposition suggests point source input(s) from the adjacent land.

Based on lithology, CRS modeling, and MAR, all cores appear to have experienced an increase in rate of sediment accumulation over the past ~100 years (Figure 4). Both steady, consistent, long-term increases were detected, as well as short-term (annual scale) pulses of sediment input. Specifically, the similarity in MAR for cores 16 and 30 from the early 1900s to ~1960, including a common peak in the mid-1950s, suggests they were both responding to the same sediment input signal. From ~1960 to the present, however, the patterns have deviated somewhat. Site 16 at the mouth of the LTOC has experienced a major increase in accumulation rate, whereas Site 30, near the center of the depositional basin, has maintained a slight but steady increase. In addition, Site 16 experienced major pulses at ~1975 and ~1990. Site 30 exhibited similar pulses, but of much less magnitude. Site 16 shows another major peak in the early 2000s, which is absent in core 30. This pattern suggests both sites are responding to the same input/processes, but Site 16 is more sensitive, which is consistent with Site 16 being closer to the source, especially since ~1960 when the LTOC was constructed.

Core 33 shows a similar pattern to Sites 16 and 30 in the mid-1950s as well as a major pulse ~1975 and a minor pulse in the early 1980s. However, a major peak in core 33 in the mid-1980s shows a similar pattern to that of nearby Core 9 suggesting that Core 33 may be reacting to multiple sources of sediment input.

The two isolated depocenters along the west shore are likely reacting to different sediment input(s). MAR in Core 9, adjacent to Phillipi Pt. reached a peak in the mid-1980s showing no correlation to Sites 16 and 30, but it does correspond to a major peak in nearby Core 33, once again suggesting that these two sites may share a common source of sediment input.

Core 57, penetrating an isolated depocenter just north of the Safety Harbor Marina, shows essentially no correlation with the other cores, but exhibits a subtle, but steady increase in accumulation rate from the early 1900s to the present.

CONCLUSIONS

1) Fine-grained, organic-rich sediments occupy three primary depocenters. The major depocenter is the relatively deep, low-energy environment of central Safety Harbor proper extending into the mouth of the LTOC. The other two depocenters are small, localized regions along the western shore adjacent to Phillipi Pt., and the Safety Harbor Marina. Quartz sands and muddy sands dominate in all other areas, with the exception of some of the deeper regions in the south central to southeastern portion of the study area. Exceptions include dredged canals or boat basins that act as natural, fine-grained sediment sinks, and possibly along the mangrove fringe of northeastern Safety Harbor proper where organic-rich sediments are produced and deposited directly by the mangroves.

2) Sources and controls governing fine-grained, organic-rich sediment input are variable and unique to each depocenter. The major depocenter in central Safety Harbor proper is relatively deep, with concomitant low energy levels, promoting the deposition of fine-grained, organic-rich material. The extension of this pattern into the mouth of the LTOC is consistent with a LTOC source. The isolated deposits adjacent to Phillipi Pt. and the Safety Harbor Marina are similarly controlled by low energy conditions resulting from a combination of depth and sheltering by sandbars, but it is difficult to predict the precise source(s) for each.

3) Although sedimentological data point to the LTOC as the primary sediment source, geochemical and ecological analyses indicate a marine source for the organic fraction. The LTOC likely provides nutrients that enhance biological productivity creating the organic detritus that is deposited along with the land-derived muds because they both react the same hydrodynamically. Thus, there are multiple sources of these two fine-grained sediment components, which likely would have gone undetected without a multidisciplinary approach.

4) The primary depocenter in central Safety Harbor proper has been accumulating fine-grained organic-rich sediments for many thousands of years, likely since being flooded by the most recent sea-level rise. Over the past ~100 years there has been a subtle increase in both mud-size sediments as well as organic content. There has also been a steady increase in accumulation rate from the early 1900s to ~1960, but since 1960 the mouth of the LTOC has experienced a major increase in accumulation rate as well as pulses of input in ~1975, ~1990, and early 2000s. The two isolated depocenters along the west shore show no correlation to each other, or to the major depocenter in Safety Harbor proper. They are likely responding to different sediment inputs, and although both have experienced increases in accumulation rates over the last ~100 years, the exact timing of events is unknown.

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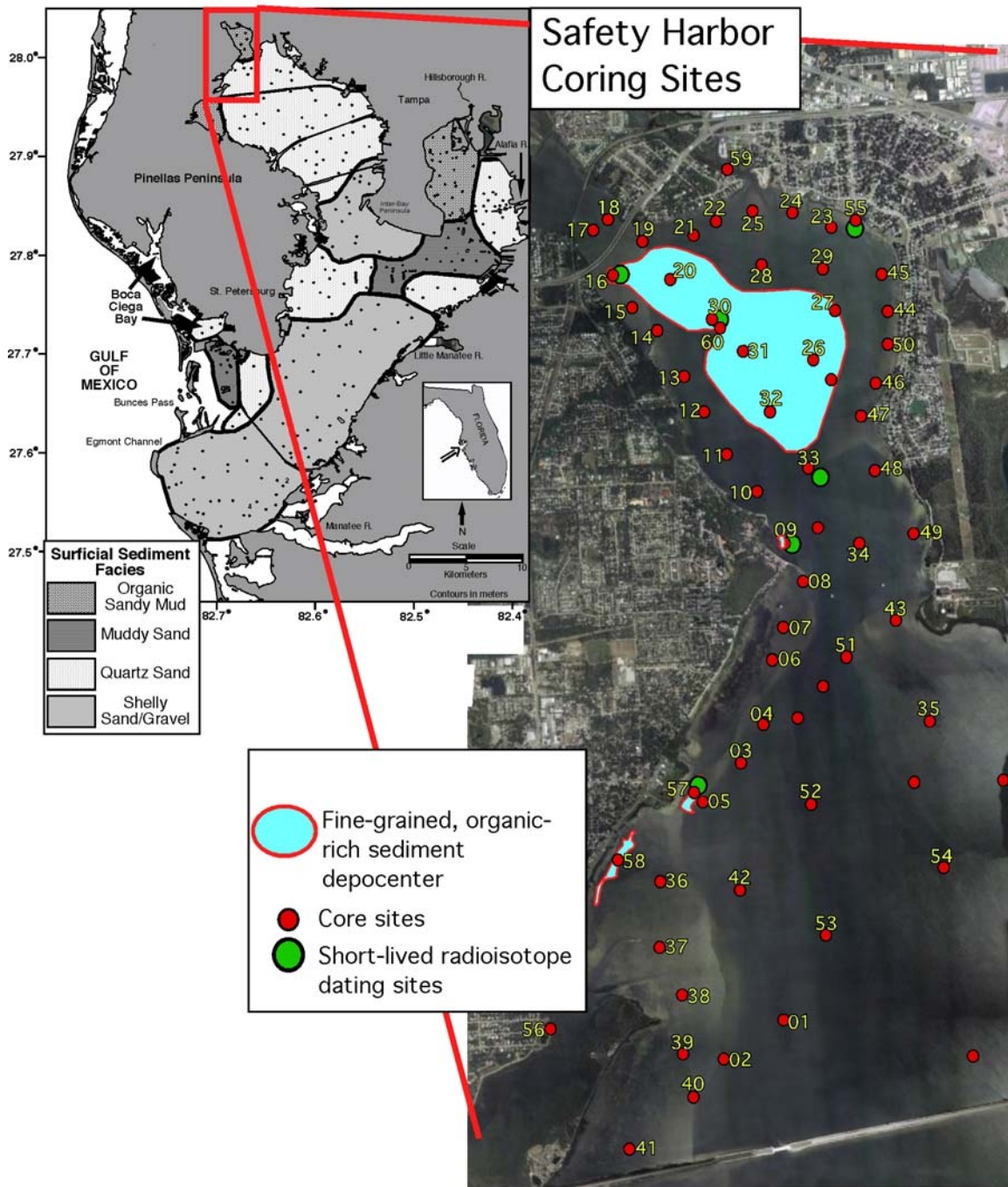


Figure 1. Map of Tampa bay showing surface sediment facies (upper left) and enlarged map of Safety Harbor study area with core sites and fine-grained, organic-rich sediment depocenters (modified from Brooks, in Press).

Table 1. Tampa Bay sediment facies (Brooks, In Press).

Facies		% Gravel	% Sand	% Silt	% Clay	% Mud	Mean Grain Size (f)	% CO3	% Organic Content (LOI)	Depositional Environment
OSM - Organic Sandy Mud	MAX	3.4	93.5	82.6	66.8	97.6	8.6	49.8	19.2	Estuarine
	AVG	0.1	36.0	34.5	29.3	63.8	5.9	20.6	9.3	
	MIN	0.0	2.4	1.4	1.7	3.1	1.6	1.3	4.0	
MS - Muddy Sand	MAX	14.1	89.8	45.4	44.4	89.8	7.5	74.6	3.9	Estuarine
	AVG	0.5	77.6	12.6	9.4	21.9	3.7	14.0	2.2	
	MIN	0.0	10.2	2.3	0.6	10.2	2.4	1.6	0.5	
QS - Quartz Sand	MAX	2.0	99.9	8.0	7.7	9.6	3.6	9.5	2.5	Estuarine
	AVG	0.1	96.6	1.6	1.7	3.3	2.7	2.8	0.7	
	MIN	0.0	90.2	0.0	0.0	0.1	2.0	0.5	0.1	
SSG - Shelly Sand/Gravel	MAX	91.6	99.7	12.0	5.7	12.3	3.2	94.1	2.7	Estuarine
	AVG	7.1	90.6	1.9	1.0	2.9	1.9	37.3	0.8	
	MIN	0.0	8.4	0.0	0.0	0.0	-1.5	10.4	0.2	
OM - Organic Mud	avg	0.8	33.1	38.4	27.7		5.7	30.6	13.1	Paralic
	max	9.7	81.9	86.1	70.0		8.6	87.6	54.7	
	min	0.0	1.0	11.1	1.7		3.1	5.0	2.9	
CM - Carbonate Mud	avg	2.1	19.7	51.7	26.2		5.9	77.0	13.0	Terrestrial/Lacustrine
	max	37.6	79.6	95.7	60.9		8.6	95.8	80.8	
	min	0.0	1.2	0.0	0.0		1.2	16.6	1.8	
OCM - Organic Carbonate Mud	avg	2.6	34.2	41.2	22.0		5.3	45.3	17.4	Terrestrial/Lacustrine
	max	37.1	94.8	66.5	55.8		8.1	84.6	57.5	
	min	0.0	3.8	0.0	0.0		2.0	3.2	3.4	
FQS - Fine Quartz Sand	avg	0.3	86.8	5.4	7.4		3.2	4.3	2.3	Terrestrial/Fluvial
	max	4.6	99.5	33.0	42.4		6.0	34.9	24.6	
	min	0.0	41.7	0.0	0.0		1.8	0.1	0.1	
SS - Shelly Sand	avg	18.5	69.7	6.3	4.8		2.0	48.6	2.1	Open Marine
	max	47.9	94.5	22.9	13.3		3.6	83.2	9.0	
	min	0.0	38.9	0.1	0.0		0.3	4.5	0.3	
SMS - Shelly Muddy Sand	avg	8.9	59.1	18.8	13.2		3.7	46.7	3.2	Open Marine
	max	39.4	87.1	38.3	29.6		5.7	76.2	7.2	
	min	0.0	40.6	6.2	1.8		1.2	15.8	0.9	
BGC - Blue Green Clay	avg	3.4	50.4	26.0	20.2		4.4	35.1	3.7	Carbonate Platform
	max	21.8	72.9	63.2	50.2		7.4	73.4	10.5	
	min	0.0	8.0	0.6	1.0		2.6	2.4	0.2	

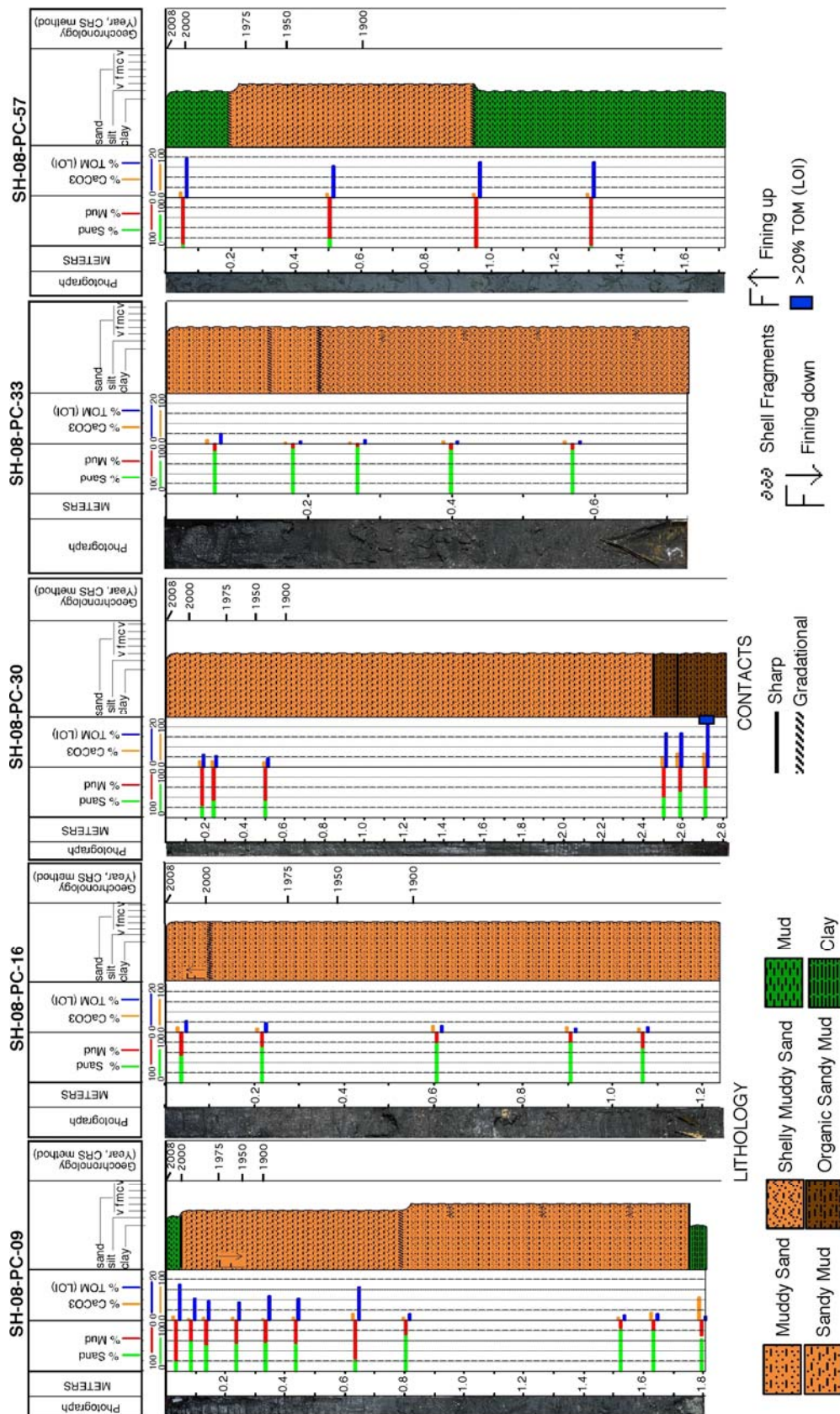


Figure 2. Core logs for the five focus sites. Refer to figure 1 for sites locations.

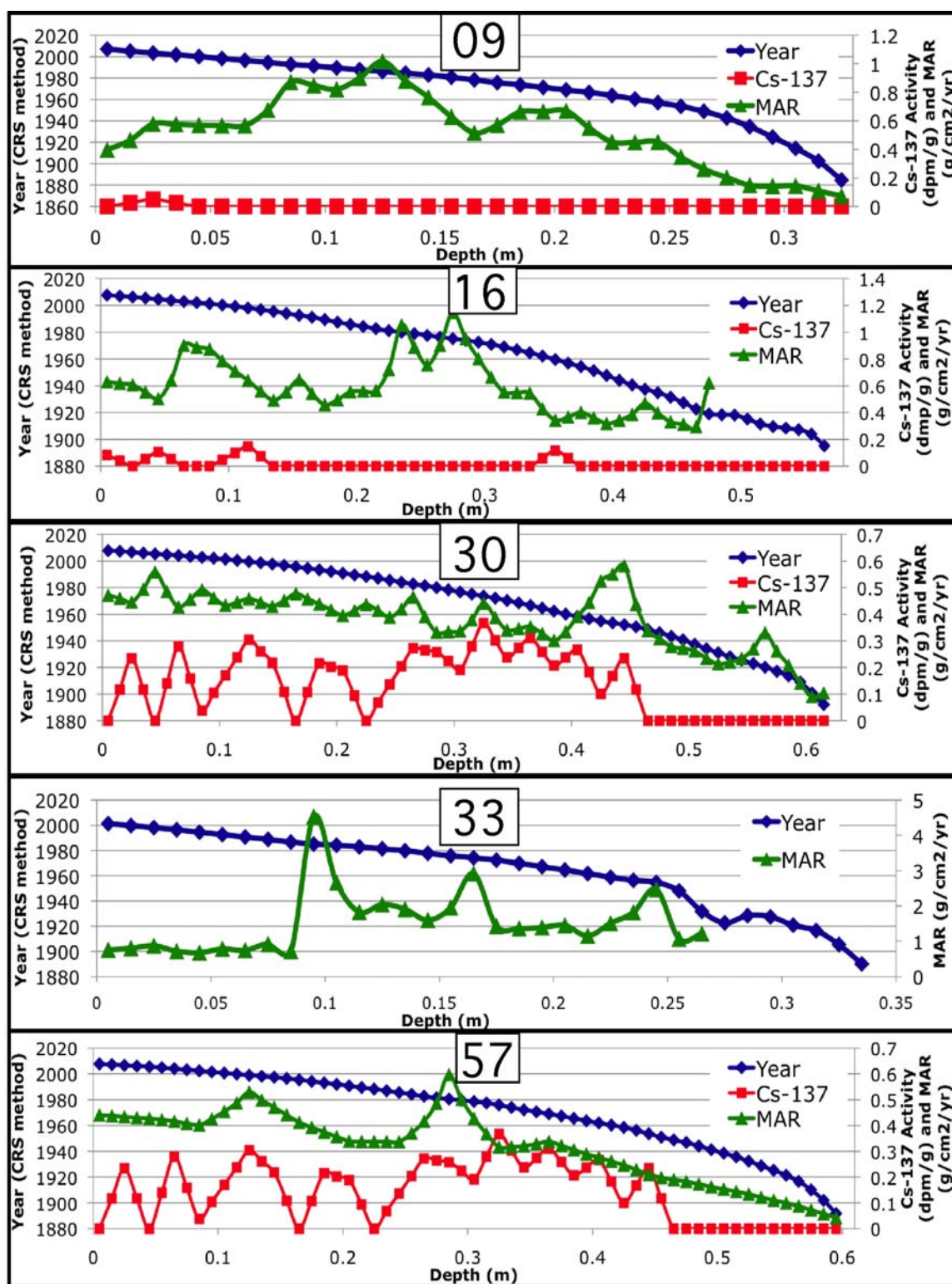


Figure 3. Results of CRS model, ^{137}Cs activity and MAR for the five focus sites.

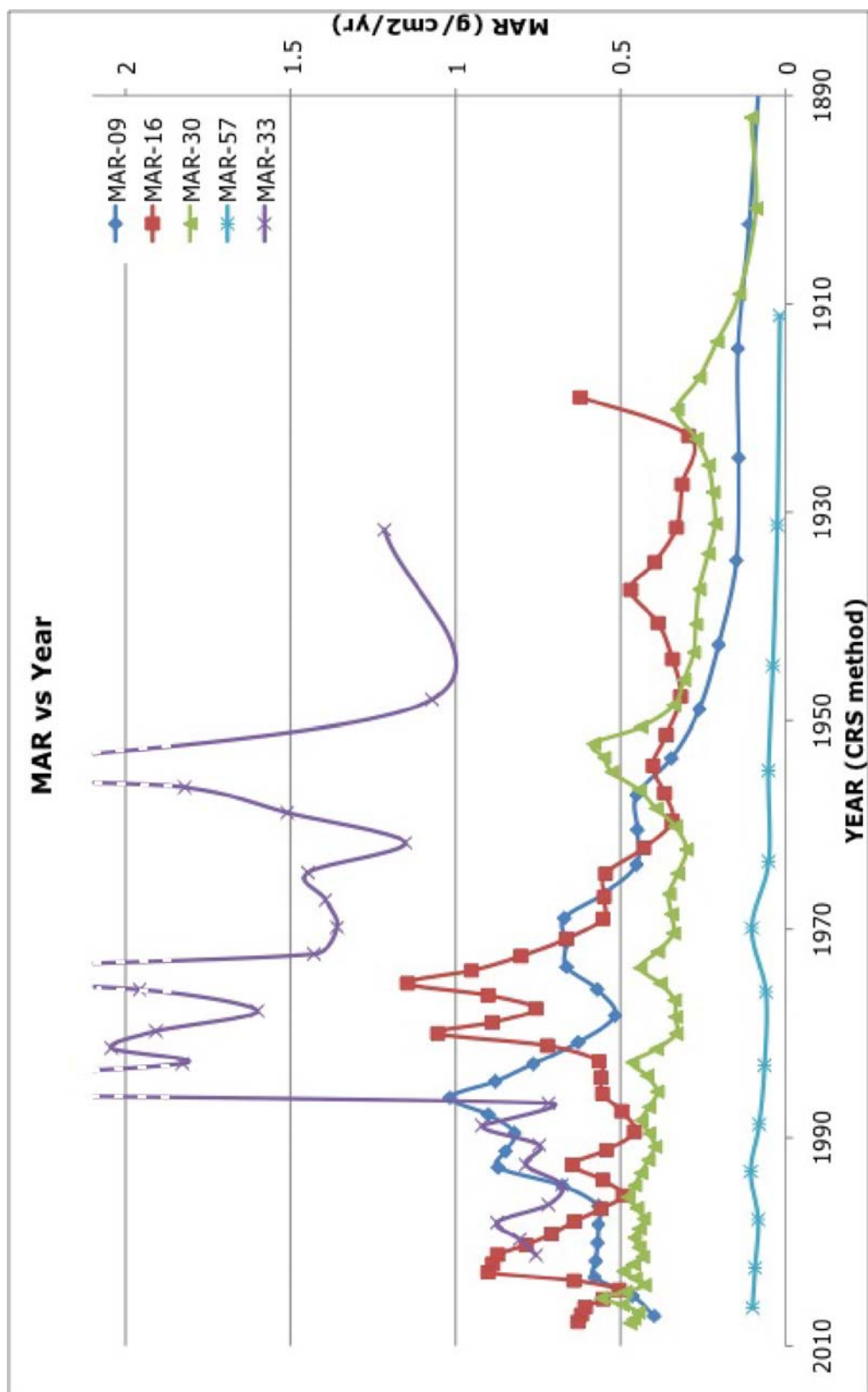


Figure 4. MAR for the five geochronology cores.

NITROGENOUS ORGANIC MATTER ACCUMULATION IN SAFETY HARBOR, FLORIDA: SOURCES AND DECADAL-SCALE TRENDS

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ABSTRACT

Microscopic and geochemical analyses of surface sediments and sediment cores were used to identify the source of organic-rich sediments that are reported to be accumulating within the Safety Harbor region of Tampa Bay. The geochemical analyses included organic C:N, total organic carbon, total organic nitrogen, stable isotopes of carbon and nitrogen, and analysis of decay-resistant plant biomarkers. All microscopic and geochemical analyses were consistent with microalgae being the predominant form of organic matter in the sediments, although vascular plants (e.g., mangrove, seagrasses) were common in nearshore locations and were present during some time periods in offshore locations. Microalgal sedimentation may occur either directly or indirectly as unassimilated material within fecal pellets that primarily originate from planktonic crustaceans. Hydrilla and other plants from Lake Tarpon were not substantial contributors to organic deposits in Safety Harbor. Likewise, sewage (whether treated or untreated) and animal feces were not dominant contributors to nutrient accumulation. Increasing trends in sediment accumulation rate, total organic carbon, and total organic nitrogen from the 1920s to around 1960 are consistent with the effects of bridge construction that started during the 1920s. Increased nutrient loading during this period also contributed to this trend. Since the 1960s, the area of Safety Harbor near the mouth of the Lake Tarpon Outfall Canal has experienced increased accumulation rates for sediment, total organic carbon and total organic nitrogen. Since the 1940s, sediments in Safety Harbor have been trending away from isotopic signatures associated with Tampa Bay microalgae and toward increased terrestrial sources, as indicated by carbon isotopes, and increased inorganic fertilizer sources, as indicated by nitrogen isotopes.

INTRODUCTION

The accumulation of soft, dark sediments (often referred to as “muck” by the public) in the Safety Harbor area of Old Tampa Bay has been perceived by local residents and government officials as a threat to the recreational opportunities, ecological health, and aesthetic appeal of the area. Initial acoustic mapping during the fall of 2006 and geochemical analyses of a single sediment core collected during 2002 confirmed the presence of a 2 m thick accumulation of fine-grained sediments in some parts of upper Old Tampa Bay, and determined that recent sediments are rich in organic matter and metals (Peter Swarzenski, USGS, unpublished data). Initial dating of material within the single sediment core using lead and cesium radioisotopes indicated the upper 40 cm of sediment corresponds to the last 100 years of deposition, with the sediment below this level being accumulated prior to 1900. This early work supported the public perception of ecosystem change, but raised questions about the source and history of the organic sediment accumulation.

During 2006, a collaborative investigation was facilitated by the Tampa Bay Estuary Program to determine if removal of organic-rich sediments was feasible and necessary for maintaining or improving ecosystem health. Before determining if the accumulated sediments should be removed, a better understanding of the source and history of the sediments was needed, which was the focus of this paper. The objectives of this project were therefore to provide interpretive information on the source and history of organic-rich sediments in the Safety Harbor area. Source was investigated using chemical fingerprints (C:N ratios, stable isotopes), and history was investigated by examining these characteristics within sediment cores that had been dated using radioisotopes (Brooks et al., these proceedings).

METHODS

Sedimentary organic matter in estuaries typically originates from multiple sources, including materials produced both locally and farther away in the estuarine watershed. To address concerns that decomposition of imported Lake Tarpon plant material was contributing to the accumulation of organic deposits in Safety Harbor, a stable-isotope survey was conducted along a transect extending from upper Lake Tarpon to an area of Safety Harbor near the lower end of the Lake Tarpon Outfall Canal (Figure 1). Leaf samples were collected from 12 vascular plant specimens, of which seven were *Hydrilla verticillata* (LT1, LT2, LT3, LT4), three were coontail (*Ceratophyllum* sp.) (LT1, LT2, LT3), one was cattail (*Typha* sp.) (LT1), and one was spatterdock (*Nuphar* sp.) (LT2). Surface sediments were collected from LT1, LT2, and SH using a Ponar benthic grab. Water was collected from all 5 sites for filtration of particulate organic matter (POM, primarily phytoplankton). The vascular plants, surface sediments, and POM were collected on October 26, 2007. Additional surface sediment samples were collected on April 25, 2008.

Upon returning to the lab, surface sediment samples were examined under light microscopy and photographed. Portions of these samples were rinsed with mild acid (0.5N HCl) to remove carbonates, rinsed with de-ionized water, dried at 50°C and homogenized. Plant samples were rinsed with de-ionized water, dried at 50°C, and then ground into a fine powder in preparation for isotopic analysis. Water samples were filtered through 0.7 µm GF/F glass fiber filters by vacuum filtration. The material retained on the filters (POM) was dried at 50°C, and dried filters were scraped to remove the organic matter along with the top layer of the GF/F filter. The scraped samples were homogenized and packed into aluminum boats and stored in a dessicator for subsequent isotopic analysis.

Stable-isotope measurements of all organic samples were obtained using a continuous-flow Finnigan MAT Delta Plus isotope ratio mass spectrometer (IRMS) coupled to a Carlo Erba elemental analyzer. Weighed samples were introduced via an autosampler into the combustion furnace of the elemental analyzer, which was set at 1050°C. Flash combustion converted the nitrogen and carbon in the samples to pure N₂ and CO₂, which were eluted from a gas chromatograph column and carried by a stream of helium gas to the mass spectrometer, where the ¹⁵N and ¹³C abundances were measured based on their unique mass-to-charge ratios.

Natural abundances of each stable isotope are expressed as per mil (‰) units using delta notation, e.g. $\delta^{15}\text{N} = [(R_{\text{sample}}/R_{\text{standard}}) - 1] \times 1000$, where $R = {}^{15}\text{N}/{}^{14}\text{N}$. Elemental compositions and ratios (% total organic carbon, % total organic nitrogen and C:N) were measured prior to delivery of combustion products to the IRMS.

To determine history of organic-rich sediments, four push cores were analyzed: cores 09, 16, 30, and 33 (Figure 1). The four core sites were from areas known to experience organic deposition that had established geochronologies (Brooks et al., these proceedings). Core 09 was collected near the west shore of Safety Harbor, adjacent to Philippi Point in shallow water. Core 16 was nearest the Lake Tarpon Outfall Canal and is the most landward of the four cores in terms of its relative position on the principal estuarine axis. Core 30 was collected near the middle of a large deposition center in central Safety Harbor. Core 33 was the most seaward of the four cores, and was not located in the large deposition center. Isotopic and elemental analyses were conducted on a weighed aliquot of the bulk sediment collected at 2 cm intervals along the core.

RESULTS AND DISCUSSION

Source—Microscopic examination of surface sediments and geochemical analyses both indicate that the source of organic-rich material deposited in Safety Harbor is primarily derived from microalgae rather than vascular plants, but there were local exceptions. Microalgal sedimentation

may occur either directly or indirectly in the form of unassimilated microalgal organic matter within fecal pellets that originate from primarily planktonic crustaceans (Figure 2). Centric diatoms (not visible in Figure 2), a type of planktonic microalgae that originates in the water column, were visible in surface sediments from northwest Safety Harbor, but generally contributed much less to sediment bulk than did crustacean fecal pellets. Throughout this report, the term “microalgae” is often used instead of “phytoplankton” to acknowledge the fact that both phytoplankton and benthic microalgae occurred in the sediments.

Evidence from microscopic examinations prompted the collection of additional surface samples (Ponar grabs) from the Palm River dead zone and the mouth of the Alafia River; two sites that are known to experience accumulation of organic-rich sediments. These samples also contained a predominance of crustacean fecal pellets and microalgae and were generally similar in composition to the Safety Harbor sediments. Crustacean fecal pellets are a dominant surficial sediment component in the northern Gulf of Mexico dead zone (Dagg et al. 2007), but may also be dominant in apparently healthy estuarine ecosystems (Peebles and Flannery 1992).

Stable isotope data from the Lake Tarpon transect are presented in Figure 3, which identifies distinct differences in the composition of the various types of organic matter. The carbon and nitrogen in the hydrilla are more isotopically enriched (larger values) than Lake Tarpon phytoplankton (as particulate organic matter, POM) and are also isotopically distinct from the organic-rich sediments of upper Safety Harbor. The broad range of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values for Lake Tarpon plants (vascular plants and POM) likely reflects differences in the biological processes controlling photosynthetic isotopic fractionation as well as the sources of dissolved nitrogen and carbon supporting plant biomass in the lake. Compared with Lake Tarpon plants, the $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ compositions of the surficial sediments from Safety Harbor (SH site in Figure 1) are invariant, with Safety Harbor sediment values centered on -23‰ for carbon and +3.5‰ in nitrogen. Safety Harbor sediment samples collected on different dates are in good agreement; the well-constrained values from Ponar grab samples correspond very well with the surface isotope values from sediment cores. These values are within the isotopic ranges typically observed for microalgal-dominated material from coastal settings, with the carbon values being somewhat depleted, suggesting a terrestrial influence.

The differences in $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values between organic materials from Lake Tarpon and surface sediments from Safety Harbor indicate neither vascular plants nor phytoplankton from Lake Tarpon were direct contributors to the sedimentary organic material in Safety Harbor. If the Safety Harbor sediment values (from Ponar grabs and cores) resulted from mixing of particulate organic matter—such as mixing of vascular plant detritus from different sources, then any irregularities in the mixed particulate ratios would create variation in the isotope values. Instead, the observed uniformity in the Safety Harbor sediment isotopes indicates dissolved carbon and nitrogen were well-mixed prior to incorporation into organic matter, as would occur in the open waters of a tidally and wind-mixed embayment such as Safety Harbor. Together, the uniformity and values of the carbon and nitrogen isotopes in organic-rich surface sediments indicate the microalgal organic material grew in place rather than being imported. In other words, nutrients and DIC imported into the Safety Harbor area from the Lake Tarpon Outfall Canal and the surrounding watershed become mixed with bay water before being incorporated into the microalgal biomass that then enters the sediment directly or as crustacean fecal pellets.

History—In each of the sediment cores from Safety Harbor, the percentage of total organic carbon (TOC) and total organic nitrogen (TON) has increased by a factor of 3-4x that of the beginning of the record, approximately 100 years before present (Figures 4 and 5). The patterns in Figure 4 and Figure 5 suggest a possible two-phase increase. The first increase starts in the 1920s and appears to be gradual. The second increase is more substantial, occurring in the late

1960s and continuing to the time of core collection. A two-phase increase in TOC and TON is consistent with the building of the Gandy Bridge and Courtney Campbell Causeway (1922-1934) and later construction of the Lake Tarpon Outfall Canal (1967-68).

Discharge of water from the Lake Tarpon watershed into Safety Harbor could have caused (1) an increase in terrestrially-derived dissolved nutrients that in turn stimulated local microalgal production in the Safety Harbor area, as indicated by the isotope transect data, and (2) direct introduction of particulate organic material directly into Safety Harbor. Several lines of geochemical evidence from the sediment core analyses suggest the first scenario is dominant. Further discussion of this follows.

One line of evidence is provided by the ratio of carbon:nitrogen (C:N) in the sediment cores. Terrestrial and emergent plants commonly have higher C:N of around 30-40, whereas microalgal and bacterial organic matter have C:N ratios of around 8-12. Submerged aquatic vascular plants (such as hydrilla) are intermediate, having C:N values around 20-30. At all sites examined, C:N had an overall trend in the range of 6-15 with a tendency to decrease and become more variable over time (Figure 6). This suggests a decreasing influence of terrestrial and emergent plants and an increasing influence of microalgae or bacteria, but could also reflect a preferential loss of microalgal material in the sediments over time (i.e., through diagenesis and recycling into the water column).

The ranges of C:N ratios in both landward and seaward sites clearly indicate that microalgal derived organic matter is the dominant biological source contributing to the sedimentary organic matter. Hydrilla and emergent plants from Lake Tarpon yielded expected C:N ratios, ranging from 20 to 25 (hydrilla) and >35 (other vascular plants). These values are substantially higher than those observed in all of the sediment cores, indicating that submerged and emergent vascular plants were not substantial sources of organic matter in the open waters of Safety Harbor (i.e., locations represented by Cores 16, 30, 33 and 9).

However, the landward site (Core 16) had higher C:N values, suggesting a greater relative, but small, contribution by vascular plants, probably due to greater proximity to sources of terrestrial plant debris. The highly variable C:N values observed in the most recent times likely reflects the variable inputs of both locally produced microalgal biomass and terrestrial plant debris.

Another line of evidence is provided by stable isotopes in the cores. $\delta^{13}\text{C}$ deposited in open-water cores (Cores 16, 30 and 33) during pre-anthropogenic times ranged between -20 and -21.5‰, values that are typical of microalgae that use dissolved inorganic carbon (DIC, primarily CO_2) from Gulf of Mexico waters (Figure 7). Sedimentary $\delta^{13}\text{C}$ isotopic values have been decreasing over time and have also (like C:N) become more variable in recent decades. The declining $\delta^{13}\text{C}$ isotopic trend observed in all cores is consistent with an increasing contribution of terrestrially-derived DIC. This decrease appeared to stop after the 1960s (consistent with construction of the Lake Tarpon Outfall Canal), suggesting a regime change in the dominant source of DIC. However, in the most recent times (post 2000), the $\delta^{13}\text{C}$ values became highly variable, with the landward site showing far more terrestrial influence, as might be expected. It is important to note that microalgal growth rates and light limitation also strongly affect ^{13}C fractionation, resulting in enriched values during periods of fast microalgal growth or increased light limitation, such as light limitation resulting from increased concentrations of colored dissolved organic matter or suspended particles associated with inflow events.

In comparison with $\delta^{13}\text{C}$ values, the $\delta^{15}\text{N}$ composition of sedimentary organic matter exhibited lower variability over time with only a slight trend towards more depleted values in recent times (Figure 8—note the small range of the x axis). The relatively constant and comparatively ^{15}N -

depleted values of the sedimentary organic matter (2-4 ‰) suggest that either nutrient sources have not changed substantially or that the changing nutrient sources had offsetting $\delta^{15}\text{N}$ compositions, with the latter being much more likely.

Figure 9 presents a comprehensive summary of the stable isotope data for open Safety Harbor waters (Cores 16, 30, and 33). The representation of Tampa Bay microalgae in Figure 9 is a representation of present-day phytoplankton isotopes from Middle Tampa Bay. Prior to anthropogenic additions of 1) sewage waste from humans, cattle, and other domesticated animals, 2) NO_x from combusted fossil fuels, and 3) commercial fertilizers synthesized using the Haber-Bosch process (which transforms atmospheric N_2 into nitrate and is therefore isotopically depleted), fixation of atmospheric nitrogen by blue-green algae and lightning would have been much more important as sources of fixed nitrogen, pushing nitrogen isotope values toward 0‰. Mixture of bay waters with Gulf of Mexico waters, combined with additions of waste from native aquatic life, birds, terrestrial animals, and the relatively small Native American population, would have driven these isotopes to slightly higher values. The observed pre-anthropogenic nitrogen isotope values of ~2-4‰ (Figure 8) are therefore within the expected range. As the Tampa Bay region became more developed during the first half of the 20th century, land was cleared and exposure to soils increased. Inputs from human and cattle waste also increased. All of these factors would have driven nitrogen isotope values higher, strengthening the coastal nitrogen isotope gradient that is observed today, where microalgal $\delta^{15}\text{N}$ is ~3-4‰ in open Gulf of Mexico waters, ~4-5‰ in Middle Tampa Bay, and >5‰ in the parts of Tampa Bay that are influenced by nitrogen derived from sewage (D. Hollander, unpublished data). This coastal nitrogen gradient can be much steeper in places—nitrogen isotopes in waters that are heavily influenced by the modern reclaimed water system have been measured at 27‰ by Elon Malkin (USF).

Since the 1940s, sediments have tended to shift slightly toward $\delta^{15}\text{N}$ values typically associated with residential fertilizers derived from the Haber-Bosch process (~2‰, Figure 9). There was a linear decrease in Core 33 $\delta^{15}\text{N}$ over time (Pearson's $r = 0.48$, $p = 0.01$), and trends in Cores 16 and 30 were better described by quadratic curves than by linear trends, with both cores indicating peak $\delta^{15}\text{N}$ values during mid-century. Regardless of trends over time, the consistently low $\delta^{15}\text{N}$ values that occurred throughout all cores clearly indicate that sewage and animal waste is not dominating nitrogen accumulations in Safety Harbor. Sewage and animal waste may contribute to the total fixed nitrogen in the Safety Harbor area, but this quantity is overwhelmed by nitrogen from depleted $\delta^{15}\text{N}$ sources such as fertilizer.

Since the 1920s, $\delta^{13}\text{C}$ has become substantially depleted (Figure 7), which is indicative of increased terrestrial DIC influence (-23‰). This trend is coincident with bridge/causeway construction, which would have caused more terrestrially-derived organic material to be retained within Safety Harbor and other parts of Old Tampa Bay.

CONCLUSIONS

Hydrilla and other plants from Lake Tarpon are not substantial contributors to organic deposits in Safety Harbor. Likewise, sewage (whether treated or untreated) and animal feces are not dominant contributors to nutrient accumulation in Safety Harbor. Microscopic and geochemical analyses indicate the source of the organic material in the areas of concern is primarily microalgae rather than vascular plants, but there are local exceptions. The microalgal sedimentation occurring in the depositional areas may occur either directly or indirectly as crustacean fecal pellets.

The increase in accumulation of organic material of microalgal origin is likely associated with increased accumulation of nutrients (eutrophication) in the Safety Harbor area. Nutrient accumulation is the net effect of nutrient loading, nutrient sedimentary fluxes, and nutrient losses

due to bay circulation (wind and tide), and other processes. Over past decades, sediments in Safety Harbor have been trending away from isotopic signatures associated with Tampa Bay microalgae and toward increased terrestrial sources (carbon) and increased inorganic fertilizer (nitrogen) sources.

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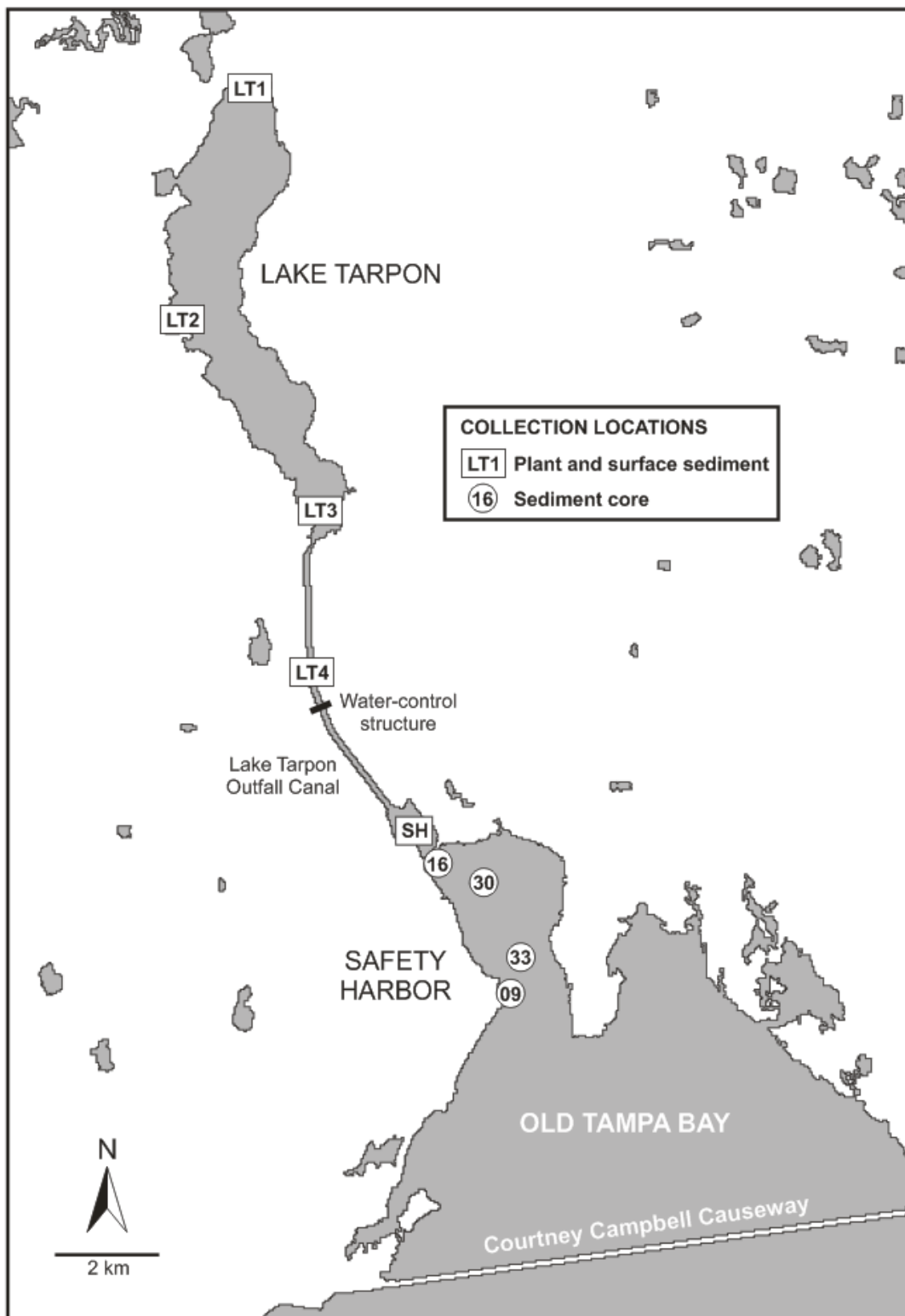


Figure 1. Locations sampled for stable isotope comparisons of plant life and sediments. A water-control structure is located between sites LT4 and SH that allows periodic, one-way flows of fresh water to Safety Harbor.



Figure 2. Photomicrograph of sediment sample obtained by Ponar grab from upper Safety Harbor (site SH in Figure 1). Opaque, brownish, oblong particles are crustacean fecal pellets. The opaque, amorphous material scattered in the background was isolated by filtration and found to be isotopically indistinguishable ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$) from the larger particles; the amorphous material appears to be degraded crustacean fecal pellets (E. Peebles photo).

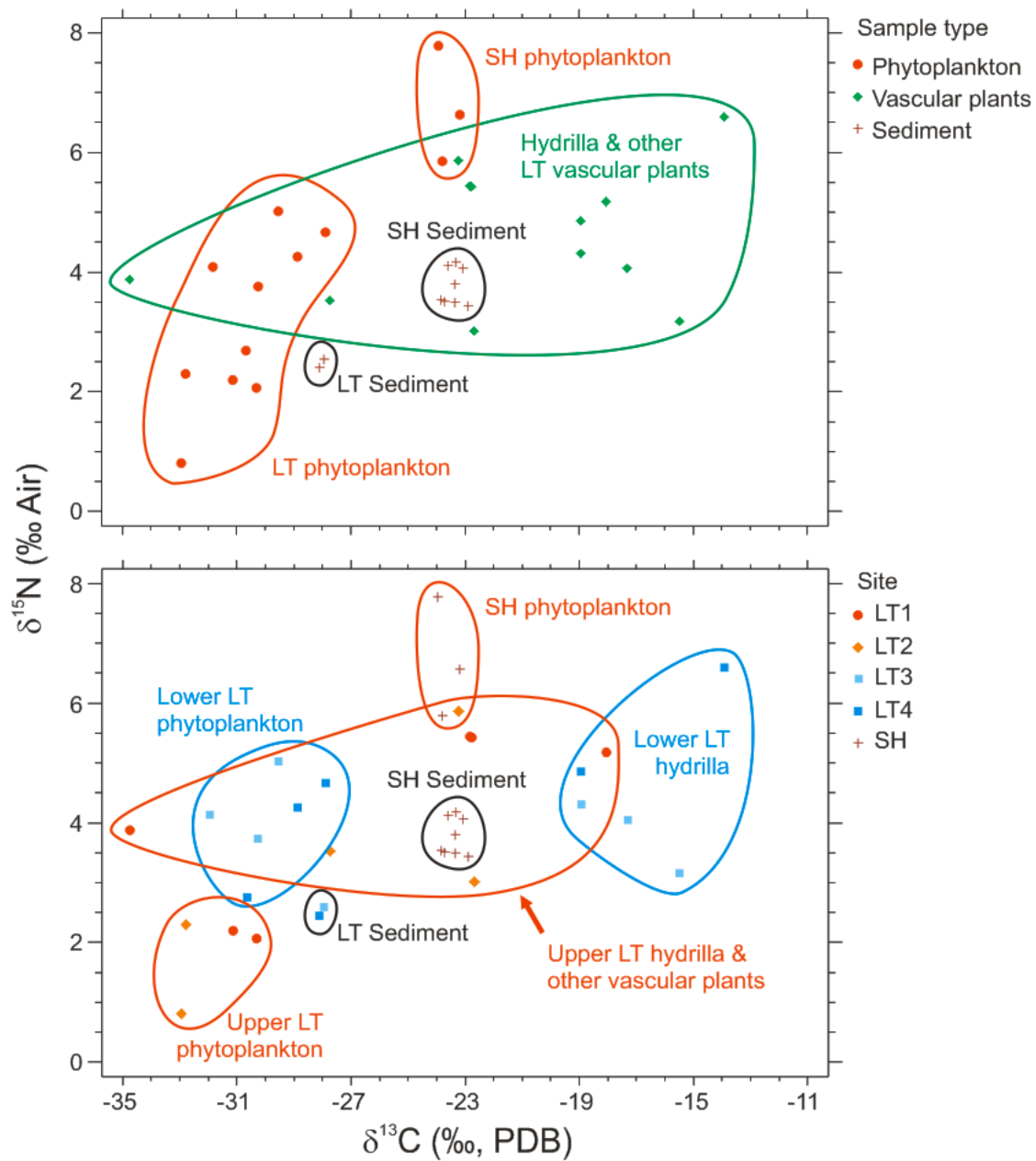


Figure 3. Carbon and nitrogen isotopes for organic material from Lake Tarpon (LT) and Safety Harbor (SH). Each data point is the average of two replicates. The two panels differ only in the manner in which groups are identified.

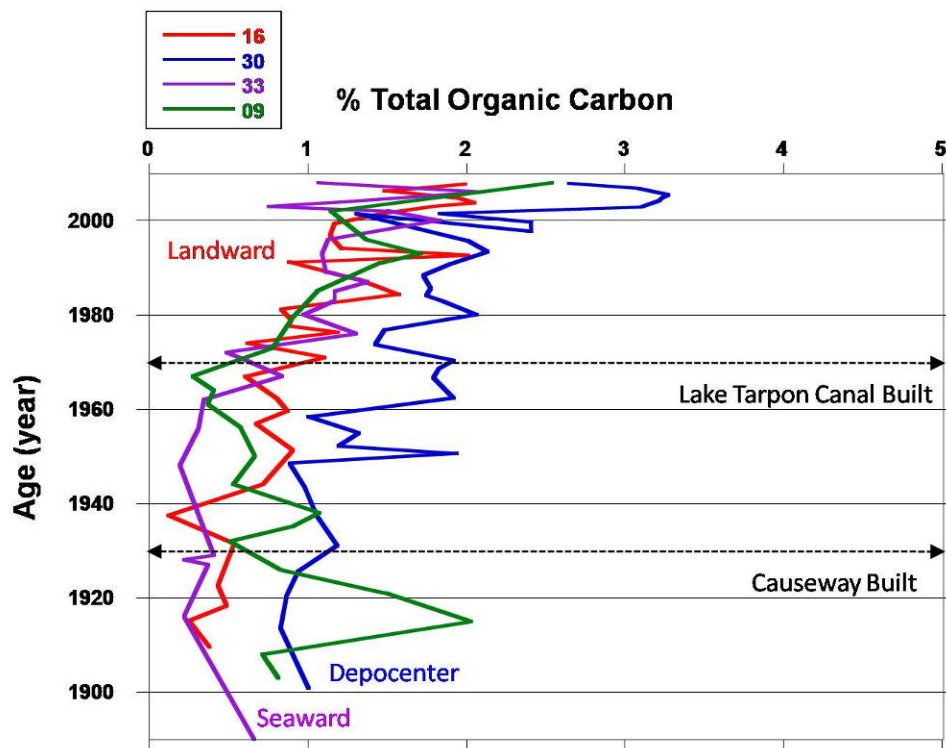


Figure 4. Percent total organic carbon vs. sediment age for cores 16, 30, 33 and 9.

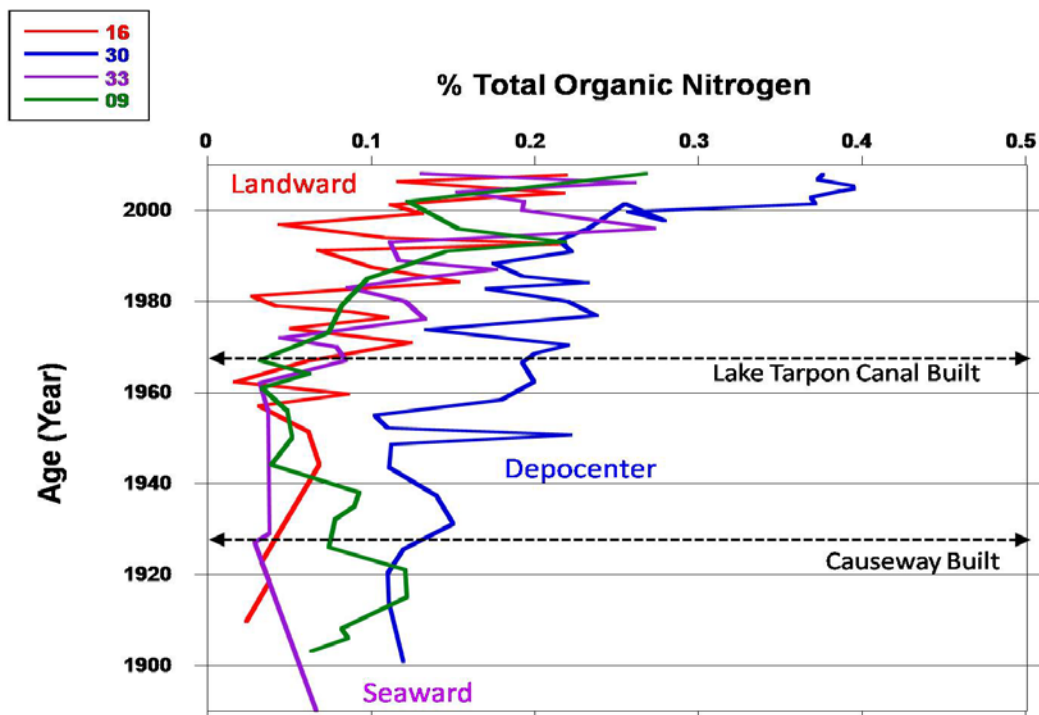


Figure 5. Percent total organic nitrogen vs. sediment age for cores 16, 30, 33 and 9.

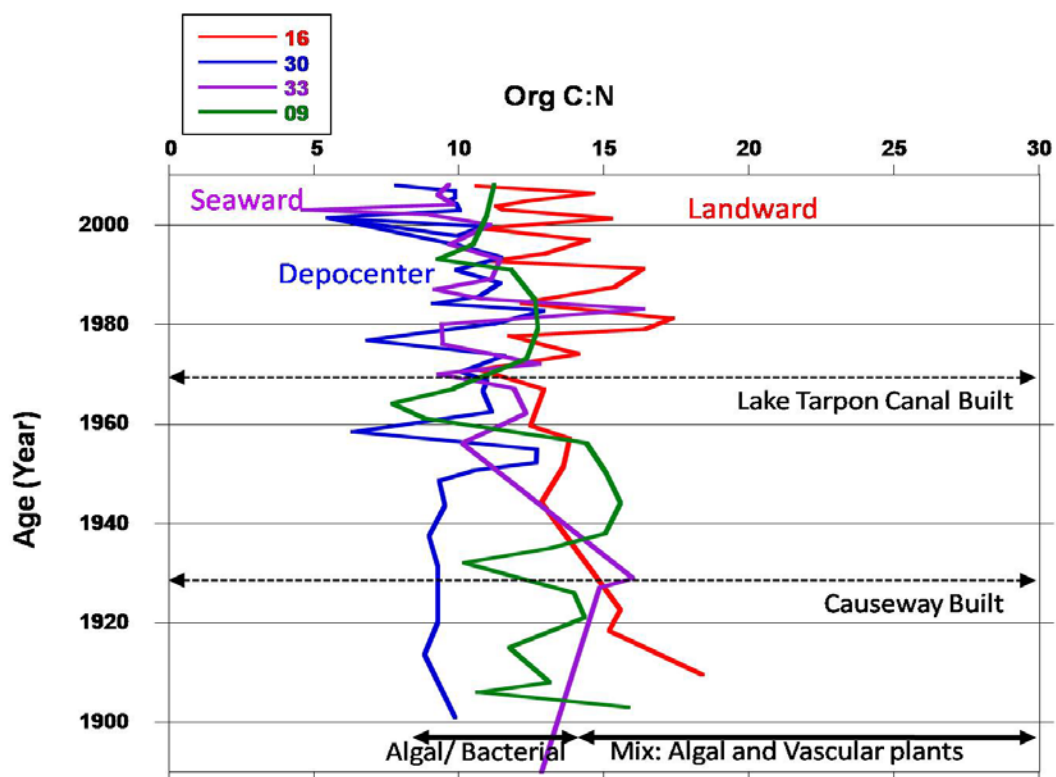


Figure 6. Organic carbon to nitrogen ratios (C:N) vs. sediment age for cores 16, 30, 33 and 9.

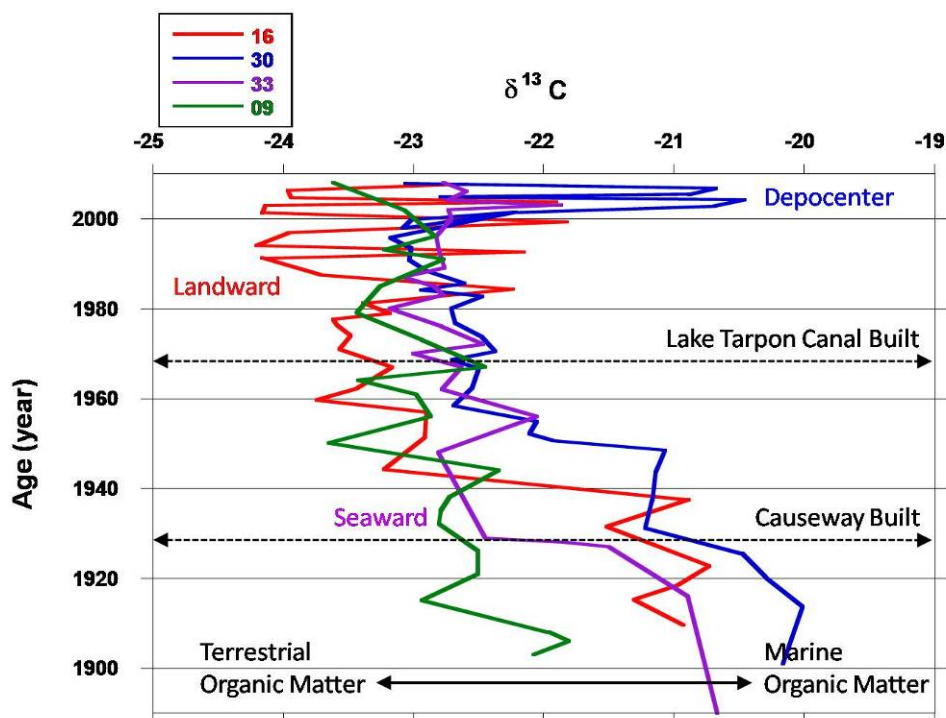


Figure 7. Carbon isotopic composition ($\delta^{13}\text{C}$) of organic matter vs. age for cores 16, 30, 33 and 9.

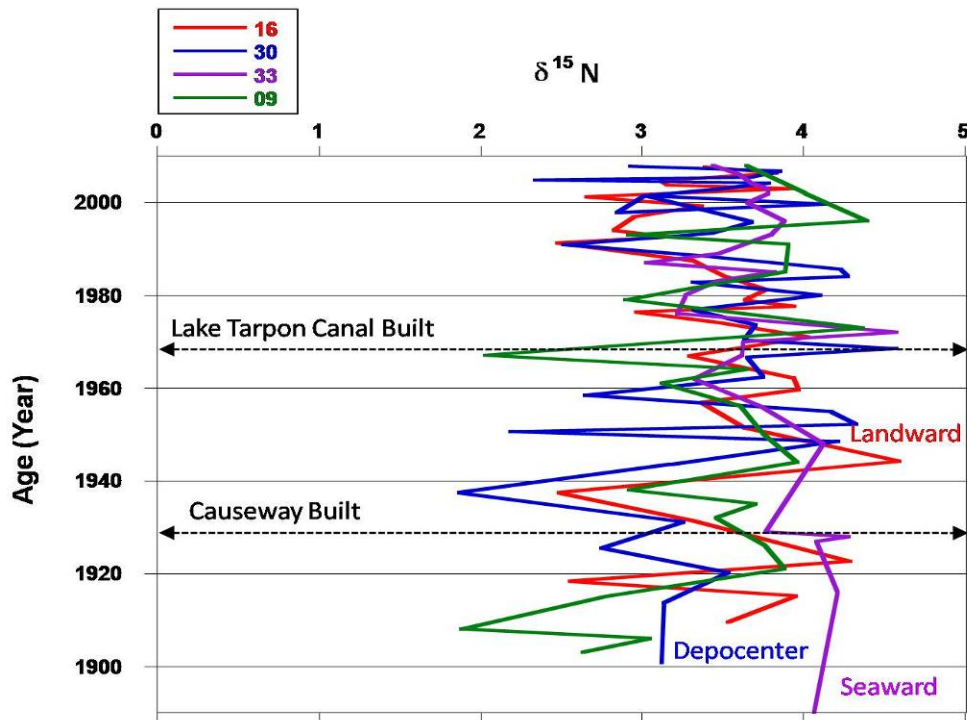


Figure 8. Nitrogen isotopic composition ($\delta^{15}\text{N}$) of organic matter vs. age for cores 16, 30, 33 and 9.

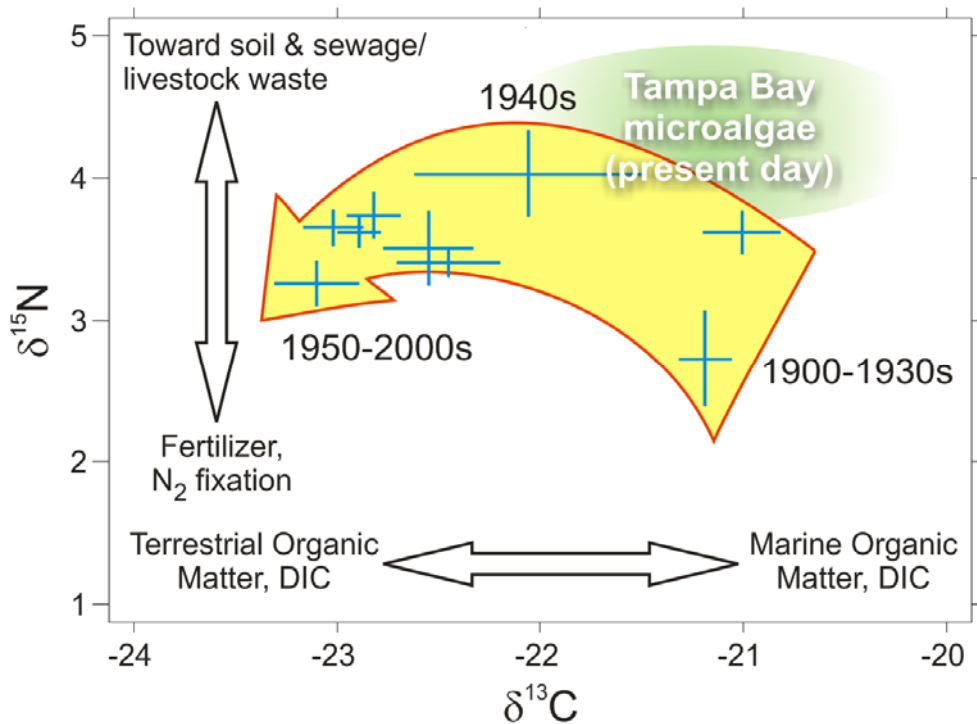


Figure 9. Comparison of mean decadal trends in central Safety Harbor core isotopes (decadal means with standard error bars for Cores 16, 30, and 33). Decades 1900s, 1910s, and 1920s were grouped due to low number of observations. The isotopic ranges indicated for Tampa Bay microalgae are based on present-day values from Middle Tampa Bay (D. Hollander, unpublished data).

HISTORICAL BATHYMETRIC ANALYSIS OF TAMPA BAY

P. Julian & E.D. Estevez

ABSTRACT

Estuarine depth is the result of coastal sediment budgets that involve terrestrial, marine, and autochthonous sources and sinks. Through the last century Tampa Bay's bathymetry has been altered drastically. Our findings corroborate and add to a previous study which determined that the mean depth of Tampa Bay has increased by more than 5% in the past century. However at finer scale, some areas of the Tampa Bay estuary have become more shallow, such as Boca Ciega Bay and a special study area labeled Feather Sound. Whether the depth of Tampa Bay will change as a result of climate change is an open but important question. Changes in the depth characteristics of Tampa Bay could affect multiple facets of the hydrodynamic system within Tampa Bay including water quality, quantity and circulation. In addition, alteration of depth has the potential to change sediment type and chemistry, the dispersion, abundance, or productivity of seagrasses, and possibly benthic communities including intertidal oysters.

INTRODUCTION

In Florida, approximately 388 to 518 square kilometers of marshland, tideland, and estuarine habitat have been lost to dredging and filling. Both new dredging and maintenance dredging exhibit the potential to damage biological resources (seagrass, benthic fauna, etc.), degrade water quality through sediment and contaminant re-suspension (Johnson 1981), and change sediment type and sediment chemistry (Jones and Candy 1981). Estuaries are altered commonly by dredging to accommodate vessel traffic (Schoellhamer 1996).

Modifications of the Tampa Bay estuary to facilitate coastal development, including port construction, have resulted in the excavation or filling of 44% of the emergent coastal wetlands (i.e. tidal marshes and mangrove forests) (Lewis et al. 1998). The Tampa Bay Estuary has from 1950 to 1980 lost 40% of its seagrass, a consequence of dredging activities, re-suspension of sediment, and nutrient-induced algal blooms. Seagrass serves as a protective nursery to fish and shellfish, as food for manatees, and as a stabilizer of bottom sediment (Poor et al. 2001).

This paper investigates the change of Tampa Bay's bathymetry over the last century via bathymetric sounding data, both historic (1879) and recent (2000). In the past, the dynamics of dredging events for increased access of commercial exports and imports had altered the bottom of the estuary and contributed to changes in vegetation and water quality. In the future, an improved understanding of bay depths and patterns of bathymetry changes will inform assessments of climate change impacts, especially changes in sea level.

MATERIALS AND METHODS

Tampa Bay Management Areas—For this study Tampa Bay was delineated into six management areas including Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay, Lower Tampa Bay, Terra Ceia Bay and Boca Ciega Bay (Figure 1). Each management unit possesses unique physical and chemical characteristics influenced by the major and minor tributaries flowing into these management areas. The major tributaries include the Alafia, Hillsborough, Little Manatee and Manatee rivers and several small streams comprising the minor tributaries which contribute approximately 70% of the annual freshwater inflow (Goodwin 1987).

General Study Area—The Tampa Bay ecosystem includes 967 km² of primarily unvegetated or sparsely vegetated estuarine waters with an average depth of 3.5 m, 72 km² of emergent coastal wetlands, and a 5700 km² watershed. Approximately 10% of the 967 km² of open water area (101 km²) has shallow (<2 m) shelves vegetated with seagrasses (Lewis et al. 1998).

Tampa Bay's historical circulation and transport processes, has been considerably modified within the past century, mainly due to dredging of ship channels, construction of major causeways, and usage of the estuarine sediments as shoreline fillings (Santschi et al. 2001).

Areas of Special Interest—Areas of interest were selected to reflect contemporary management concerns for issues related to sedimentation or erosion, and submerged aquatic vegetation (SAV). Areas of interest included an area within Feather Sound, the Southern Interbay Peninsula (labeled MacDill AFB), and a system of offshore bars along the southeast bay shore (labeled Long Bar).

The Feather Sound site is located in Old Tampa Bay on the central eastern shore of the Pinellas peninsula, between the Courtney Campbell causeway and the Gandy causeway with the Interstate 275 causeway intersecting a portion of the study area. The MacDill AFB study site is located at the southernmost tip of the Interbay Peninsula dividing Old Tampa Bay and Hillsborough Bay. The final study location, Long Bar was divided into three units, North Long Bar, Middle Long Bar and South Long Bar respectively, which are located on the eastern shoreline of the Middle Tampa Bay management unit (Figure 1).

Bathymetry Data—The 1879 bathymetry data consists of depth measurements from an 1879 hydrographic chart of Tampa Bay. The chart was created by the U.S. Coast and Geodetic Survey, Chart No. 77. A digital copy of the chart was obtained from the Office of Coast Survey, National Ocean Service, National Oceanic and Atmospheric Administration (NOAA). The digital copy was then georeferenced and depth soundings were digitized and attributed with depth values. Depths were referenced to mean low water (MLW) (USGS web access: <http://gulfsci.usgs.gov/library/index.html>) The historic 1879 depth data were imported into ArcGIS as point data and interpolated by kriging using ArcGIS Geo-Statistical Analysis (ESRI 2009).

The 2000 bathymetry data were selected with a GIS query procedure that applied spatial and temporal filters to the 47 digital NOAA hydrographic surveys, dating from 1945 to 1996, which cover the Tampa Bay region. Approximately 600,000 soundings were transformed from multiple orthometric and tidal vertical datums to a common vertical reference, the NAD83 ellipsoid (GRS80) (USGS web access: <http://chartmaker.ncd.noaa.gov/bathytopo>). The data were further processed using NOAA's VDatum Software (Ver 2.2.7, NOAA, Washington D.C., USA) to convert the vertical datum from NAVD 88 to MLW during the current tidal epic.

A difference analysis was performed using ArcGIS Spatial Analyst (Ver 9.3, ESRI, Redlands, CA, USA) to determine how Tampa Bay bathymetry has changed between the time periods. ArcGIS Spatial Analyst zonal statistics (ESRI 2009) were used for management areas and study locations with Tampa Bay (Figure 1).

Statistical Analysis—To determine statistical significance of depths between years and within study areas, non-parametric Wilcoxon 2-Sample Test, Normal Approximation was used. The critical level of significance was set at $\alpha = 0.05$. Results are reported as mean $\pm$ standard error where appropriate. All statistical operations were performed with JMP® (Ver 7.0.4, SAS, Cary, NC, USA).

RESULTS

No statistical analysis was performed on Tampa Bay management areas, but qualitatively, Boca Ciega Bay shallowed. Lower and Middle Tampa Bay, Hillsborough Bay, Old Tampa Bay, and Terra Ceia Bay deepened (as mean depth at MLW) (Figure 2 and Table 1).

It was determined using Wilcoxon non-parametric statistics (Tables 2 and 3) that the only area of special interest that was not significantly different between survey periods was the MacDill AFB site. The Long Bar system (Long Bar North, Middle and South) was determined to have deepened significantly and the Feather Sound site had significantly shallowed.

DISCUSSION

Tampa Bay's benthic landscape has been altered drastically through dredging operations aimed at improving the bay for large maritime operations, as evident in the 2000 bathymetry (Figure 1). A major shipping channel has been dredged from the mouth to the upper reaches of Lower Tampa Bay, where it splits into two branches, one entering Old Tampa Bay and the other going into Hillsborough Bay (Galperin et al. 1991). Another factor responsible for overall bay deepening has been the loss of shallow, near-shore environments caused by coastal development (Goodwin 1984). In addition, estuarine depths vary due to inputs of terrestrial and marine sediments, autochthonous biogenic production, natural and anthropogenic changes to inflows, tidal characteristics, bottom profiles, and sea level rise.

A comparison of historical (1879) and modern (2000) depth data registered to mean low water (MLW) allowed an assessment of depth changes for the entire bay, bay management areas, and areas of special interest (Feather Sound, MacDill AFB, and Long Bar). It was determined that the only area of special interest that was not different between the two surveys was MacDill AFB, which could be caused by the variability of sediment fluxes from Old Tampa Bay, and Hillsborough Bay. The MacDill AFB site may also serve as a point of accumulation due to altered bay circulation. Feather Sound was determined to be significantly shallower, which could have been sheltered from strong circulation forces within Old Tampa Bay due to the three causeways constructed within Old Tampa Bay, and would result in a low degree of sediment transport and deposition. The Long Bar sites were determined to be significantly deeper between the two surveys, which could have been caused by dredging, management activities, and altered bay circulation from earlier dredging activities which could cause altered sediment fluxes. However, Hearn (these proceedings) has also resolved annual to decadal migrations of longshore bar systems in the Terra Ceia Aquatic Preserve, south of the Long Bar system we studied.

The data presented within this study are at two different spatial scales: 2000 bathymetry is extremely fine resolution (more spatially-sensitive) and 1879 bathymetry is more coarse resolution (less spatially-sensitive). Using the best available data at the time the general trend in the data is that, overall, there have been changes to the estuaries' bottom, with more shallowing at Boca Ciega Bay and the Feather Sound special study sites. Boca Ciega Bay is composed mainly of sand and grass flats with a few tidal channels. In addition, Boca Ciega Bay has been dredged and filled extensively, resulting in a large decrease in estuarine habitat. More than 75% of this area is less than 2 m in depth (Brooks and Doyle 1998). Furthermore, Boca Ciega Bay is guarded by barrier islands, which have been documented to be sediment sinks (Penland et al. 1998).

CONCLUSIONS

Estuarine depth is the product of coastal sediment budgets that involve terrestrial, marine, and autochthonous sources and sinks. Most previous accounts of estuarine depth changes usually have been related to anthropogenic changes to sediment dynamics via dredging activities or alteration of natural hydrology such as in Safety Harbor and Old Tampa Bay (Peebles et al.

2009). The mean depth of Tampa Bay has increased by more than 5% in the past century (Goodwin 1984), which is consistent with our findings. Whether the depth of Tampa Bay will change as a result of climate change is an open but important question. Changes in the depth characteristics of Tampa Bay could affect water quality, the dispersion, abundance, or productivity of seagrasses, and possible benthic communities including intertidal oysters (see Tomasko, this volume).

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Table 1. Mean depth (meters below MLW) for different water bodies within Tampa Bay (Mean $\pm$ Standard Error)

	Hillsborough Bay	Terra Ceia Bay	Lower Tampa Bay	Old Tampa Bay	Boca Ciega Bay	Middle Tampa Bay
1879	2.391 $\pm$ 0.003	1.519 $\pm$ 0.002	4.327 $\pm$ 0.003	2.465 $\pm$ 0.002	1.717 $\pm$ 0.003	3.983 $\pm$ 0.002
2000	3.155 $\pm$ 0.007	1.654 $\pm$ 0.008	4.914 $\pm$ 0.006	2.734 $\pm$ 0.004	1.207 $\pm$ 0.005	4.49 $\pm$ 0.005
	-	-	-	-	-	-
Difference	0.759 $\pm$ 0.005	0.133 $\pm$ 0.007	0.586 $\pm$ 0.002	0.271 $\pm$ 0.002	0.513 $\pm$ 0.004	0.504 $\pm$ 0.002
Rank	1	5	2	4	6	3

Table 2. Mean Depth at MLW (meters) for Special Interest Areas in Tampa Bay (Mean $\pm$ S.E.)

	Feather Sound	MacDill AFB	Long Bar North	Long Bar Middle	Long Bar South
1879	1.33 $\pm$ 0.02	2.86 $\pm$ 0.05	2.90 $\pm$ 0.05	2.63 $\pm$ 0.06	4.10 $\pm$ 0.08
2000	1.23 $\pm$ 0.04	2.65 $\pm$ 0.08	3.28 $\pm$ 0.07	2.69 $\pm$ 0.09	4.50 $\pm$ 0.01
Difference	0.09 $\pm$ 0.03	0.21 $\pm$ 0.05	-0.33 $\pm$ 0.03	-0.08 $\pm$ 0.04	-0.38 $\pm$ 0.05

Table 3. Wilcoxon Statistics, for Special Interest Areas within the Tampa Bay Estuary.

Study Site	Z Score	Prob> Z
Feather Sound	-3.91	<0.05
MacDill AFB	0.49	0.62
Long Bar North	3.90	<0.05
Long Bar Middle	-3.59	<0.05
Long Bar South	2.91	<0.05

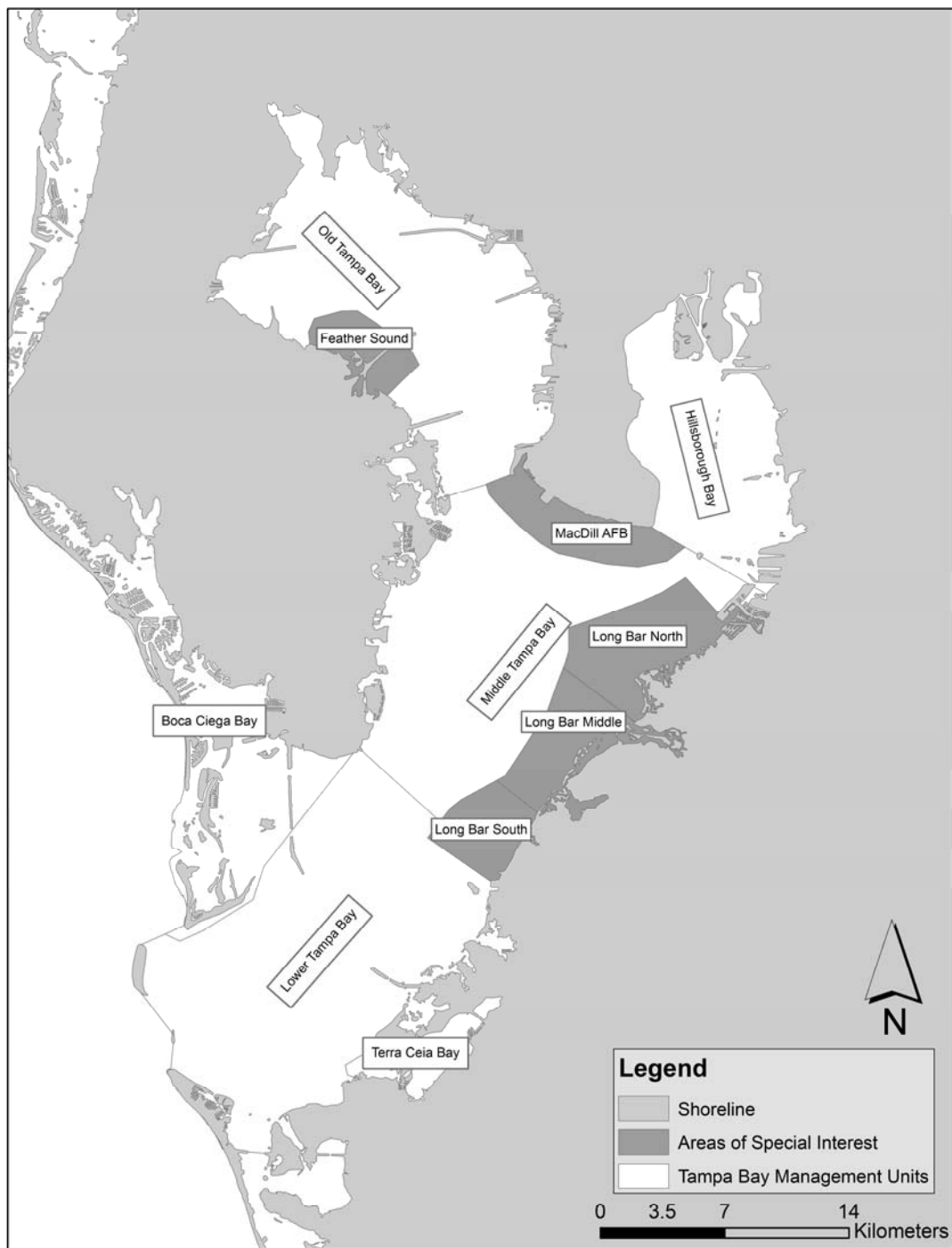


Figure 1. Tampa Bay and component management units and areas of special interest.

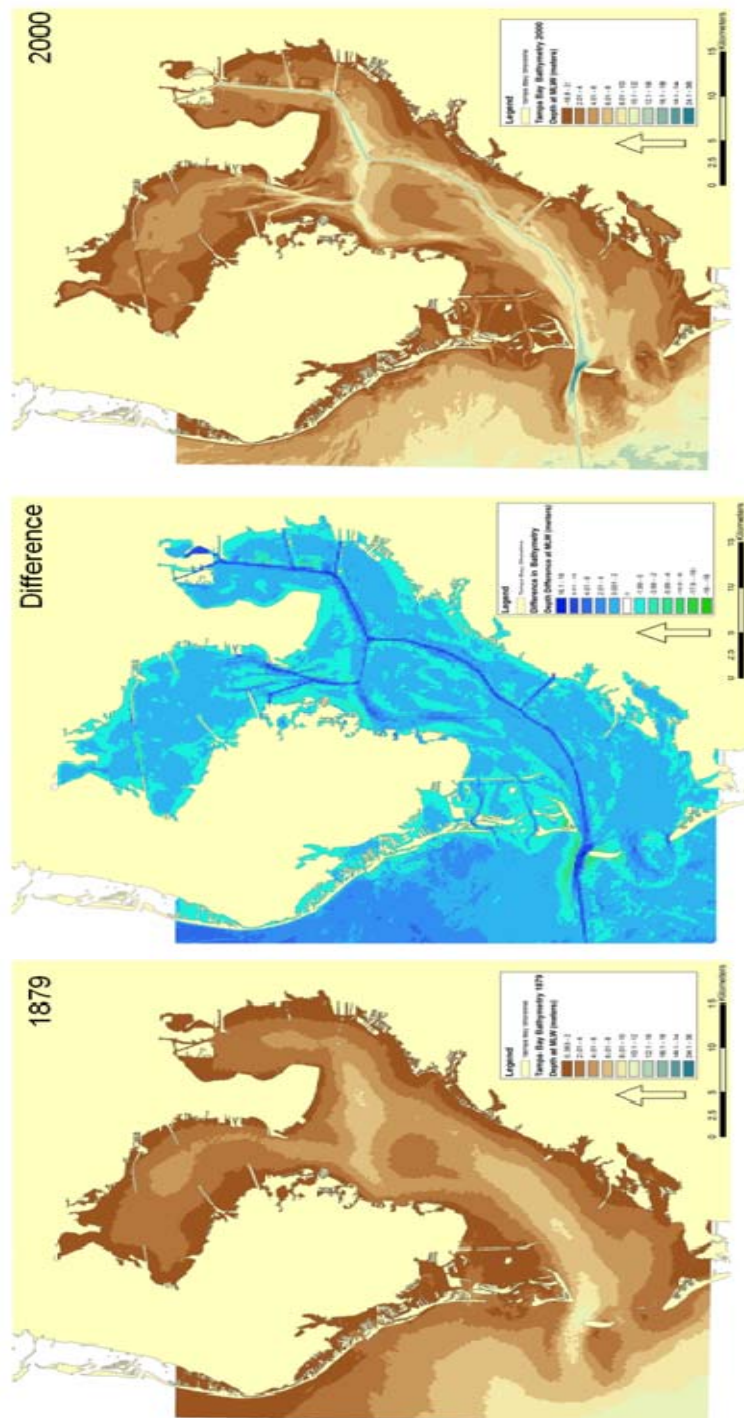


Figure 2. Comparison of Mean Low Water bathymetry of Tampa Bay between 1879 and 2000.

ABSTRACTS

Session 1: GEOLOGY & SEDIMENTS

PRESENTATIONS

A Siliciclastic-Filled Sedimentary Basin in a Mid-Carbonate Platform Setting, Tampa Bay, Florida. B.C. Suthard (Coastal Planning & Engineering, Inc.), A.C. Hine, S.D. Locker (University of South Florida), D.S. Duncan (Eckerd College), and R.A. Morton (United States Geological Survey). Tampa Bay, a 40 km-long shallow (4 m average depth) estuary situated along the west Florida coast, is underlain by a sedimentary basin of Neogene age, but of uncertain origin. This basin formed in the middle of one of the largest carbonate platforms on earth and, along with other Florida valleys and basins became a depocenter mostly filled by siliciclastic sediments during the Neogene siliciclastic invasion, as the Peninsular Florida carbonate platform became inundated with terrigenous quartz shed off the southern Appalachians.

Over 800 km of seismic reflection data collected within Tampa Bay was correlated to surrounding stratigraphic control in order to create a geologic framework of the underlying basin and fill. This framework defined a topographically-irregular (>40 m relief), subsurface basin formed sometime between Arcadia Formation limestone deposition (mid-Miocene, open-marine) and the Peace River Formation siliciclastic deposition (mid-to-late Miocene/early Pliocene fluvio-deltaic). The basin was not formed by fluvial-incision, but by deep-seated, structurally-controlled, karst processes, creating an irregular topographic low. Two main depocenters within the basin have been filled by a total of nine sedimentary units during five distinct phases of multiple Neogene and Quaternary sea-level cycles. The existence of this interior-platform basin during multiple sea-level cycles allowed it to control highstand and lowstand sedimentary deposition and erosion. In particular, the existence of this interior-platform basin at times when remobilized siliciclastics were already being transported along the Florida Platform created a unique situation whereby this basin recorded the local deposition of these siliciclastics.

The Tampa Bay basin is unique in its formation, location, and fill, and does not fit any existing interior-platform examples. This newly described example of a midplatform basin begins with the open-marine deposition of a carbonate platform. Structurally-focused, deep-seated karst processes dissolve voids within the center of this carbonate platform. Subsequent sea-level lowstands allowed for the destabilization and collapse of these voids, transferring the deformation vertically through the overlying strata, creating an irregular topographic low. After the formation of this basin, sea-level highstands allowed fluvio-deltaic and marine deposition, and, during lowstands, erosion of the existing sedimentary units, minor karst deformation, and restricted freshwater deposition. [bsuthard@coastalplanning.net]

Progress in the Application of Acoustic Bottom Classification Methods for Mapping and Characterizing Tampa Bay Benthic Environments. S.D. Locker, S.C. Dunn, A.C. Hine (University of South Florida), L.L. Robbins (United States Geological Survey), and E. Watkins (A&E Project Manager, LLC). Over the past several years a number of bottom-type mapping projects in Tampa Bay have been conducted for exploratory or site-specific mapping objectives. These efforts have included sidescan sonar and QTC single-beam acoustic classification. The Quester Tangent (QTC) acoustic classification method has been applied to both single-beam and sidescan data sets. The sidescan sonar results have proven useful for mapping bottom types such as submerged aquatic vegetation (SAV), hard bottom, and sediment type. The QTC single-beam acoustic classification has proven to be particularly useful for mapping sediment type distribution patterns.

A few large-area projects have been conducted that provide insight into the benthic variability, both nearshore and across the Bay. These projects, such as the Gulfstream gas pipeline benthic survey, and the Safety Harbor Sediment Project, will be reviewed in terms of what methods are most useful and what has been learned for application to future mapping efforts. The importance of multidisciplinary investigations is emphasized. [stan@marine.usf.edu]

ABSTRACTS
Session 2: CIRCULATION

PRESENTATIONS

The Circulation of Tampa Bay Driven by Buoyancy, Tides and Winds, and Its Connection with the Adjacent Gulf of Mexico. L. Zheng and R.H. Weisberg (University of South Florida). A three-dimensional, density-dependent, finite volume coastal ocean model (FVCOM) with high spatial resolution is used to investigate the circulation of the Tampa Bay estuarine complex, including Tampa Bay, the intracoastal waterway, Sarasota Bay, and the inner portion of the West Florida Shelf. Model performance over the three-month interval, September to November, 2001 was assessed against tide gauge and velocity profiler data before using the model to describe the circulation driven by rivers, tides and winds. Because of a mean wind velocity vector directed down the estuary axis during this time interval we ran a parallel experiment without winds to distinguish the estuarine circulation by gravitational convection from the mean wind effects. With or without winds, Tampa Bay exhibits a robust, two-layered estuarine circulation that concentrates on the deep channels. The mean outflow at the surface converges on the channels where the free surface elevation is locally a minimum. The mean inflow near the bottom also concentrates in the channels where the baroclinic pressure gradient is largest. Geometry thus guides the mean circulation; hence the salinity and all other water properties. At the bay mouth, mean outflows exist both in the deeper Egmont Channel and the shallower South Pass, whereas a mean inflow is limited to Egmont Channel. A residence time based on the Egmont Channel influx is estimated to be about 100 days. Consistent with previous studies we conclude that gravitational convection is a major contributor to the water property distributions of Tampa Bay. Additionally, by resolving all of the major water conveyances into and out of the bay, the model provides a fundamental framework for addressing problems of environmental concern for the greater Tampa Bay area, as well as for other estuarine systems along Florida's west coast. [LZheng@marine.usf.edu]

Storm Surge of Ivan-Like Hurricane Making Landfall Near Tampa Bay. R.H. Weisberg, L. Zheng, Y. Huang (University of South Florida). What may have occurred in the vicinity of Tampa Bay had Hurricane Ivan made landfall here instead of at the Alabama/Florida border? This question is explored using a three-dimensional, primitive equation, finite volume coastal ocean model. The results show that Tampa Bay storm surges are potentially large, especially for landfalls located to the north of the bay mouth. The worst case considered is for landfall at Tarpon Springs, such that the maximum wind is positioned at the bay mouth. By coupling a wave model with the circulation model we find an incremental increase in surge height via the wave radiation stress that adds to the wind stress. Significant wave height distributions also show substantial wave impacts throughout the bay region. Together, the surge and wave results suggest that the Tampa Bay region is as susceptible to damage and destruction as was coastal Mississippi during Hurricane Katrina. Along with such regional findings, we consider the dynamics of hurricane storm surge and in particular the differences that ensue from the use of three-dimensional, versus two dimensional (vertically-integrated) models. With hurricane storm surge deriving from the vertically integrated pressure gradient force tending to balance the differences between the surface and bottom stresses, we show that three-dimensional structure is intrinsically important, and that two-dimensional models, by overestimating bottom stress, tend to underestimate surge height. Such underestimation for our Tampa Bay Ivan-like simulations amounted to as much as 30%. This leads to the recommendations that agencies charged with hurricane storm surge simulation should consider transitioning to fully three-dimensional models and that additional studies are necessary to improve upon the surface and bottom stress parameterizations required by these models. [Weisberg@marine.usf.edu]

TRENDS IN ATMOSPHERIC NITROGEN OXIDES CONCENTRATIONS AND IMPLICATIONS FOR NITROGEN DEPOSITION TO TAMPA BAY

N.D. Poor

ABSTRACT

In recent years, atmospheric concentrations of nitrogen oxides (NO_x) in Hillsborough County have significantly declined. For example, annual average NO_x-NO₂ concentrations at an urban monitoring site decreased by >50% between 2002 and 2008, from 28 µg/m³ to 12 µg/m³. During this time frame, NO_x emissions from local power plants were reduced by ~18,000 metric tons/year. CALPUFF atmospheric transport modeling suggested that local power plant emissions reductions corresponded to a 7 µg/m³ decrease in NO_x-NO₂ concentration measured at an urban monitor and a ~130 metric ton/year decrease in nitrogen deposition to Tampa Bay. The remainder of the observed drop in NO_x concentration could be explained by a ~30% decrease in non-point source emissions with an estimated ~80 metric ton/year decrease in nitrogen deposition to Tampa Bay, based on CALPUFF modeling, however, other explanations are possible.

INTRODUCTION

This investigation has as its focus recent trends in atmospheric nitrogen oxides concentration and the implications for nitrogen deposition to Tampa Bay. A brief summary of results from CALPUFF modeling of nitrogen deposition to Tampa Bay is augmented with a discussion of the impact of local power plant reconfigurations and non-point source contributions to nitrogen deposition. Non-point source contributions include on-road, off-road (ships, airplanes, diesel equipment), agricultural sources, and miscellaneous combustion sources.

THEORY/CALCULATION

Nitrogen deposition estimates were based on CALPUFF modeling using 2002 meteorology and the 2002 National Emissions Inventory (NEI), which in 2008 was the most recently-completed criteria pollutant emission inventory of the United States Environmental Protection Agency (USEPA). CALPUFF modeling was done as a project for the Hillsborough County Pollution Recovery Fund, and details of the modeling effort are presented in the project's final report (Poor 2008), which can be obtained from the Environmental Protection Commission of Hillsborough County (EPCHC). Some aspects of the modeling are important for discussing this analysis. First, the modeling effort included only air pollution sources located within central peninsular Florida; second, biogenic emissions were not included. Thus, emissions modeled represent those source emissions that either are or could be regulated by state and local agencies. Finally, an assumption was made that 18% of the nitrogen deposited to Tampa Bay's watershed was transferred to Tampa Bay (Pollman and Poor 2003).

RESULTS

The CALPUFF-modeled predicted total nitrogen deposition to Tampa Bay from all source categories was 1,100 metric tons/year, of which Hillsborough County sources contributed 36%, Polk County 17%, Pinellas County 11%, and Manatee County 6%. The remaining central Florida countries contributed 30% (Figure 1).

Other estimates have been made of total nitrogen deposition to Tampa Bay. In 2006, Poor's (2006) best estimate was that atmospheric deposition delivered 2,550 metric tons-N/year to Tampa Bay. For wet deposition, her estimates were based on the product of rainfall concentrations of nitrogen and the rainfall depth; for dry deposition, the product of atmospheric

concentrations of nitrogen and deposition velocities, which were calculated from the National Oceanographic and Atmospheric Administration (NOAA) buoy model using mid-bay winds and temperatures (Mizak et al. 2007). Rainfall and atmospheric concentrations of nitrogen were measured at an urban monitoring site at the eastern end of Gandy Bridge in Tampa.

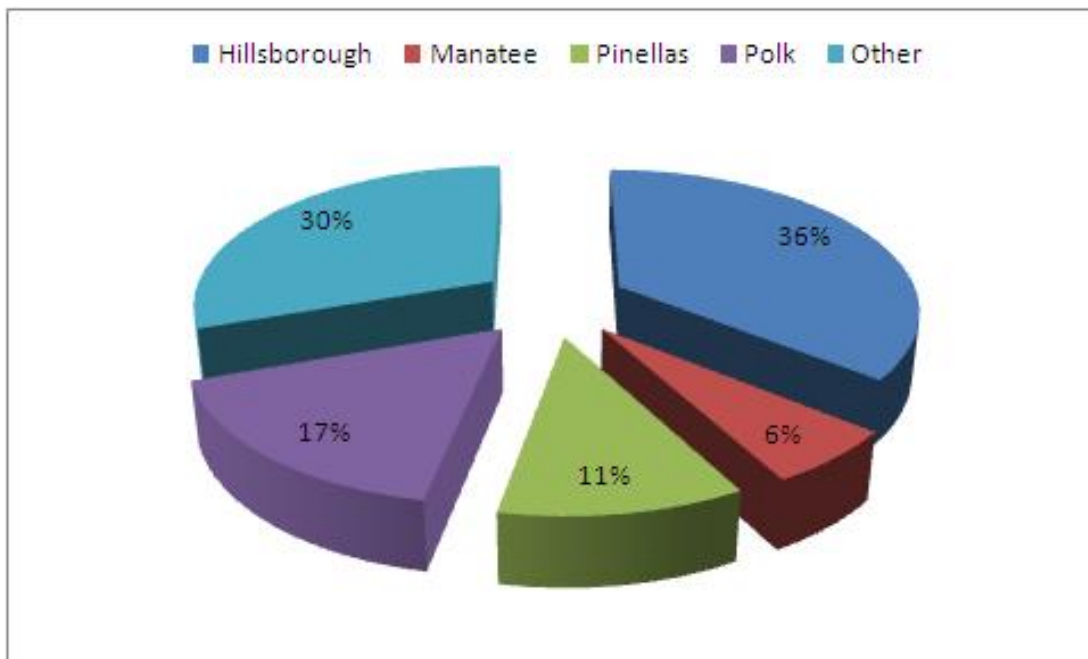


Figure 1. CALPUFF-modeled apportionment among neighboring counties of nitrogen deposition to Tampa Bay.

Dennis and Arnold (2007) reported on NOAA's nitrogen deposition modeling using the Community Multiscale Air Quality (CMAQ) model. Their modeled estimate was 1,600 metric tons-N/yr for a national modeling domain, also using the 2002 NEI but including biogenic emissions. Interestingly, CMAQ and CALPUFF models predicted the same direct deposition to Tampa Bay of 470 tons-N/year, compared with Poor's estimate of 850 tons-N/yr (Poor 2006). Part of the discrepancy between predictions based on measurements versus modeling may be that land-based concentrations of nitrogen may not be representative of mid-bay concentrations; another part of the discrepancy may be that wet deposition of nitrogen from long-range transport of air pollutants is under-estimated by air quality and atmospheric deposition models (Dennis and Arnold 2007; Strayer et al. 2007; Calderón et al. 2008).

Figure 2 shows that for emissions from anthropogenic air pollution sources located in central Florida and atmospherically depositing to Tampa Bay, and using an 18% nitrogen transfer rate from the watershed to the bay: direct deposition was 43% and indirect 57%; dry deposition was 67% and wet deposition 33%; nitrogen oxides contributed 63% and ammonia 37%; and non-point sources were 74% and point sources 26% of the total atmospheric nitrogen deposition.

The CALPUFF modeling effort looked at the effect on nitrogen deposition to Tampa Bay from the re-powering of the F.J. Gannon plant and the installation of NO_x controls at the Big Bend plant. Based on CALPUFF modeling, the removal of ~18,000 metric tons-N/yr of power plant emissions from the local inventory reduced the amount of nitrogen reaching Tampa Bay by ~130

metric tons/yr. This figure is consistent with an estimate by Dennis and Arnold (2007). Note that ratios of emission to deposition for these two power plants were both 140 tons/ton (Table 1).

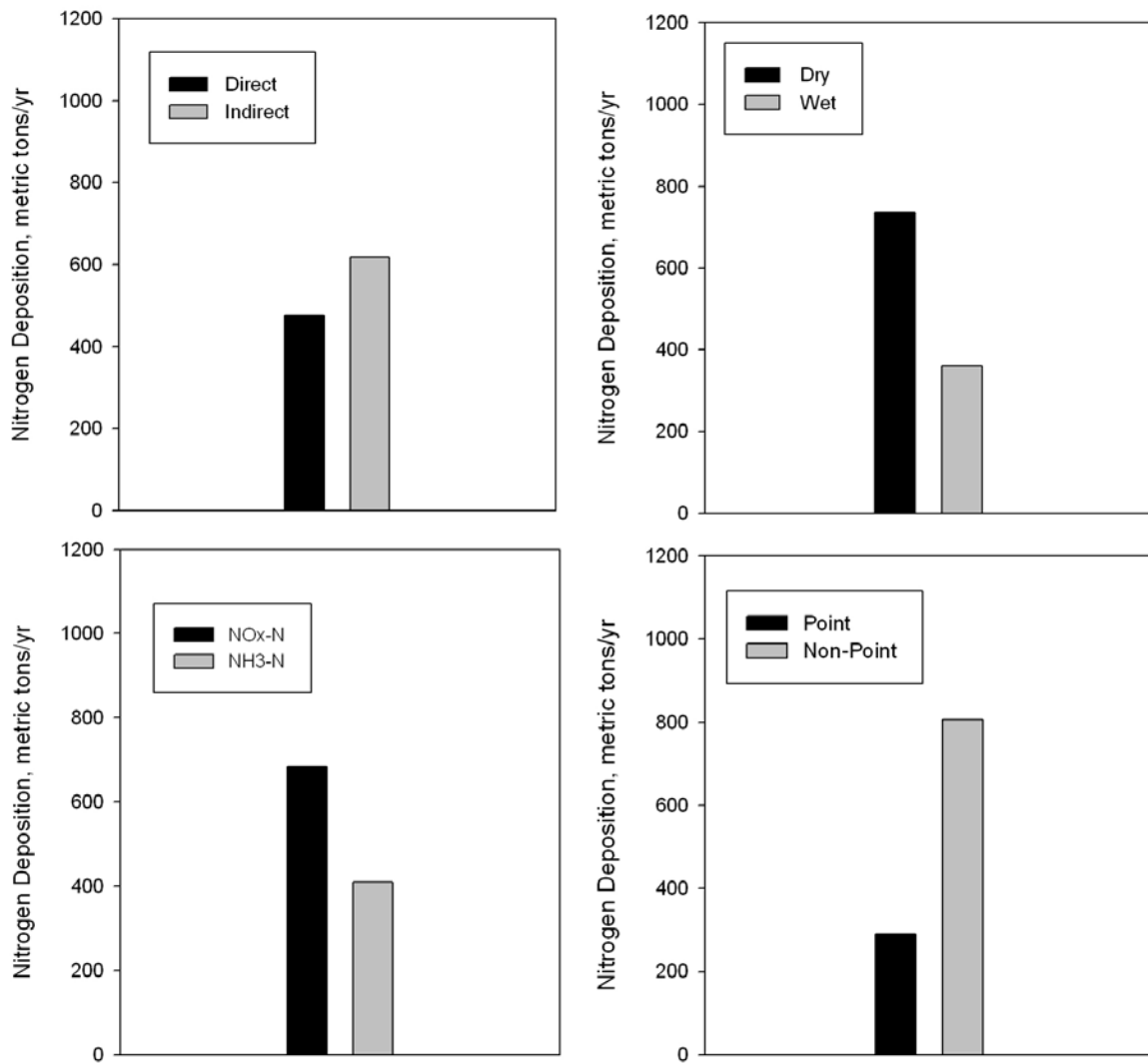


Figure 2. CALPUFF- modeled apportionment of direct versus indirect nitrogen deposition to Tampa Bay.

Modeled results of non-point source contributions from counties adjacent to and upwind of Tampa Bay indicated that of these counties, Hillsborough County had the largest contribution at 150 metric tons/yr and also the lowest emission to deposition ratio at 68 tons/ton (Table 2), not an unexpected result given the size of the county and its proximity to Tampa Bay. These results suggest that reductions in non-point source emissions of nitrogen in Hillsborough County would have a greater impact on nitrogen deposition to Tampa Bay than the same reductions made in neighboring counties. A 30% drop (2,100 metric tons N) in annual on-road emissions within Hillsborough County relates to a ~30 metric ton/yr decrease in total nitrogen deposition to Tampa Bay.

Table 1. CALPUFF-modeled impacts of Tampa Electric Company's (TEC's) power plant reconfigurations on atmospheric deposition of nitrogen to Tampa Bay. Units are metric tons.

Power Plant	Control	Change in Nitrogen Emissions	Change in Nitrogen Deposition	Ratio, Emission/Deposition
Big Bend* (NOx: 1998/2010)	Installed SCR for NOx control	9,240	66	140
F. J. Gannon** (NOx: 1998/2005)	Re-powered from coal to natural gas	8,810	63	140
Hooker's Point (NOx: 1998/2002)	Shut down in 2002	165	2.1	78

*Assumed in 89% reduction in NOx emissions

**Used 2005 emissions for Bayside Power Plant

Table 2. CALPUFF-modeled impacts of non-point source emissions on atmospheric deposition of nitrogen to Tampa Bay. Units are metric tons.

County	2002 Non-Point Nitrogen Emissions	2002 Nitrogen Deposition	Ratio, Emission/Deposition
Hillsborough	10,344	152	68
Manatee	3,307	28.7	115
Pinellas	8,220	67.4	122
Polk	4,962	56.4	88

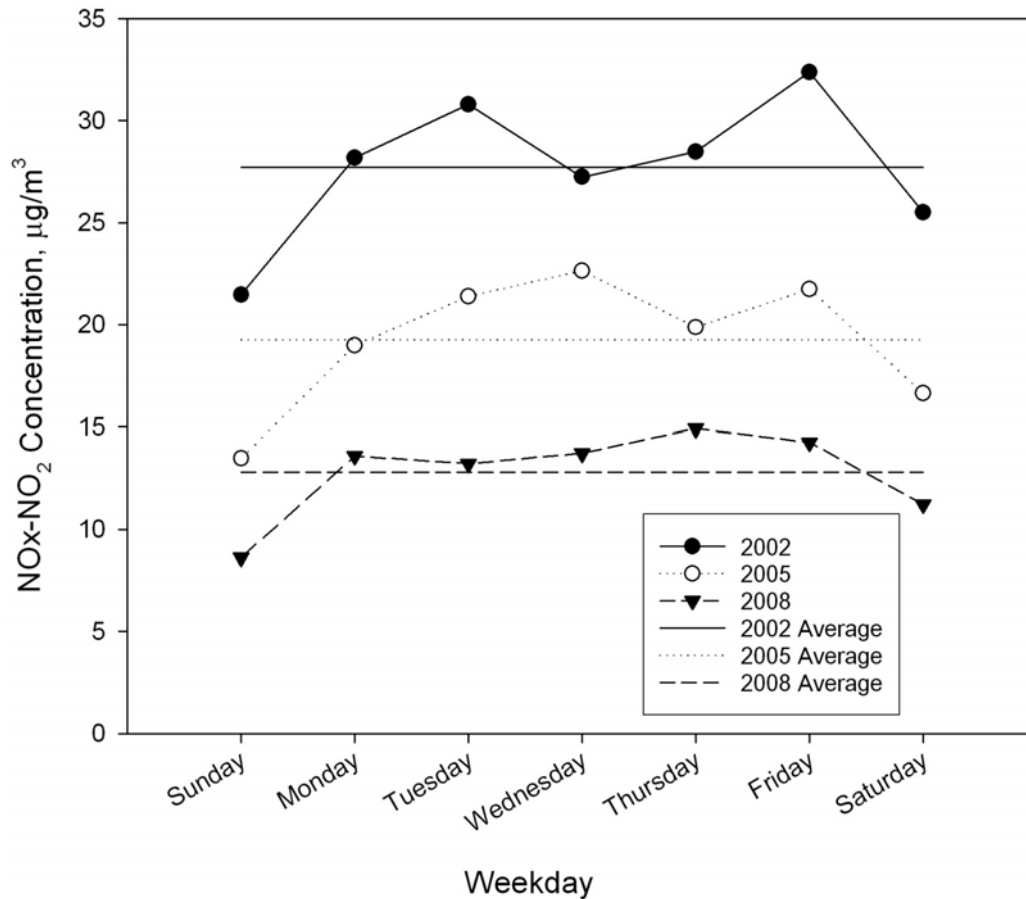


Figure 3. Temporal trends in weekday concentrations of nitrogen oxide (expressed as NO₂) concentrations at a monitoring site at the eastern end of Gandy Bridge (Poor 2008). Note: the 2008 trend line was based on data January through October, obtained in November 2008.

NO_x concentrations measured at the eastern end of Gandy Bridge were examined in the absence of data from a nitrogen deposition monitor (Figure 3). Weekday plots show that NO_x concentrations are the lowest on Saturdays and Sundays, consistent with a strong influence from on-road emissions. Previous research has shown a strong correlation between weekday average NO_x concentrations at EPCHC monitors and weekday average traffic counts on I-275 (Poor and Eckenrod 2007). The three plots from top to bottom represent trend lines for 2002, 2005, and 2008. In 2002, the average was 28 µg/m³; in 2005, the average was 19 µg/m³; and in 2008, the average was 12 µg/m³, a drop in the annual average NO_x concentration of 16 µg/m³. Not only did the average concentration drop but so did the range of weekday concentrations: from 11 µg/m³ in 2002 to 6 µg/m³ in 2008. The latter trend suggests not only a change in overall fleet emissions but a change in driving behavior. Note that in 2008, gasoline prices reached \$4/gallon.

DISCUSSION

How much of the downward trend (Figure 3) could be attributed to reduced power plant or other point source emissions and how much to changes in non-point source emissions? For a receptor located at the eastern end of Gandy Bridge, CALPUFF predicted for 2002 an annual average concentration of $22 \mu\text{g}/\text{m}^3$ as NOx-NO_2 (Table 3), which was lower than the measured average concentration of $28 \mu\text{g}/\text{m}^3$. This may be a consequence of the volume source approach taken to model non-point emissions, which smoothes concentration gradients present near emission source regions such as roadways and intersections.

As modeled, the TEC Big Bend and Gannon power plant reconfigurations caused a $\sim 7 \mu\text{g}/\text{m}^3$ decrease in the average annual NOx concentrations at the Gandy Bridge site (Table 3). Also as modeled, a hypothetical 30% drop (2,100 metric tons N) in annual on-road emissions within Hillsborough County would cause a $\sim 2 \mu\text{g}/\text{m}^3$ decrease. [Note: off-road emissions from construction equipment may have also dropped significantly in 2008 due to recessionary economics.] Similar percentage reductions in Pinellas, Polk, and Manatee Counties of on-road emissions would be needed to account for the observed decrease in weekday average NOx-NO_2 concentrations. Thus, one explanation is that longer-term regulatory-imposed emissions reductions on motor vehicles coupled with shorter-term high fuel prices and recessionary economics and in combination with local power plant reconfigurations contributed to a dramatic decline in urban NOx-NO_2 concentrations. If the counties shown in Table 2 reduced their non-point source nitrogen emissions by 30%, the corresponding reduction in nitrogen deposition would be ~ 80 metric tons/yr.

Table 3. CALPUFF-modeled apportionment of NOx-NO_2 at an EPCHC monitoring site.

Monitoring Site	All Non-Point Sources $\mu\text{g}/\text{m}^3$	All Power Plant Sources $\mu\text{g}/\text{m}^3$	All Remaining Point Sources $\mu\text{g}/\text{m}^3$
Gandy Bridge	14	7	1

Another explanation is that the 2002 NEI did not include or underestimated one or more major sources of NOx emission, including non-point source emissions. Power plant emissions were based on hourly NOx emissions from in-stack measurements; stationary source emissions on annually-averaged company-reported releases; and non-point source emissions from annually-averaged population-based estimates made with USEPA-approved models, as examples.

A third explanation is that choices of CALPUFF model algorithms did not adequately represent the nitrogen oxide emissions or atmospheric transport, transformation, and deposition, for example, and nearly all of the NOx-NO_2 reductions were due to TEC power plant reconfigurations. For a broader discussion of CALPUFF-modeled versus measured air pollution concentrations and nitrogen deposition rates, see Poor (2008).

CONCLUSIONS

Results of CALPUFF modeling suggest that declines in atmospheric nitrogen oxides concentrations over the past six years as seen at an urban monitor may have been a consequence of reductions not only in power plant emissions but also in mobile source emissions. CALPUFF-modeled reductions in power plant emissions corresponded to a decrease in nitrogen deposition to Tampa Bay of ~130 metric tons/yr. Reductions of mobile source emissions during this time frame may have been as much as 30% of the non-point source emissions inventory, which corresponds to a decrease in nitrogen deposition to Tampa Bay of ~80 metric tons/yr.

ACKNOWLEDGEMENTS

CALPUFF modeling of atmospheric nitrogen deposition to Tampa Bay was funded by the Hillsborough County Pollution Recovery Fund, which is managed by the Environmental Protection Commission of Hillsborough County.

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TEMPORAL TRENDS IN TROPHIC STATE PARAMETERS FOR LAKES CLUSTERED IN NORTHWESTERN HILLSBOROUGH COUNTY

M.J. Moreno & N.D. Poor

ABSTRACT

Temporal trends of 10 lakes located within the Tampa Bay watershed in northwestern Hillsborough County were investigated. These lakes were classified as oligotrophic or mesotrophic. Water quality data for 1990 to 2007 were obtained from the Hillsborough County Water Atlas. Water quality data were divided into two periods: Period 1, 1990 to 1998; and Period 2, 1999 to 2007. A randomized complete block design was used for this analysis. Period was the treatment. To account for lake (drainage basin) effect, the data were blocked by lake. The responses were lake water concentrations of total phosphorus (TP) and chlorophyll- α , and the ratio of total nitrogen to total phosphorus (TN:TP). The experimental unit was lake water in the Tampa Bay watershed. A two-way analysis of variance (ANOVA) was conducted with SYSTAT[®] to test the research hypotheses under an assumption that samples were collected at random (randomly distributed in time). A two-tailed t-test was applied to detect differences between periods within each lake at the 95% confidence level. With few exceptions, concentrations of TP and chlorophyll- α significantly increased and the TN:TP ratio significantly decreased between periods and within lakes over the 17-year period. Regression analysis showed that apparent positive correlations between rainfall and annual average TP and chlorophyll- α concentration, and a negative correlation between rainfall and annual average TN:TP ratio, when these trophic state parameters were averaged across all lakes, were not significant at the 95% confidence level.

INTRODUCTION

The 570,000 ha Tampa Bay watershed lies within the counties of Hillsborough, Pinellas, and Manatee and extends into parts of Sarasota, Pasco, and Polk Counties. Among the six counties that occupy part of the Tampa Bay watershed, Hillsborough County has the largest proportion of its area included within the watershed, the largest population, and the highest recent increase in population. Hillsborough County had a 19.8% increase in population during the time interval between 1990 and 2000 while Pinellas County only had 8.2% (US Bureau of the Census 2008a) and the entire United States had 13.2% (Perry et al. 2001). The population estimate for Hillsborough County for 2008 was 1,180,784 (US Bureau of the Census 2008b). There is strong concern over the impact of this fast growth on the cultural eutrophication of natural lakes located in the Tampa Bay watershed. Furthermore, such concern relates to not only the bodies of water within the watershed, but more broadly, to the possible effects on the receiving bay.

Lake eutrophication is a natural process based upon a gradual progression in the concentration of nutrients and productivity, however, it can be hastened by human activities (Greeson 1969). Population growth and watershed development have often been associated with increased nutrient concentration in naturally-occurring bodies of water (Rast and Thorton 1996; Smith et al. 2003). Nutrients found in the urban runoff are increasingly important contaminants and eutrophication factors in estuaries and other aquatic ecosystems (Florida Department of Environmental Protection (FDEP) 2006). Terrell et al. (2000), however, did not find a trend of increasing water concentration of total phosphorus (TP), total nitrogen (TN), and chlorophyll- α across 127 Florida lakes during a time period of growing population between 1967 and 1997. FDEP (2008) has reported that out of 349 lakes for which there were sufficient data for trend analysis between 1997 and 2007, 92 showed an improving trend in trophic water quality, 141 were stable, and 116 were degrading.

Surface water enrichment stimulates abundance of phytoplanktonic species, some of which can be toxic. Abundant productivities of any toxic phytoplankton are better known as harmful algal blooms (HABs). Freshwater HABs are a growing concern for public health in Florida because of their potential for adverse health effects when present in surface drinking water resources and recreational sites (Williams 2004, as quoted by the report 305 (b) from the FDEP 2008).

Upon this background, this study provides an analysis of the nutrient and chlorophyll- α concentration trend in a group of lakes located in the northwestern corner of Hillsborough County within the Tampa Bay watershed. This study examined the change in lake water concentrations of eutrophication-related parameters between 1990 and 2007 in these lakes. These findings will provide more evidence to help corroborate or contradict the theory of cultural eutrophication associated with watershed development (Smith et al. 2003). On a broader scale, the knowledge of a trend, if one exists, would assist surface water managers and community stakeholders in their efforts to create sustainable development in the Tampa Bay watershed.

METHODS

Data Sources and Sampling Methods—To examine the temporal variability of trophic status on 10 lakes located in the northwestern corner of Hillsborough County (within the Tampa Bay watershed), existing record data for lake water concentrations of TN, TP, and chlorophyll- α were compiled. The data on trophic status analyzed in this study were extracted from the Water Atlas (2008), web site of the Florida Center for Community Design and Research at the University of South Florida. The data were collected and reported by citizen volunteers sponsored by the LAKEWATCH water quality monitoring program. Surface water samples were collected at least once per month over the 18-year study period, generally by boat during daylight at one to six (depending upon water body size) mid-lake, 0.5-m depth locations. The average value from the different locations was the value reported for each parameter at any given sampling date. The samples were analyzed in the Department of Fisheries and Aquatic Sciences' water quality laboratory at the University of Florida, following the Comprehensive Quality Assurance Plan # 910157, which is approved by the Florida Department of Environmental Protection. The methods used to collect the data are described in Brown et al. (1998). Acid-cleaned triple-rinsed Nalgene bottles of 250 ml were provided by the Department of Fisheries and Aquatic Sciences for collecting monthly water samples at each sampling site for TP and TN analysis. Upon sample collection, bottles were stored frozen. Additional surface water to estimate concentration of planktonic algae was collected at each sampling location by using 4-L, tap-water rinsed plastic jugs. A measured volume of water taken from the plastic jugs was filtered through a Gelman Type A-E glass fiber filter. Water samples and the glass fiber filters were maintained in a freezer during one to four months until delivered to the laboratory of the Department of Fisheries and Aquatic Sciences at the University of Florida.

TP was determined by oxidizing phosphorus with potassium persulfate (Menzel and Corwin 1965) and measuring the liberated phosphorus with the colorimetric technique of Murphy and Riley (1962) as cited in Menzel and Corwin (1965). TN was determined by a persulfate oxidation technique (D'Elia et al. 1977) followed with nitrate-nitrogen determination by ultraviolet derivative spectroscopy of second order (Bachmann and Canfield 1996; Simal et al. 1985; Wollin 1987). Determination of chlorophyll- α was done by extracting the pigment with ethanol (Sartory and Grobbelar 1984) and then measuring it with spectrophotometry following the Standard Method (SM) 10200 H method (Clescerl et al. 1998). Additionally, data on rainfall generated by the Southwest Florida Water Management District (SWFWMD) at the Tampa International Airport and also made available by the Water Atlas (2008), was also used.

Selection Criteria and Location—Out of all lakes for which information on water concentrations of TP, TN, and chlorophyll- α was provided by the Water Atlas (2008), only 10 (for TP and TN), and 8 (for chlorophyll- α) met the three inclusion criteria: containment within the Tampa Bay watershed, data availability for at least 75% of the 18-year study period (1990 to 2007), and a trophic state classification of either oligotrophic or mesotrophic based on the overall mean of lake water TP concentration for each lake during the study period. The classification criteria followed the Trophic State Classification System of Forsberg and Ryding (1980). According to this system and based on TP concentration, a surface water is oligotrophic with a concentration lower than $15 \mu\text{g l}^{-1}$, a concentration between 15 and $25 \mu\text{g l}^{-1}$ would correspond to mesotrophic, and between 25 and $100 \mu\text{g l}^{-1}$ would be eutrophic water. Concentrations of TP greater than $100 \mu\text{g l}^{-1}$ would be indicative of hypereutrophic waters.

The locations of the lakes used for this analysis are represented graphically in Figure 1. For general information about the lakes, refer to Table 1. The total data set of available values for TP, chlorophyll- α , and ratio of TN to TP (TN:TP) from each lake were used over the 18-year time period. Lake water TN concentration is analyzed here as relative to that of lake water TP because the 10 lakes analyzed are phosphorus-limited according to the nutrient limitation criteria suggested by Brezonik (1984). This criteria proposes that lakes with TN:TP ratios greater than 30 are phosphorus-limited while those with ratios less than 10 are nitrogen-limited. Nutrient limitation in lakes is balanced (both nutrients are limiting) if TN:TP is between 10 and 30. Additionally, considering the theory of Schindler (1977), there is further reason for using the TN:TP rather than TN alone. According to this theory, nitrogen-fixing cyanobacteria, a phylum of phytoplankton that can be toxic, would be favored over other species of phytoplankton by increasing conditions of relative limited nitrogen and abundant phosphorus in the water supply to the lake. The 10 lakes analyzed for trends in concentrations of TP and TN:TP ratios were Armistead, Calm, Carroll, Crenshaw, Deer, Hiawatha, Juanita, Keystone, Magdalene, and Sunset. Due to lack of data available for at least 75% of the 18-year study period (1990 to 2007), Lake Carroll and Deer Lake were excluded from the chlorophyll- α analysis.

Statistical Analysis—Measurements taken over time in this study were not independent, residuals were not normally distributed, data were not collected at a constant set of time points, and many values were missing. These characteristics make standard regression analysis inapplicable for a longitudinal study (Lin and Ying 2003; Ware 1985; Zeguer et al. 1988). For each parameter in each trophic group of lakes, however, least squares linear regressions were plotted as a preliminary visual assessment of the overall 18-year trends but not as a conclusive method of analysis. Points that were 5 times greater or smaller than the mean and that individually influenced the mean value of the eutrophication parameter were removed. The total numbers of points removed by this method were 3 out of 1,232 for TP and none for chlorophyll- α .

A randomized complete block design (Ott 1993) was used for this analysis. Period was the treatment. To account for lake effect, the data was blocked by lake. The response was lake water concentration of TP and chlorophyll- α , and TN:TP ratio and the experimental unit was lake water in Tampa Bay watershed. It was an assumption that samples were collected at random (randomly distributed in time). The null hypotheses for each of the sets are given below:

1. The means of the periods are equal.
2. The means of the lakes are equal.
3. There is no interaction between periods and lakes.

A two-way analysis of variance (ANOVA) was conducted with SYSTAT[®] to test the mentioned hypotheses. The first factor was period. The data set for each trophic group was divided in two

periods: Period 1 included all data sets from 1990 until 1998 and Period 2 from 1999 until 2007. The second factor was lake (lake drainage basin). ANOVA is a tool typically applied to controlled experiments rather than to "natural" experiments; however, ANOVA has been used in natural studies to test for changes in TP (Smith and McCormick 2001) and mercury (Babiarz et al. 1998) concentrations over time or space. T-test considering unequal variances was applied to detect differences between periods within each lake. A significant level of $\alpha = 0.05$ was used to perform all the statistical tests. Additionally, annually-averaged (Zeguer et al. 1988) TP and chlorophyll- α concentrations, and TN:TP ratio across all lakes were plotted by year for the time period between 1990 and 2006 to visualize any possible cycling in the data. This plot was further compared with the plot of annual cumulative values of rainfall generated by SWFWMD for rainfall measured at the Tampa International Airport during the same time period.

RESULTS AND DISCUSSION

TP—Results from the two-way ANOVA showed a significant difference between means of lakes for all the trophic state variables considered ($p < 0.0001$). Interactions were found between periods and lakes for all the trophic state variables considered ($p < 0.003$). Over the 18-year study period, the overall annual average of TP concentration increased from $11.71 \mu\text{g l}^{-1}$ in 1990 to $15.00 \mu\text{g l}^{-1}$ in 2007. The best-fit line for a plot of data points suggested a weak correlation ($r = 0.3$) and a positive slope (Figure 2). Overall average annual TP concentrations for the analyzed lakes ranged from a minimum of $10.47 \mu\text{g l}^{-1}$ in 1994 to a maximum of $19.09 \mu\text{g l}^{-1}$ in 2004. Individual lake values ranged from a minimum of $2.00 \mu\text{g l}^{-1}$ in 1994 to a maximum of $59.00 \mu\text{g l}^{-1}$ in 2003. Many values in the upper annual range exceeded the limit of $25 \mu\text{g l}^{-1}$ of TP used as a limit beyond which the lake would be considered eutrophic. However, the means for each one of these lakes remained within the $25 \mu\text{g l}^{-1}$ limit according to the criteria.

A significant difference was found with two-way ANOVA between the means of the first and second periods ($p < 0.0001$) for TP concentration. The t-tests performed on this trophic state variable for each lake between Periods 1 and 2 showed a statistically significant difference ($p \leq 0.05$) in all the lakes of this group except for two. The fact that not all lakes increased significantly between period 1 and 2 confirmed the significant interaction between periods and lakes found with two-way ANOVA. Standard deviation (SD) and coefficient of variation (CV) in TP concentrations ranged respectively from 3.77 and 27% in 1999 to 10.59 and 60% in 2003.

Chlorophyll- α —For chlorophyll- α , the study period in the eight lakes analyzed started with an overall annual average concentration of $6.52 \mu\text{g l}^{-1}$ in 1990 and ended with an average of $5.27 \mu\text{g l}^{-1}$ in 2007. The temporal distribution of data points plotted suggested an increasing trend ($r = 0.23$, Figure 3). The trend in chlorophyll- α concentration appears to follow a similar increase as TP because these lakes are phosphorus-limited. It has been extensively documented a strong association between lake water nutrients concentration and phytoplankton abundance as measured by chlorophyll- α concentration, these results were consistent with the two-way ANOVA ($p < 0.001$). The t-tests of chlorophyll- α concentration for each lake revealed that concentrations in 3 of the 8 lakes showed no significant change between Period 1 and Period 2 ($p > 0.05$) and confirmed an interaction between periods and lakes. Annual averages values ranged from 4.38 in 1996 to $13.80 \mu\text{g l}^{-1}$ in 2005, and SD from 3.17 in 2007 to 11.05 in 2003 and CV from 60% to 101% for the same years. Lake values ranged from 1.00 to $48.00 \mu\text{g l}^{-1}$.

The overall group of lakes presented annual mean TN:TP ratios slightly decreasing from 55.35 in 1990 to 44.14 in 2007 and ranged from 35.82 in 2004 to 55.35 in 1990. Individual values ranged from 12.41 in 2004 to 132.00 in 1993, the annual SD ranged from 10.04 in 2005 to 18.14 in 1993 and CV from 26% to 33% for the same years. A plot of values of TN:TP ratios shows a decreasing line with a weak correlation ($r = 0.26$, Figure 4) for this group of lakes. Two-way

ANOVA confirmed this trend by finding a significant difference in TN:TP ratios between the first and second periods, being lower in the second ($p < 0.001$). Even though all the 10 lakes showed a decrease in TN:TP from the first to the second period, when tested individually by t-test, this change was not statistically significant ($p > 0.05$) for four lakes. The effect of period on TN:TP was not consistent between lakes, indicating interaction between period and lake.

TN:TP—Since TN:TP is a ratio of TN to TP, its plot opposed to that of TP is explained by the change in lake water TN concentration. Although TN is also increasing for the 10 lakes (results not shown), the decreasing trend in the TN:TP ratio is due to the faster growth on the denominator (TP) as compared to the numerator (TN) as time progresses. As the plot of ratios shows, the annual means for the overall set of lakes are included within the phosphorus limitation criteria of Brezonik (1984), that is, all ratios ≥ 30 . However, if the downward trend continues, it would result in a gradual tendency from being phosphorus-limited toward being phosphorus-and nitrogen-limited.

Annual averages of trophic state variables and annual cumulative rainfall—Plotted annual averages of TP and chlorophyll- α concentrations and TN:TP ratio for the interval 1990-2006 (Figures 5, 6, and 7 respectively) suggest a possible cycling behavior for all the variables analyzed in our group of lakes. The curves for lake water concentration of TP and chlorophyll- α show three peaks going in ascending order during the time interval 1990-2006 (Figures 5 and 6, respectively). The highest averaged values for TP and chlorophyll- α occurred in 2004 and 2005, respectively, and the next two higher concentrations for both variables occurred in order in 1998 and 1991.

Water quality has been related with rainfall on coastal waters of Florida (Lipp et al. 2001a). This is explained by an excess in nutrients-loading runoff brought as a consequence of an excess in rainfall (Carlston et al. 2003). This may be the reason for the apparent direct concurrence between the plot of annual averages of TP and that of annual cumulative rainfall at the Tampa International Airport (Figures 5 and 8, respectively). Similarly, since an increase in lake water TP concentration is associated with an increase in lake water concentration of chlorophyll- α (Bachmann et al. 2002; Canfield 1983), it can consequently explain also some similarity between the plots of annual averages of chlorophyll- α and annual cumulative rainfall (Figures 6 and 8, respectively). That similarity attempts to support the relationship between these two variables reported by Carlston et al. (2003) along the west coast of Florida. A Pearson correlation analysis, however, showed that the associations between rainfall and the three trophic state parameters considered were not significant (rainfall with TP, $r = 0.29$ and $p = 0.26$; with chlorophyll- α , $r = 0.22$ and $p = 0.41$; and with TN/TP ratio, $r = -0.35$ and $p = 0.17$).

High increases in rainfall have also been associated with El Niño episodes in Florida (Schmidt et al. 2001), which may explain why the peaks in the plot of cumulative rainfall tend to coincide with years typical of El Niño (Figure 8). This supports the presumption of an association between water quality and the effects of El Niño-Southern Oscillation Events suggested by Lipp et al. (2001b).

Rising lake water TP and chlorophyll- α concentration in these 10 oligotrophic and mesotrophic lakes located in northeastern Hillsborough County appear to trend along with County population growth and may be evidence of an urban signature in lake water quality. An increase in urban land use means more impervious cover and thus increased rainfall run-off (Xiana and Craneb 2005). Additionally, since the geology in northeastern Hillsborough County is characterized by a karst formation (van Beynen et al. 2007) and households on this area rely on onsite sewage septic

systems, leaks and infiltration of nutrients along with the groundwater flow may be facilitated by this type of geology. Another explanation could be a possible increase in rainfall causing an increase in lake water TP and chlorophyll- α concentration.

CONCLUSIONS

Trophic state parameters of 10 lakes located in the Tampa Bay watershed showed significant trends between 1990 and 2007. Results obtained from this group of oligotrophic or mesotrophic lakes during the study period indicated for the group of 10 lakes an increase in the lake water concentrations of TP and a decrease in the TN:TP ratios, while eight of the 10 lakes presented an increase in lake water concentrations of chlorophyll- α . Results on chlorophyll- α were not known for the two remaining lakes of the group. The plot of annual averages for the three variables studied suggests a possible cycling behavior that seems to resemble the curve of annual rainfall and El Niño-Southern Oscillation events.

There are three main findings regarding these trends.

1. The concentration of phosphorus (as TP, for 10 lakes) and phytoplankton (as measured by chlorophyll- α for eight lakes) increased over time for lakes classified as oligotrophic or mesotrophic.
2. The ratio of nitrogen to phosphorus (TN:TP) declined for 10 lakes classified as oligotrophic or mesotrophic. For the lakes studied a trend line suggests that in ~20 years primary productivity may be nitrogen-limited.
3. Oscillation on lake water concentration of TP and chlorophyll- α , and on lake water TN:TP ratio of 10 lakes classified as oligotrophic or mesotrophic appears to follow the oscillation of annual rainfall. However, statistical analysis showed that correlation between these trophic state parameters and rainfall was not significant.

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Table 1. General Information about Lakes Examined for Eutrophication Trends.

Lakes	Area (ha)	Mean Depth (m)	Volume (m³)	Latitude	Longitude	Lake region
Keystone	175	3.3	5714175	28.1331	-82.5900	Keystone Lakes
Magdalene	83	2.4	2385712	28.1319	-82.4819	Land-o-Lakes
Carroll	82	2.4	2044183	28.0511	-82.4875	Land-o-Lakes
Hiawatha	55	3.3	1873595	28.1694	-82.5742	Keystone Lakes
Calm	47	2.7	1477849	28.1422	-82.5817	Keystone Lakes
Armistead	14	2.7	347739	28.1011	-82.5597	Tampa Plain
Deer	14	3.6	501064	28.1678	-82.4625	Land-o-Lakes
Sunset	13	2.4	352140	28.135	-82.6255	Keystone Lakes
Crenshaw	12	1.5	42304	28.12583	-82.4958	Land-o-Lakes
Juanita	10	2.7	246887	28.1175	-82.5889	Keystone Lakes

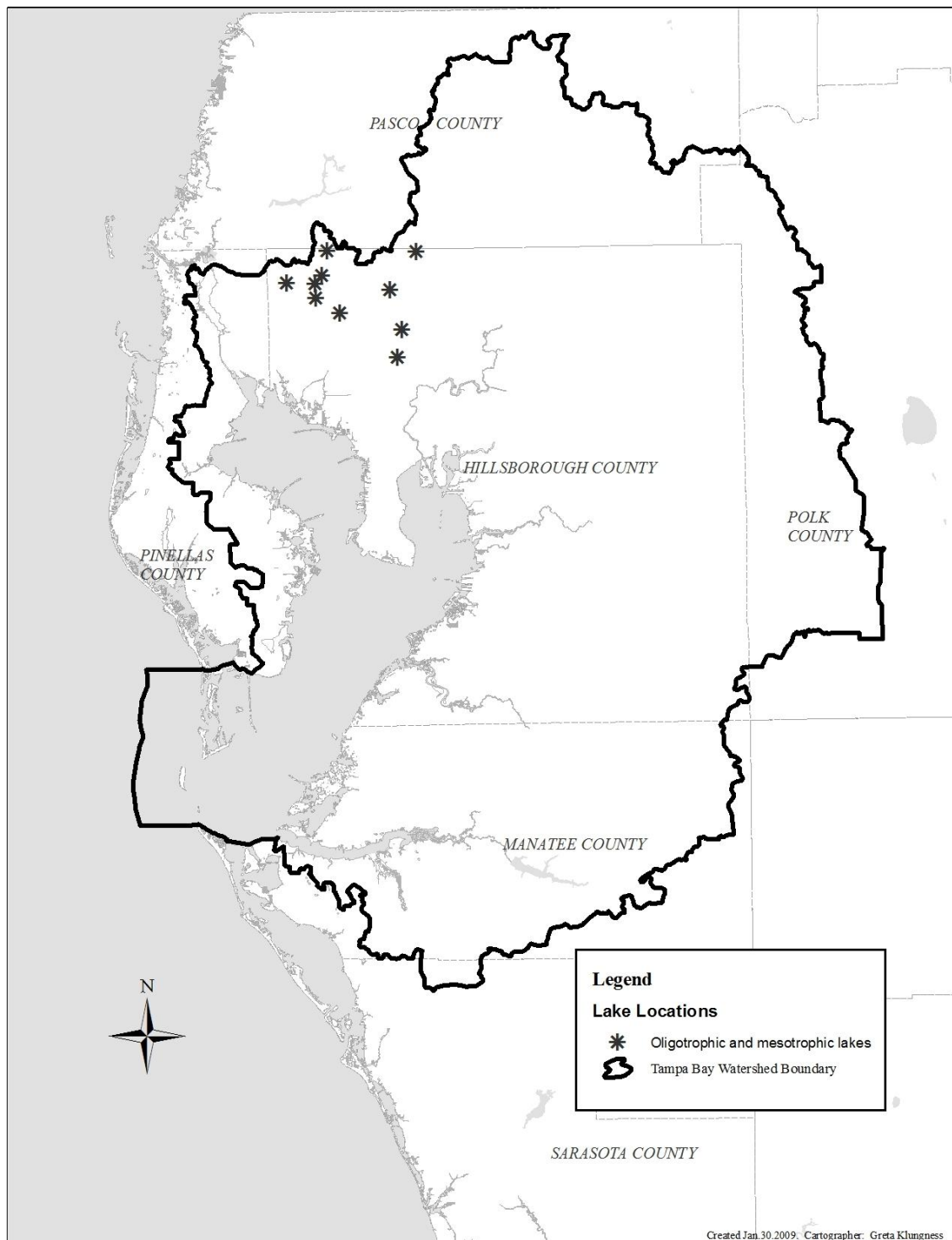


Figure 1. Lakes analyzed for trends in trophic state variables during the time period 1990 to 2007.

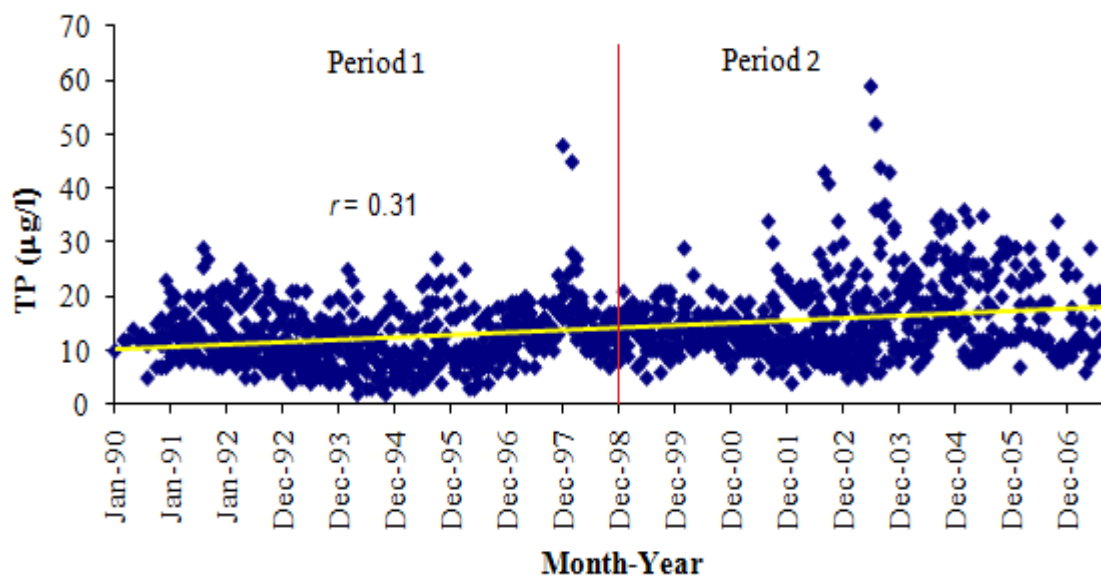


Figure 2. Plot of TP concentrations for 10 oligotrophic and mesotrophic lakes for the time period 1990-2007.

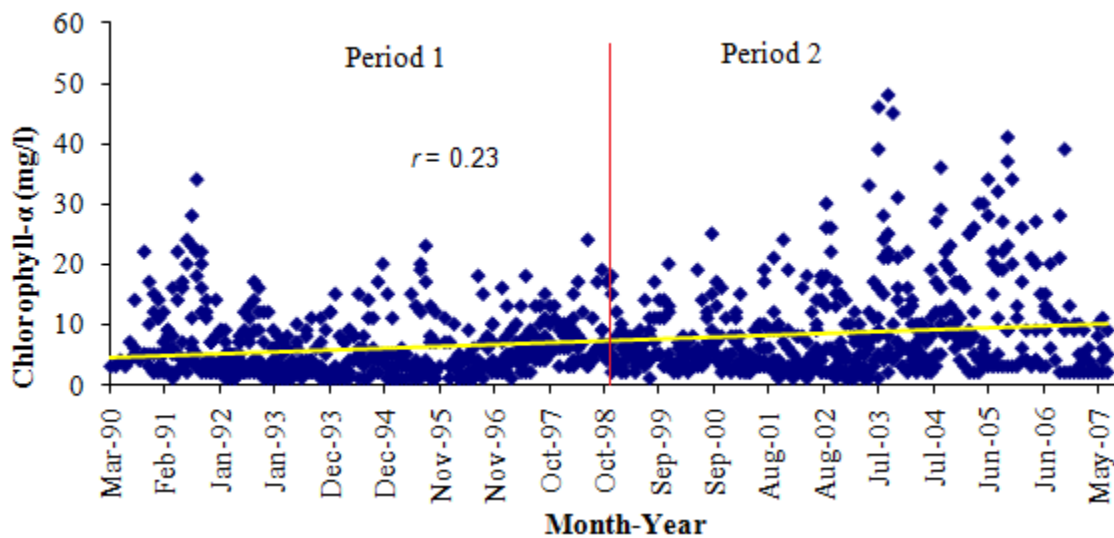


Figure 3. Plot of chlorophyll- α concentrations for 8 oligotrophic and mesotrophic lakes for the time period 1990-2007.

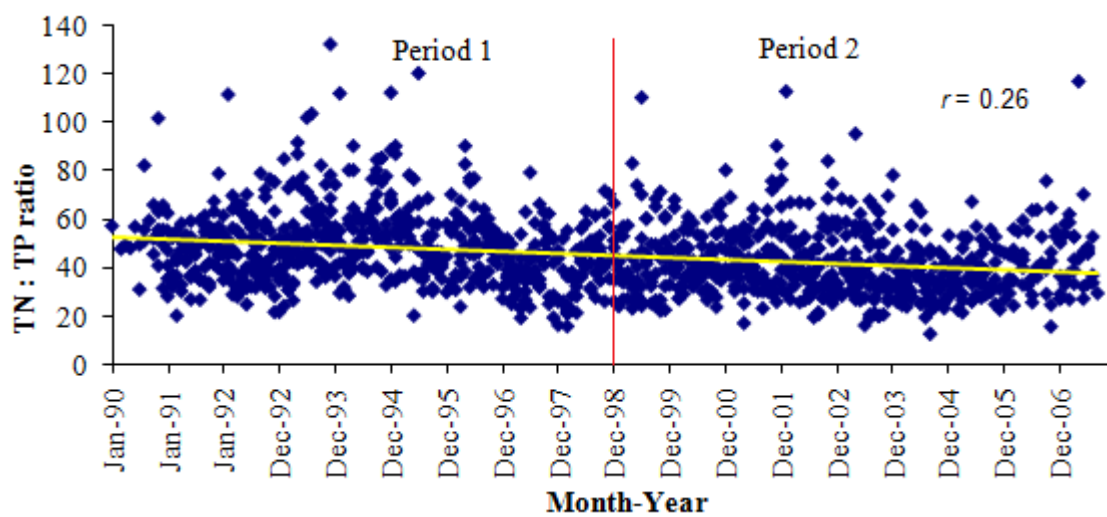


Figure 4. Plot of TN:TP ratios for 10 oligotrophic and mesotrophic lakes for the time period 1990-2007.

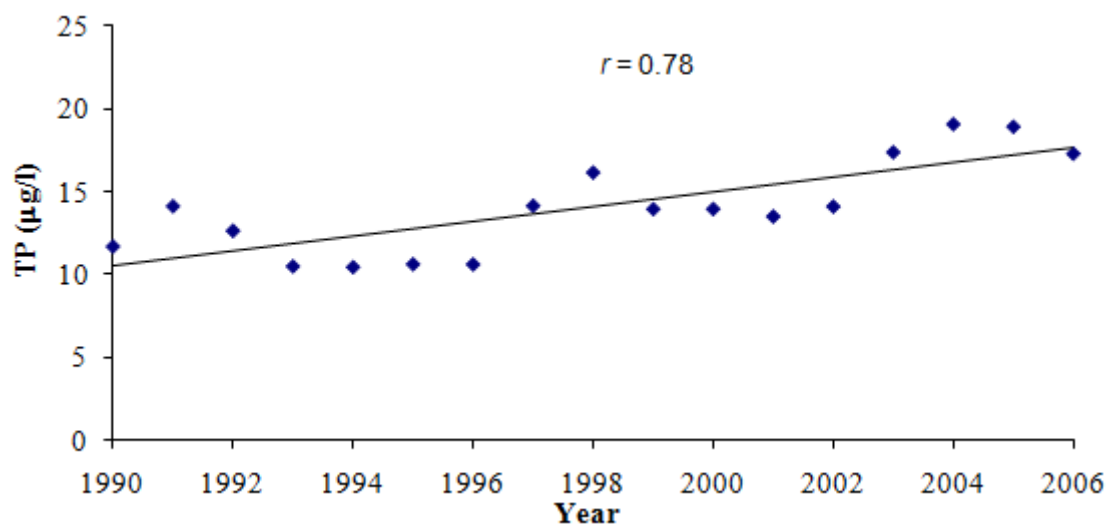


Figure 5. Plot of annual averages of TP concentrations for the time period 1990-2006.

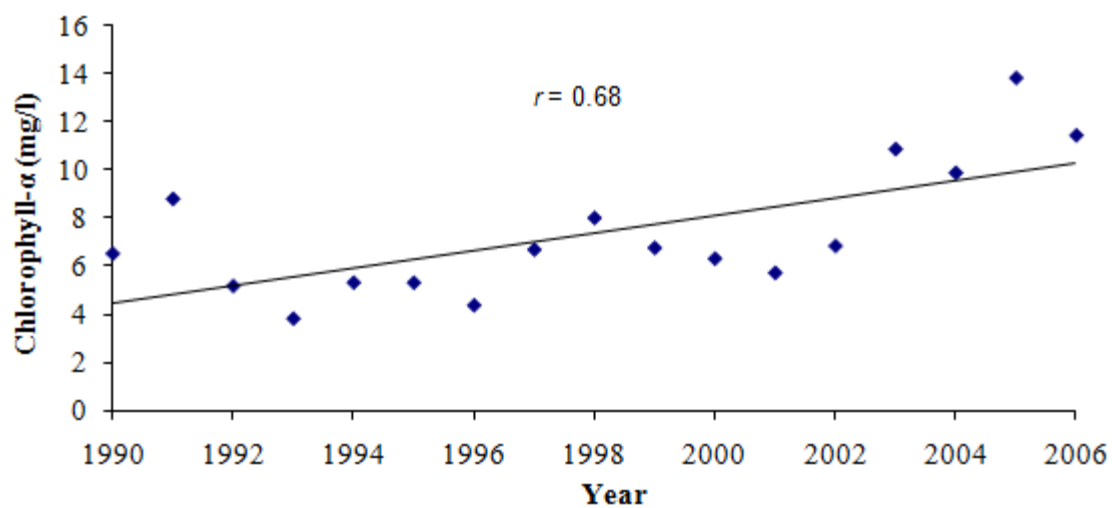


Figure 6. Plot of annual averages of chlorophyll- α concentrations for the time period 1990-2006.

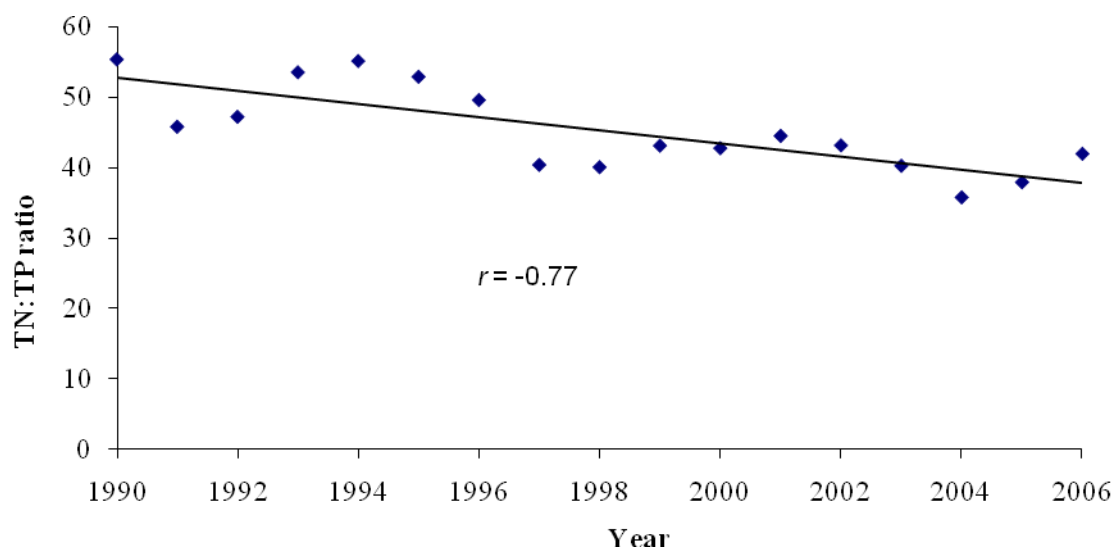


Figure 7. Plot of annual averages of TN:TP ratios for the time period 1990-2006.

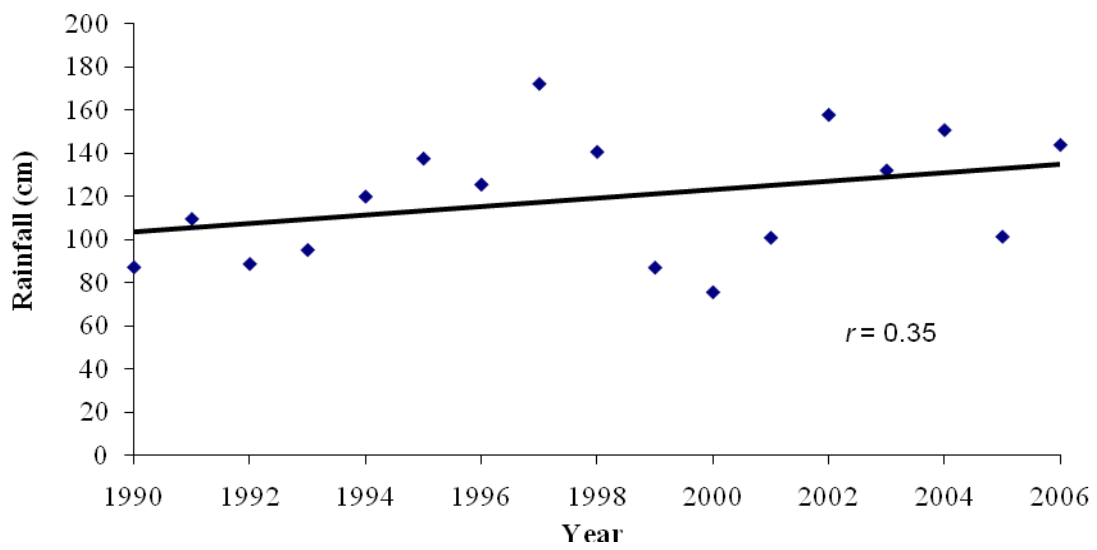


Figure 8. Plot of annual cumulative rainfall at the Tampa Airport for the Interval 1990 – 2006.

HIGH TEMPORAL RESOLUTION ASSESSMENTS OF TAMPA BAY WATER QUALITY USING SATELLITES

C. Hu, Z. Chen, F. Muller-Karger, M. Luther & C. Kovach

ABSTRACT

The present monitoring of Tampa Bay water quality is largely based on monthly *in situ* surveys. Using data collected from the Moderate Resolution Imaging Spectroradiometer (MODIS), the Sea-viewing Wide Field-of-view Sensor (SeaWiFS), and the Advanced Very High Resolution Radiometer (AVHRR), we show the advantages and shortcomings of satellite remote sensing in estimating several water quality parameters. Sea surface temperature from MODIS and AVHRR showed Root-Mean-Square (RMS) difference of about 0.5°C from *in situ* measurements. Water turbidity from MODIS showed RMS uncertainties of <30% ($R^2=0.73$, $n=43$, 0.9-8.0 NTU). Similarly, using SeaWiFS data and a semi-analytical algorithm, we estimated water clarity (SDD, Secchi Disk Depth in meters) and 20% percent surface light depths (PSL) with RMS uncertainties also within 30% ($R^2>0.67$, $n=80$, $0.9<SDD<8$ m). The data were used to construct long-term time series to study the seasonal patterns and inter-annual variability. Compared with those from the *in situ* measurements, the satellite time-series showed stronger seasonal patterns (e.g., best water clarity in May) because of the increased number of observations (on average, more than once per week). Further, there appears to have been an overall improvement in water clarity from the period 1997-2001 to the period 2002-2007 for the lower bay, with concurrent improvement in seagrass coverage. The satellite-derived water quality data provides valuable information that complements the *in situ* surveys. However, several limitations remain, including: 1) the data accuracy significantly degraded for shallow waters (< 2-3 m) due to interference by light reflected from the bottom; and 2) we were unable to obtain reliable estimates of chlorophyll-a concentration or abundance of colored dissolved organic matter (CDOM). These issues will be addressed in the future. Our present effort is to routinely generate high quality products from the satellite sensors to help manage the bay's water quality.

INTRODUCTION

Estuaries provide vital ecological functions to many marine species, and provide important economical and environmental values to coastal residents. Some of the largest estuaries in the U.S. have experienced significant ecosystem changes over the past 40 years. In the state of Florida, Tampa Bay is the largest open-water estuary (~910 km²; Rosenshein et al. 1977). Historical records showed marked decline in water quality as well as seagrass extent (Lewis et al. 1985; Jackson et al. 2001). In 1990, the Tampa Bay National Estuary Program (TBNEP) was established to coordinate and integrate efforts to restore and protect the Bay. In 1996, the TBNEP developed a Comprehensive Conservation and Management Plan (CCMP) which focused on restoration of seagrass to levels similar to those observed in the 1950s by reducing nutrient (primarily nitrogen) inputs into the Bay (Janicki and Wade 1996). Water quality of the Bay has gradually improved, and some of the seagrass has recovered (Johansson 2000; Tomasko et al. 2005).

Based on the approach recommended and routinely exercised for all Gulf of Mexico estuarine waters by the US EPA (EPA 2001; 2005), a decision support system (DSS) has been developed and used to assess the effectiveness of management activities (e.g., nutrient load regulations) for Tampa Bay for 10 years through the use of a water quality decision matrix (WQDM; Janicki et al. 2000; 2001). The WQDM is based on annual means of monthly water quality measurements at 54 fixed stations in the four bay segments: Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay, and Lower Tampa Bay.

Most of these measurements have been conducted by the Environmental Protection Commission of Hillsborough County's (EPCHC) Tampa Bay water quality monitoring program (Boler et al. 1991).

The EPCHC time-series data and similar data from other local programs provide the long-term water quality state of Tampa Bay, based on which management decisions are made (e.g., regulations on total nutrient load). However, due to the fast-changing nature of estuaries, it is unknown if these once-per-month measurements can capture all the variability and thus provide unbiased mean and anomaly water quality states. On the other hand, satellite remote sensing provides more synoptic and frequent measurements of the surface ocean (from the air-sea interface to the first optical depth), thus may potentially be used to complement the field surveys. Traditional satellite data do not have sufficient accuracy and ground resolution for estuaries, yet significant progress has been made in the past several years in algorithm development and data quality. Here, we review this progress for Tampa Bay using data from several satellite instruments. Our objective is to provide an overview of the current status of satellite remote sensing in monitoring Tampa Bay's water quality, including its strengths and weaknesses as well as ongoing efforts to make further improvements.

WATER QUALITY PARAMETERS FROM SPACE: PRINCIPLES AND PRACTICE

There are many water-quality parameters routinely monitored in estuaries, for example dissolved oxygen and pH; however, only a few can be estimated from remote sensing, which mainly include sea surface temperature (SST), water turbidity, water clarity, concentrations of chlorophyll-*a* (Chl), total suspended sediments (TSS), and colored dissolved organic matter (CDOM). They can be remotely measured because of their modulations on the electromagnetic properties of the surface ocean, from the thermal-infrared (for SST) to the visible and near infrared (ocean color, for all other water quality parameters).

Remote sensing of water quality started in the 1970s with the launch of Landsat-1 (1972 – 1978), CZCS (Coastal Zone Color Scanner, 1978 – 1986), the AVHRR series (Advanced Very High Resolution Radiometer), followed by modern sensors such as SeaWiFS (Sea-viewing Wide Field-of-view Sensor, 1997 – present), MODIS (Moderate Resolution Imaging Spectroradiometer, 1999 – present for Terra and 2002 – present for Aqua), MERIS (Medium Resolution Imaging Spectrometer, 2002 – present), and others.

The algorithms to derive SST were published several decades ago and recently reviewed by Martin (2004). The multi-channel sea-surface temperature (MCSST) algorithm is based on the calibrated sensor radiances (McClain et al 1985):

$$SST = C_1 T_4 + C_2(T_4 - T_5) + C_3(T_4 - T_5)(\sec\theta - 1) + C_4, \quad (1)$$

where T_4 and T_5 are the blackbody temperatures of channels 4 and 5, θ is the satellite zenith angle, and C_1 to C_4 are the coefficients determined by least square regression between satellite-estimated SST and *in situ* SST. More recently, a non-linear algorithm (NLSST), which is similar to the MCSST algorithm but with coefficients adjusted to account for variations in atmospheric water vapor, was proposed and used to generate SST products (Walton et al, 1998). For most open ocean waters the RMS uncertainty in the retrieved SST is generally smaller than 0.5°C.

Deriving water quality parameters from satellite ocean color measurements is more complicated than SST. There are generally two steps involved in this process. The first is to derive the spectral water-leaving reflectance from the satellite measurements (Gordon 1997):

$$\rho_w(\lambda, \Theta) = [\rho_t(\lambda, \Theta) - \rho_r(\lambda, \Theta) - \rho_a(\lambda, \Theta)]/[t(\lambda, \Theta) t_0(\lambda, \Theta)], \quad (2)$$

where λ is the wavelength, Θ represents the solar/viewing geometry (solar zenith, satellite zenith, and relative azimuth), ρ_t is the satellite-measured reflectance (after adjustment for ozone and other gaseous absorption), ρ_r is the reflectance due to Rayleigh (molecular) scattering, ρ_a is the reflectance due to aerosol scattering (including aerosol-Rayleigh interactions), t_0 is the diffuse transmittance from the sun to the target, and t is the diffuse transmittance from the target to the satellite. For simplicity the correction of sun glint and whitecaps (foams) is omitted here. The process described in Eq. (2) is called atmospheric correction, after which the spectral water-leaving reflectance ($\rho_w(\lambda, \Theta)$) is used in a bio-optical inversion algorithm to derive the various water quality parameters. These algorithms are derived either empirically through least-square regression (e.g., O'Reilly et al. 2000) or semi-analytically based on radiative transfer theory (e.g., Lee et al. 2002).

Significant progress has been made in the past two decades in ocean color algorithm and product development for the open ocean (e.g., McClain et al. 2004, and references therein). However, for coastal and especially estuarine waters, due to the complex optical properties in both the water and the atmosphere, reliable algorithms/data products are scarce, and most of them are through empirical regressions (e.g., Miller et al. 2005, and references therein). Such empirical approaches, when applied to different water bodies, often need to be tuned and validated because of the unique optical characteristics (for example, spectral shapes of CDOM absorption and particle backscattering) in each region. Adding to this lack of uniformity in algorithm/product is the limit in spatial resolution. Most ocean color instruments (SeaWiFS, MODIS, and MERIS) are designed to have ~1-km resolution at satellite nadir. This is often inadequate for small estuaries.

Recently, through support of NASA, NOAA, and USGS we have developed several robust water quality data products for Tampa Bay using AVHRR, SeaWiFS, and MODIS (Hu et al. 2004; Chen et al. 2007a, b; Hu et al. 2009). Table 1 presents a brief summary of the available data products, which are described in more detail below.

Table 1. Remote sensing water quality data products for Tampa Bay, Florida. a_{CDOM} (443 nm) is the CDOM absorption coefficient at 443 nm, a proxy for CDOM concentration.

Data Product	Resolution	Frequency*	Sensors	Period	Quality
SST (°C)	1000 m	> 2/week	MODIS, AVHRR	1993 -	high
Turbidity (NTU)	250 m	> 1/week	MODIS	1999 -	high
Clarity (SDD, m)	1000 m	> 1/week	MODIS, SeaWiFS	1997 -	high
Chl (mg m ⁻³)	1000 m	> 1/week	MODIS, SeaWiFS	1997 -	low
a_{CDOM} (443 nm, m ⁻¹)	1000 m	> 1/week	MODIS, SeaWiFS	1997 -	low

*These are modest estimates based on cloud cover statistics and rigorous data quality control.

REMOTE SENSING WATER QUALITY DATA PRODUCTS

Sea surface temperature (SST)—Using data collected by AVHRR, MODIS, and marine stations, Hu et al (2009) developed a practical cloud filter to remove cloud contaminations of the SST data products. The cloud filter used a 10-year climatology and ± 3 days statistics to flag suspicious image pixels. Results showed that for individual images RMS uncertainty is <1°C for SST between 18 and 32°C, and for weekly mean images RMS uncertainty is reduced to <0.6°C with a bias of -0.3°C. For weekly anomaly images RMS uncertainty is <0.5°C with near-zero bias. The weekly and monthly data products (mean and anomaly) have been generated and

updated weekly at the University of South Florida, with all products made compatible to Google-Earth and available online at http://imars.usf.edu/merged_sst (Figure 1).

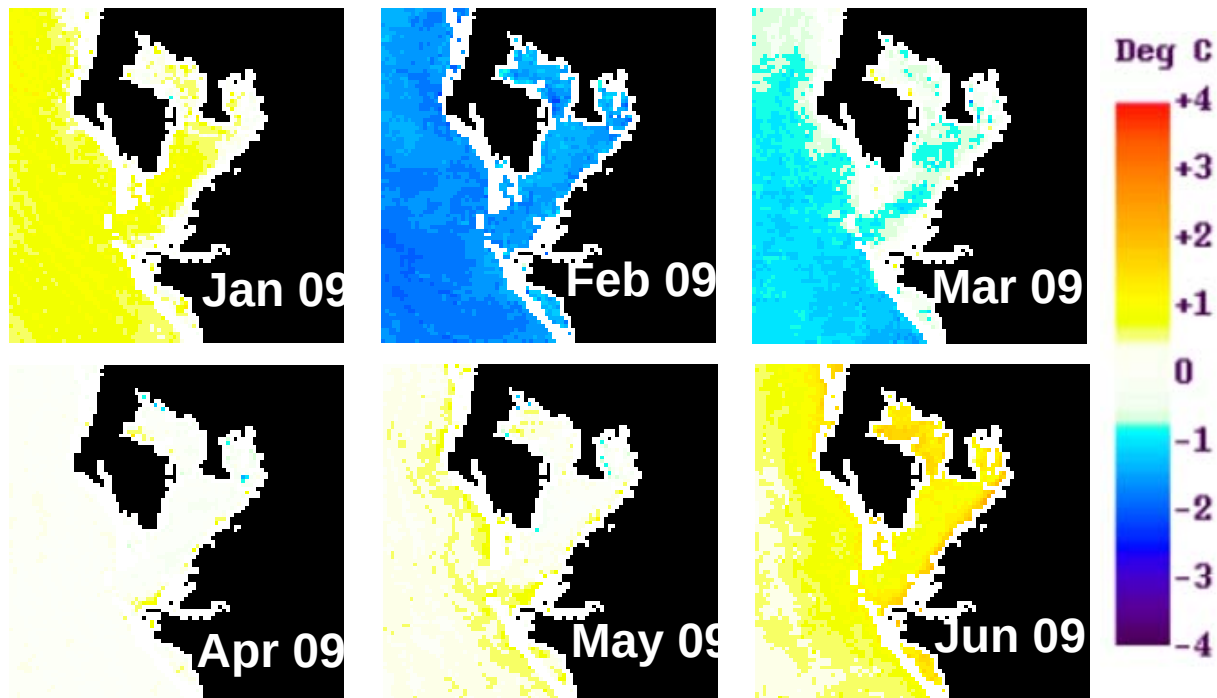


Figure 1. SST monthly anomaly images for Tampa Bay, Florida, derived from AVHRR and MODIS. Weekly and monthly mean/anomaly images from 1993 to date can be accessed at http://imars.usf.edu/merged_sst. These products are fully Google-Earth compatible, so a user can load the image in Google-Earth and navigate to any particular locations. The algorithms and accuracy assessment can be found in Hu et al. (2009).

Water turbidity—Turbidity is a water quality index that serves as a proxy for total suspended solids or sediments (TSS). In the field, it has been measured monthly by the EPCHC using a Hach[®] Model 2100N Turbidimeter. Using a simple atmospheric correction and MODIS 250-m resolution data, Hu et al. (2004) and Chen et al. (2007a) found an excellent relationship between MODIS-derived reflectance and *in situ* turbidity (Figure 2). Using this relationship, time series of MODIS turbidity can be generated at any location in Tampa Bay when water depth is >2.8 m (for shallower waters, the bottom reflection interferes with the algorithm and thus degrades the performance. This represents a limitation to the remote sensing algorithm); Figure 3 shows two examples. Because of multiple MODIS measurements during any individual month, MODIS-derived turbidity is more stable than *in situ* turbidity (one measurement per month), with the latter being more prone to episodic events. Such a difference can also be visualized by the monthly “mean” turbidity maps generated by the two methods (Figure 4). While they show similarity in the general distribution patterns, there are also differences in several bay segments (e.g., Lower Tampa Bay). Note that the MODIS turbidity maps in Figure 4 also show substantial interannual variability due to changes in wind forcing (Chen et al. 2007a).

Currently, these products are being used for research to understand the driving factors that regulate spatial/temporal changes in turbidity. The validated products will be generated automatically and made available online (similar to the SST products shown above) in future efforts.

Figure 2. Relationship between MODIS derived reflectance and *in situ* turbidity. The former was determined using MODIS 250-m resolution data and a simple atmospheric correction algorithm (Hu et al. 2004; Chen et al. 2007a), and the latter was determined by the EPCHC using with a Hach® Model 2100N Turbidimeter. Figure adapted from Chen et al. (2007a).

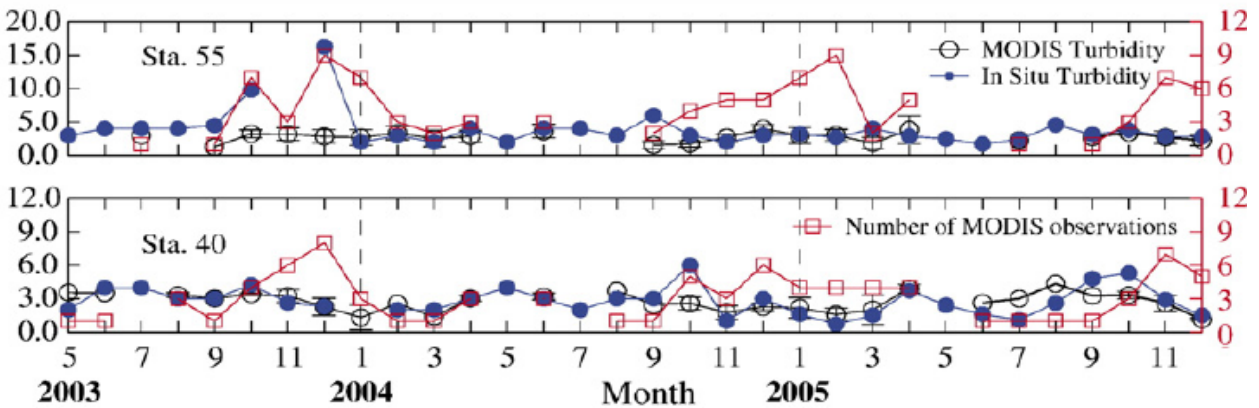
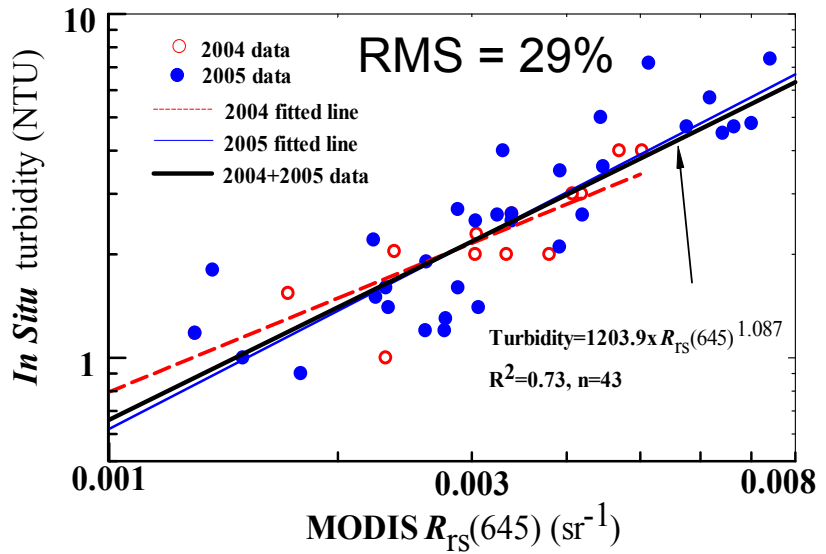


Figure 3. Monthly turbidity at two stations in Hillsborough Bay and Old Tampa Bay, respectively, from *in situ* measurements (once per month) and from MODIS estimates (1-9 measurements per month). The high *in situ* turbidity in October and December 2003 at Station 55 is apparently due to episodic events, which cannot represent the monthly mean conditions. In contrast, because of the multiple measurements per month, MODIS provides an improved measure of the monthly conditions. Figure adapted from Chen et al. (2007a)

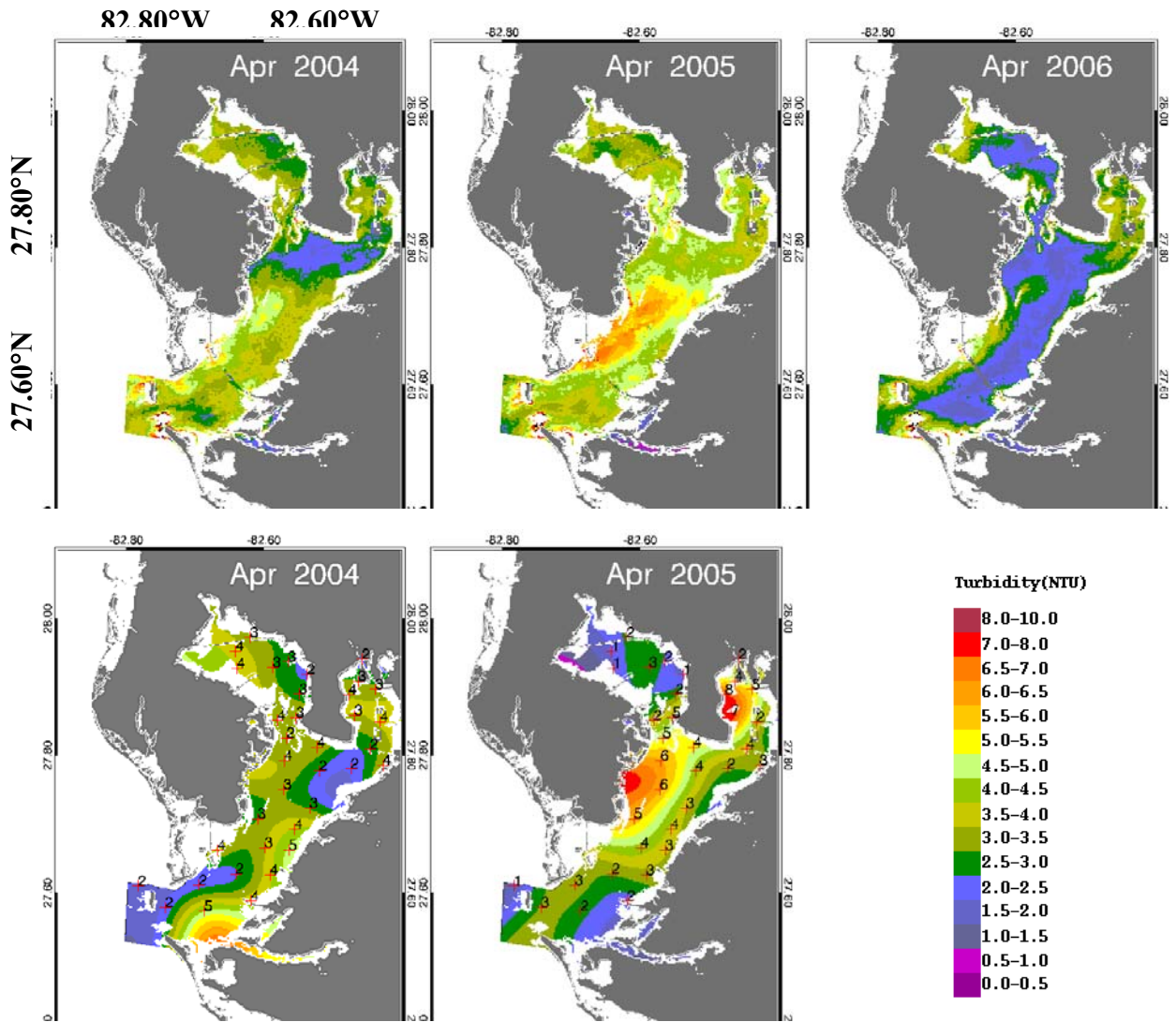


Figure 4. Monthly turbidity maps derived from MODIS (top panels) and *in situ* measurements (bottom panels). The maps show substantial spatial differences and interannual variability. There is also a difference between the MODIS and *in situ* observations for the same month, which is primarily due to the different sampling frequencies (one *in situ* measurement per month versus multiple MODIS measurements per month) Figure adapted from Chen et al. (2007a)

Water clarity—In Tampa Bay, water clarity has been measured by the EPCHC every month using a Secchi Disk. A black and white disk is lowered into the water, and the depth at which the disk is not visible is called Secchi Disk Depth (SDD, in meters). The technique is simple, easy, and low cost, but it is also subjective and depends on the observer; however, it provides a quick and crude measurement of water clarity (Kirk 1994).

In satellite remote sensing, water clarity can be defined as the light attenuation coefficient, which can be derived either empirically (Mueller 2000) or semi-analytically (Lee et al. 2005). The empirical algorithms work well in most open-ocean waters where light attenuation is small, but may create large uncertainties when applied to estuaries. In contrast, semi-analytical algorithms reduce the uncertainties significantly because more wavelengths (in particular, the red wavelengths) are used in the algorithm, thus enabling more accurate retrievals for a larger water clarity range. Figure 5 shows that while the empirically-derived $K_d(490)$ (diffuse attenuation

coefficient at 490 nm) correlates poorly with *in situ* SDD, the semi-analytically derived $K_d(490)$ can be used as a robust index for SDD between 1 and 10 m.

Figure 5. Relationship between SeaWiFS-derived water attenuation coefficient at 490 nm ($K_d(490)$) and *in situ* Secchi Disk Depth. The former was determined using SeaWiFS data and two bio-optical inversion algorithms (EA or empirical algorithm, and SA or semi-analytical algorithm), and the latter was determined by using the EPCHC Secchi Disk measurements. Figure adapted from Chen et al. (2007b)

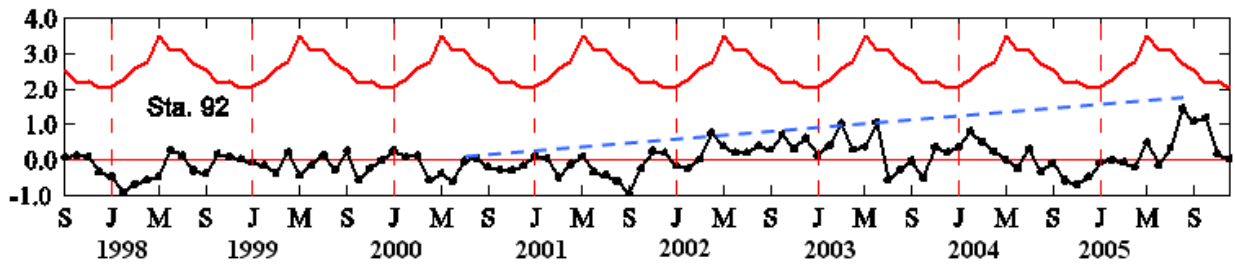
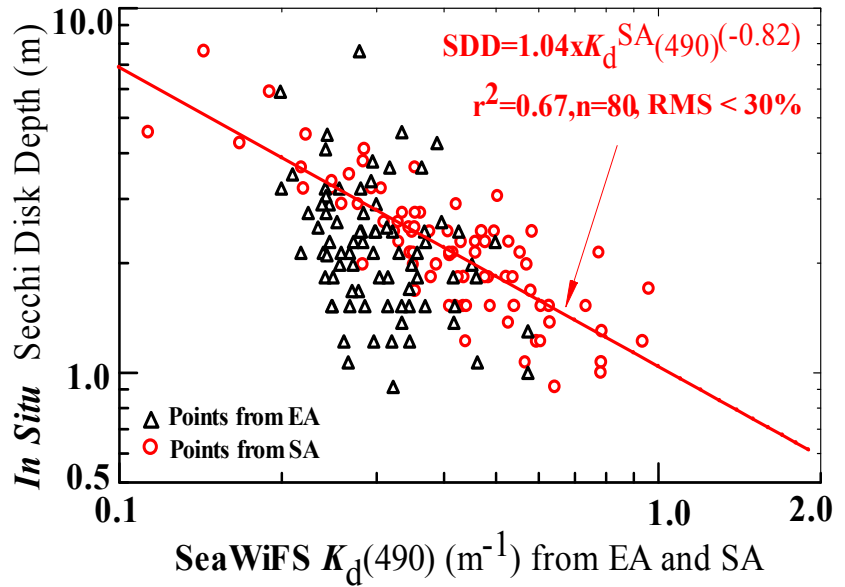


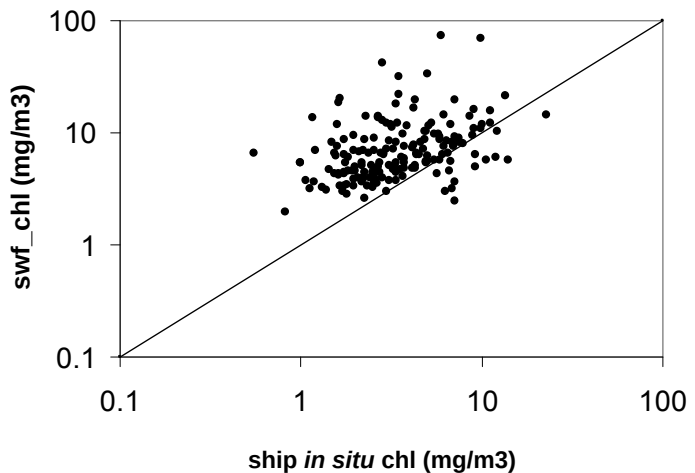
Figure 6. SeaWiFS-derived monthly water clarity (in Secchi Disk Depth, meters) at a Tampa Bay station in the Lower Tampa Bay. The solid red line shows the 8-year climatology, while the black line shows the monthly anomaly SDD (i.e., monthly mean minus climatology). There appears to be a slightly increasing trend in recent years (2001 – 2005). Figure adapted from Chen et al. (2007b)

Similar to the MODIS turbidity data products, SeaWiFS-derived SDD products provide long-term mean and anomaly state of water clarity for any given location when water depth is >2.8 m. Figure 6 presents an example for a Lower Tampa Bay station, where the 8-year monthly climatology (red line) shows seasonal cycles, with the clearest water (highest SDD) in May. The monthly anomaly SDD at this location shows that water clarity is better after 2001, as compared with that in the 1998–2000 periods. Whether this “trend” continues in the future requires further investigation, as well as whether these changes can best be related to management actions in the bay, or to changes in climatological conditions (e.g., rainfall, wind). These SeaWiFS water clarity products (SDD) can be used to assess Tampa Bay water clarity in a semi-automatic fashion once a system is implemented for data processing in near real-time.

Chlorophyll-a concentration and CDOM absorption—Chlorophyll-a concentration (Chl in mg m^{-3}) is an important index for biomass, primary productivity, and coastal eutrophication. The TBNEP nutrient reduction plan has specific goals to keep Chl below certain threshold values.

Unfortunately, because both CDOM and suspended sediments strongly modulate the blue-green light from which Chl is estimated, it is inherently difficult to obtain reliable estimates of Chl in CDOM-rich and/or sediment-rich waters (Figure 7). CDOM absorption coefficient at 443 nm ranged from about 0.5 m^{-1} in the dry season (winter and spring) to 3 m^{-1} in the wet season (summer and fall) (Chen et al, 2007c), equivalent to absorption by phytoplankton for Chl, 60 mg m^{-3} and 1400 mg m^{-3} , respectively. In Tampa Bay, Chl is $<10 \text{ mg m}^{-3}$ most of the time, and TSS ranged from 2 to 20 mg L^{-1} (Chen et al. 2007c). Thus, the modulation of blue-green light by CDOM and TSS creates difficulty in estimating Chl from the satellite data. Similarly, it is inherently difficult to estimate CDOM in this complex environment (especially in the dry season when CDOM is relatively low) because both phytoplankton and TSS can contribute substantially to the blue light modulation.

Figure 7. Comparison between Chl obtained from concurrent shipboard measurements and SeaWiFS estimates between 1998 and 2005 for Tampa Bay, Florida. There is a general overestimate by SeaWiFS, while there appears to be no systematic trend in this overestimate.



Clearly, more work is required to make improved estimates of Chl and CDOM in Tampa Bay from satellite measurements. In the meantime, solar-stimulated fluorescence data from MODIS and MERIS may be used as a surrogate for Chl (e.g., Hu et al. 2005). The fluorescence line height (FLH) data product from MODIS and MERIS is based on red and NIR wavelengths that are immune to interference by CDOM. When TSS is low (this can be verified by examining the RGB true color imagery), FLH may be used to show spatial patterns of biomass. Figure 8 shows several MERIS FLH images (300-m resolution) where phytoplankton bloom patterns can be inferred. There is a noticeable disparity in the various bay segments. Old Tampa Bay and Hillsborough Bay experienced more bloom events than Middle Tampa Bay and Lower Tampa Bay. This is confirmed by concurrent field data collected by the EPCHC. The FLH data may be combined with *in situ* Chl data to develop an empirical algorithm for a FLH-based Chl data product in the near future.

SUMMARY

Compared with *in situ* surveys for estuarine water quality, satellite remote sensing provides more synoptic and frequent measurements to reduce potential spatial/temporal aliasing when assessing the mean and anomaly water quality states. However, it is not straightforward to generate such remotely sensed water quality data products due mainly to the lack of reliable algorithms to convert the satellite signal to meaningful parameters. Several recent advances led to reliable products of SST, water turbidity and water clarity for Tampa Bay, Florida. The latter two work well only for relatively deep waters (bottom depth $>2.8 \text{ m}$), and estimation of Chl and CDOM

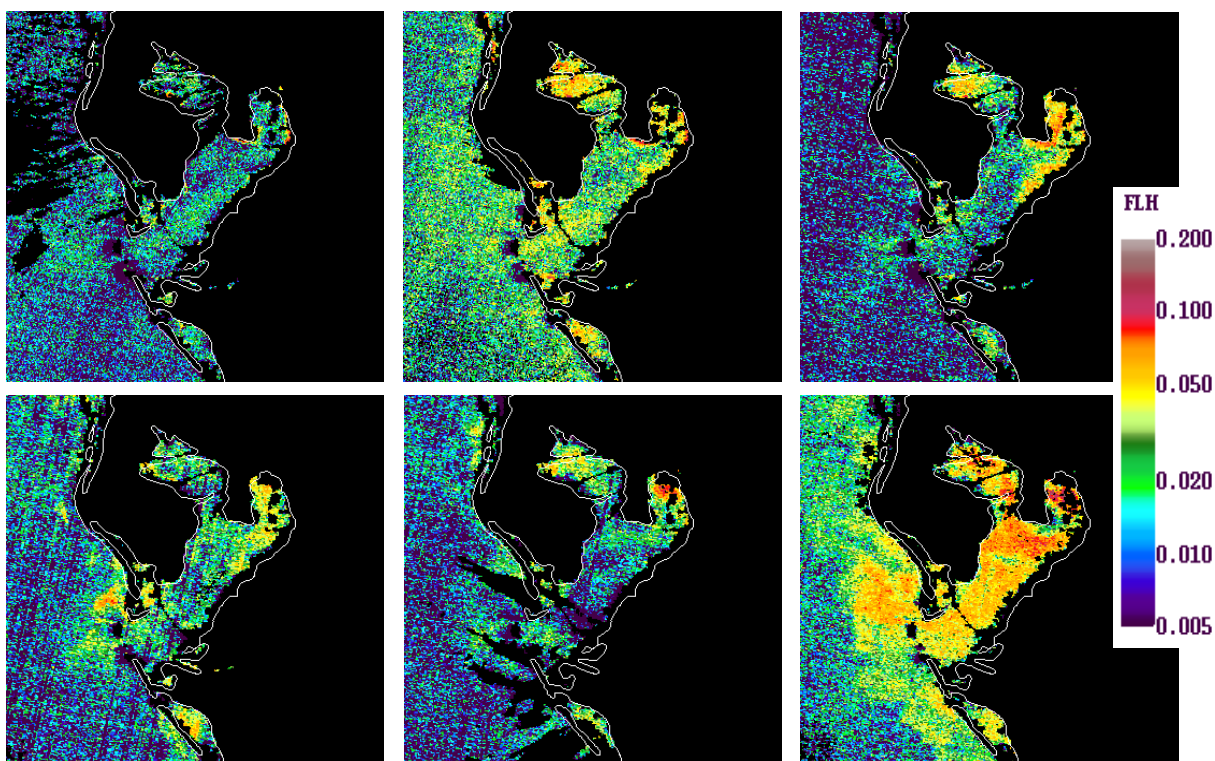


Figure 8. MERIS fluorescence line height (FLH, $\text{mW cm}^{-2} \mu\text{m}^{-1} \text{sr}^{-1}$) showing bloom patterns in Tampa Bay, Florida. Old Tampa Bay and Hillsborough Bay experienced more blooms than other bay segments.

still requires dedicated effort. While the validated SST mean and anomaly products are being generated and made available online in an automatic fashion, similar products should be generated for all other remotely-sensed water quality products. At the current time, remote sensing products such as the MODIS and MERIS FLH provide qualitative information on phytoplankton blooms in this optically-complex estuary. Collectively, all these remote sensing products (validated or provisional) provide complementary information to the *in situ* surveys to better quantify the water quality states in Tampa Bay.

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LONG-TERM AND SEASONAL TRENDS IN PHYTOPLANKTON PRODUCTION AND BIOMASS IN TAMPA BAY, FLORIDA

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ABSTRACT

Phytoplankton production is a basic process in aquatic ecosystems that converts inorganic carbon into organic matter and provides an important indicator of trophic state. The City of Tampa Bay Study Group maintains a 32 year-long monthly record of phytoplankton production rates and biomass (chlorophyll-a) in Hillsborough Bay (HB) and Middle Tampa Bay (MTB), and a recent record during the last nine years for Old Tampa Bay (OTB). Production is measured using the classic *in situ* ^{14}C method with samples incubated vertically in the water column. Annual production rates during the most recent decade are about 410gCm^{-2} for HB, 350gCm^{-2} for MTB, and 390gCm^{-2} for OTB. The current rates for the two former bay segments are near half the rates measured during 1980-1985. Reductions in biomass have been greater; current HB and MTB concentrations are near 70% lower than those measured during 1980-1985. Water column-averaged chlorophyll-a concentrations during the most recent decade are about 10mgm^{-3} for HB, 5.8mgm^{-3} for MTB, and 6.3mgm^{-3} for OTB. The decreases in production and biomass, and also reductions in phytoplankton abundance, are reflected in a large reduction in anthropogenic nitrogen loading to the bay that primarily occurred during the late 1970s and early 1980s. This is strong evidence the long-term trend in Tampa Bay phytoplankton production and biomass has been regulated by the supply of nitrogen from external sources. However, similar to other productive estuarine and coastal systems, in-bay recycling provides a substantial fraction of the nitrogen needed to sustain the observed daily production rates. The vertical distribution of phytoplankton production in HB and MTB has shifted during the study period, most noticeable during the wet summer seasons, from a large proportion of total water column production occurring in the upper meters to a more even distribution with depth. Further, seasonal water column production generally reaches maximum during the summer months and follows variations in water temperature. Finally, a comparison of current phytoplankton and seagrass carbon production in the bay segments studied indicates that pelagic phytoplankton dominates production and will most likely continue to do so in the future.

INTRODUCTION

The phytoplankton community is the major producer of organic matter to most of the world's estuaries, including Tampa Bay, and is thus an important contributor to the estuarine food web in those systems. This organic matter sustains diverse estuarine ecosystems by providing energy to secondary producers and higher trophic levels.

The autotrophic conversion process of inorganic substances into organic matter by the phytoplankton is measured as carbon production rates. These measurements provide information on the trophic state of water bodies which cannot directly be derived from instantaneous phytoplankton biomass determinations. Further, the measured production rates can provide information about the physiological state of the phytoplankton population that can be related to estuarine conditions and processes, including water column light availability and nutrient uptake and recycling.

Tampa Bay is one of the largest river-dominated estuaries in the southeastern United States with an open water surface area of about 1000km^2 and with a water volume of nearly four billion m^3 . The current average daily standing crop of the phytoplankton contained within this large water body is, when expressed as carbon weight, close to 2000 metric tons. A fundamental understanding of phytoplankton carbon production rates is required to, at least at a basic level,

comprehend the complex estuarine processes involved with the maintenance and utilization of this biomass.

The purpose of this report is to build on the early Tampa Bay phytoplankton community information that was presented at the first Tampa Bay Area Scientific Information Symposium (BASIS) near 30 years ago (Johansson et al. 1985); and also to update more recent information presented at BASIS 4 in 2003 (Johansson 2005).

METHODS

The City of Tampa Bay Study Group (BSG) maintains a 32 year-long monthly record of phytoplankton production rates in Hillsborough Bay (HB) and Middle Tampa Bay (MTB), and a recent record during the last decade for Old Tampa Bay (OTB). Two locations have been sampled in HB (COT4-depth 3.8m and COT12-depth 5.4m) since 1978, one location in MTB (COT13-depth 6.4m) since 1979 and one location in OTB (COT40-depth 4.8m) since 2000 (Figure 1). For this report, measurements from the two HB stations have been combined to reflect the overall conditions in HB.

Production measurements were conducted using the classic *in situ* ^{14}C method developed by Steemann Nielsen (1952) and modified by Strickland and Parsons (1968). Water collected at 1m intervals from surface to near bottom is screened through a 202 μm mesh to remove larger grazers. The water is then dispensed into glass bottles, which are inoculated with a measured amount of sodium [^{14}C] bicarbonate solution. Triplicate bottles from each depth level are suspended in the water column at the depth of collection for a 3hr to 5hr day-light period. After the incubation period, the samples are brought to the laboratory and the phytoplankton community is captured on 0.45 μm pore-size membrane filters. The radioactivity of the filters is determined in a liquid scintillation counter. The hourly carbon production rate during the incubation period is calculated and then corrected to daily production rates using daily solar radiation data for Tampa.

Also, un-screened aliquots are taken from the water samples at each depth level for phytoplankton biomass (chlorophyll-a) estimates, and from the surface samples only for microscopic determinations of phytoplankton abundance and composition. Additional measurements include: standard multi-probe physical parameters (temperature, salinity, dissolved oxygen and pH), PAR light attenuation, beam attenuation at 660nm, Secchi disk depth, turbidity; and concentrations of colored dissolved organic matter (CDOM), total suspended solids (COT 4 only), total carbonate and dissolved ammonia.

RESULTS

Long-Term Trends in Phytoplankton Production and Biomass:

Trends in annual phytoplankton production and biomass, expressed on an areal basis, for the period of record and for the three bay segments studied are shown in Figure 2. It is clear that large reductions in both production and biomass have occurred in HB and MTB since the late 1970s and early 1980s. A long-term record for production rates is not available for OTB, however, the recent OTB record tracks HB and MTB closely. Further, based on the long-term Environmental Protection Commission of Hillsborough County's (EPCHC) chlorophyll-a record for OTB (Boler 2002), it is known that large phytoplankton biomass reductions have also occurred in this bay segment (Table 1). It is reasonable to assume that phytoplankton production also has decreased in OTB in a similar manner as seen in HB and MTB.

A comparison of production rates and biomass concentrations in the bay segments studied (Table 1) show that total water column production in HB and MTB during the current decade is near half of that measured during the 1980-1985 period. In contrast, water column biomass concentrations

have decreased almost three-fold. Similar changes have also occurred in surface production and biomass for the two bay segments between the two time periods. However, estimated biomass in OTB has apparently only been reduced by a third between the early and current periods.

Long-Term Trends in HB Phytoplankton Parameters and External DIN Loading:

Hillsborough Bay is the only Tampa Bay segment with a long-term record of dissolved inorganic nitrogen (DIN) loading from external sources; the form of nitrogen most readily available for phytoplankton growth (Johansson 2005). A comparison of HB records for DIN loading and phytoplankton production and biomass (Figure 3) indicates that major reductions, in both production rates and biomass concentrations have followed the substantial DIN loading reductions during the late 1970s and early 1980s. However, biomass concentrations apparently responded faster than production rates to the DIN reductions.

Biomass was substantially reduced in 1984, about five years following the initiation of the large DIN loading reduction. At that time, the amount of a seasonally-occurring large filamentous planktonic blue-green alga (*Schizothrix* sp.) was greatly reduced in HB and other bay segments (Johansson and Lewis 1992; Dixon et al. 2009). Phytoplankton production, in contrast, was maintained at high levels throughout the DIN loading transition period and was not substantially reduced until 1988.

The reduction in biomass was clearly linked to the decline in blue-green biomass and abundance; however, it is less certain why production remained elevated for four years following the substantial decline in biomass. An examination of seasonal production patterns for HB during the four-year period shows that summer production rates were very high. These high rates may have been maintained by temperature-regulated internal nitrogen recycling from the high-organic sediments present in HB (see Johansson and Squires 1989).

Long-Term and Seasonal Vertical Distribution in Phytoplankton Production and Biomass:

The collection of water samples at one meter intervals from near surface throughout most of the water column allows for long-term and seasonal evaluations of trends in the vertical distribution of both phytoplankton production and biomass.

Long-Term Trends:

The vertical distribution patterns of phytoplankton production and biomass have changed very differently during the long-term in both HB and MTB (Figures 4 and 5). Phytoplankton biomass shows a near uniform vertical distribution for the two bay segments during the period of study. This distribution pattern indicates that the water column was well-mixed and has lacked density stratification on most sampling occasions and that the large historical reductions in biomass generally occurred throughout the water column. Vertical chlorophyll-a patterns in Tampa Bay have also been discussed by Johansson (2006).

In contrast to the vertical patterns seen in phytoplankton biomass, the long-term distribution pattern for production rates shows large vertical differences, with higher rates, as expected, near the surface and lower rates deeper in the water column. It is evident that the major reductions in phytoplankton production rates during the period of record have mainly occurred in the upper portion of the water column. The vertical pattern seen for production indicates that water column light attenuation is controlling production with depth. Further, it appears that reduction in light penetration caused by the phytoplankton community itself (i.e. self-shading) plays an important role in the vertical distribution of production rates. Of course, other factors also affect water column light attenuation including turbidity, CDOM concentrations and the water itself (Kirk 1994). Light reductions caused by these constituents would also be reflected in the seasonal distribution patterns of production rates.

It is also apparent, and expected, that a greater fraction of the water column production currently occurs at deeper depths in HB and MTB than during the early period of higher production and biomass. This is likely because of increased water clarity associated with reductions in algal biomass.

Seasonal Trends:

The average monthly vertical distribution patterns of phytoplankton production rates and biomass concentrations for the three bay segments during the last decade are shown in Figures 6, 7 and 8. It is evident that a strong seasonal signal is present in all three bay segments for both parameters and that, as expected, high rates and concentrations generally occur during summer and early fall. Similar to the long-term vertical distribution patterns, the seasonal distribution of biomass has relatively small differences with depth while production has large vertical differences, specifically during the warm period. Further, the vertical production pattern indicates that water column light attenuation is also controlling the seasonal production with depth.

The seasonal patterns for biomass are similar between the three bay segments with maximum concentrations generally found from August through October. The seasonal patterns of phytoplankton production in the upper portion of the water column are, on the other hand, quite dissimilar between OTB and the other two bay segments. Both HB and MTB generally have maximum surface rates between July and October, while OTB has a prolonged season of elevated rates from late spring to fall. Further, there appears to be a distinct depression of near surface production rates in OTB in August. Reasons for the prolonged elevated production season in OTB and the August depression are not clear at this time.

Seasonal Variation in Production Rates and Abiotic Parameters: The seasonal variation in HB water column phytoplankton production rates and several abiotic parameters are shown in Figure 9 as monthly means for the recent 10-year period.

Total water column production peaks during a prolonged period in the summer and follows the seasonal variation in water temperature most closely; also, as expected, there is a general agreement between seasonal production and solar radiation (Day et al. 1989). The seasonal pattern in production does not appear to be strongly associated with the seasonal variations in external DIN loading to HB.

DISCUSSION

Long-Term Trends in Tampa Bay Phytoplankton Production and Biomass:

Tampa Bay phytoplankton production sustains the phytoplankton biomass present in the bay and, therefore, it could be expected that temporal changes noted in production rates would be relatively equivalent to changes in biomass. However, the current biomass concentrations in HB and MTB have decreased almost three-fold when compared to the early period of the record, while current production rates are near half of the early rates (Table 1). There are several explanations for the disproportional changes in phytoplankton production and biomass.

First, a major change in phytoplankton composition occurred in the upper bay segments in the mid 1980s when the abundance of the seasonally-occurring planktonic blue-green alga (*Schizothrix* sp.) was greatly reduced (Johansson and Lewis 1992; Dixon et al. 2009). Subsequent to the blue-green losses, the phytoplankton community has most often been dominated by diatoms and small flagellates, and biomass has been substantially reduced. The efficiency of the phytoplankton community to produce carbon per unit biomass may have changed with the change in communities. If the current diatom and flagellate community is capable of a higher carbon production per unit biomass than the blue-green dominated community, then a disproportional

increase in phytoplankton production relative to biomass would have resulted during the period of study. A comparison of ratios between production rates and biomass concentrations from samples collected near the surface in HB and MTB during the period of record show a general trend of increasing values, in support of this scenario. However, specific information of carbon production per unit biomass for the two major phytoplankton communities is lacking for Tampa Bay.

A second, and more easily substantiated scenario, is related to the documented improving trend of water column light penetration in the upper portions of Tampa Bay (Boler 2002; Johansson 2002; Greening and Janicki 2006). The improving light penetration, which to a great degree has been caused by a reduction in phytoplankton biomass, has deepened the photic zone. The deeper photic zone has allowed for a proportionally greater amount of production to now occur deeper than during the early period (Figures 4 through 8). Phytoplankton biomass, on the other hand, has been relatively homogenous with depth and the reduction in biomass has occurred throughout the water column. Therefore, as a consequence of these changes, a proportionally smaller reduction has occurred during the period of record in water column production relative to biomass.

Nutrient Limitation by the Tampa Bay Phytoplankton Community:

The finding of a strong positive association between the long-term HB trends of external DIN loading rates, and phytoplankton production rates and biomass concentrations (Figure 3) was anticipated based on earlier studies of phytoplankton biomass and nitrogen relationships in Tampa Bay (Johansson 1991; Janicki and Wade 1996) and for other estuaries as well.

The strong nitrogen control of the Tampa Bay phytoplankton community is also evident from ongoing nutrient addition experiments that the BSG has conducted since 1993 on the natural community found in the four major bay segments (Figure 1). The bioassay method used is modified from Fisher et al. (1992). Of 152 bioassay tests conducted to date, 149 have shown that nitrogen was the most limiting nutrient to phytoplankton growth (Figure 10). None of the tests have indicated that phosphorous was the most limiting nutrient; only three tests in OTB have shown a lack of response in growth to either nitrogen or phosphorous additions.

One of the three OTB tests which resulted in a “no response” to nitrogen and phosphorous additions was conducted in late August 2009 during an extensive dinoflagellate (*Pyrodinium bahamense*) bloom in this bay segment. *P. bahamense* dominated the test community; and it is possible that the dinoflagellate was dependant on organic nitrogen for growth and less so on inorganic nitrogen that was added to the bioassay vessels. It is well known that dinoflagellate species, as well as phytoplankton species from other groups, have the ability to assimilate organic forms of nitrogen (e.g. Glibert et al. 2006).

Finally, the finding that nitrogen is the most limiting nutrient to phytoplankton growth in the four major Tampa Bay segments has also been concluded from other bioassay studies. Results from a bay-wide study conducted in 1993 and 1994 by Vargo et al. (1994), using the green alga *Dunaliella tertiolecta* as assay organism, also showed that nitrogen was the most limiting nutrient and that additions of phosphorous did not enhance biomass.

Estimates of Nitrogen Loading Required to Sustain Phytoplankton Production:

The amount of nutrients that are needed to sustain an observed production rate can be estimated by assuming that the phytoplankton assimilate nutrients during the photosynthetic process in proportion to Redfield ratios (e.g. Boynton et al. 1995). Results from the calculations (Table 2) indicate that estimated historic (worst case) external total nitrogen (TN) loading to HB potentially could provide about 50% of the nitrogen needed to sustain the observed production rates during the early period of the record. The current external TN loading to HB can potentially provide about one third of the nitrogen required. The corresponding estimates for MTB and OTB (current

only) were much lower, indicating that the external TN loading to these bay segments may supply less than one tenth of the nitrogen required.

Although recognized as uncertain, the estimates generally agree with studies from other productive estuaries (e.g. Boynton et al. 2008; Nixon et al. 2009). They are also in general agreement with water quality model-derived estimates for Tampa Bay (Morrison et al. 1999; Wang et al. 1999). The model studies concluded that the external TN loads to Tampa Bay could support less than 20% of the observed Tampa Bay phytoplankton production and resulting biomass. The finding that the current external TN load to Tampa Bay only appears to supply a fraction of the observed daily production suggests that internal nitrogen recycling, occurring in both the sediments and the water column, may be a substantial nitrogen source. This conclusion appears supported by the finding that the elevated summer production in HB does not appear to be strongly associated with inputs of external DIN loading, but is better associated with water temperature and solar radiation (Figure 9). Temperature-regulated internal nitrogen cycling (e.g. Kemp and Boynton 1984) may therefore have contributed to the high summer production noted in HB.

Field studies of nitrogen releases from the sediments in HB, which were conducted in the mid 1980s by the BSG, also support the potential importance of internal nitrogen recycling in Tampa Bay (Johansson and Squires 1989). Those studies showed that nitrogen releases from sediments at that time had the potential to supply a substantial fraction (34%) of the observed phytoplankton production nitrogen demand. However, it is not certain that the fraction of nitrogen supplied by the sediments in HB has remained constant since those studies were conducted (see Nixon 2009). It could be expected that the nitrogen fraction supplied by the sediments would decrease during time as a result of the long-term reductions in external nitrogen loading to HB and a continuous depletion of the reactive sediment nitrogen supply (Johansson 1991; Wang et al. 1999).

The nitrogen supply estimates reported above suggest that a large fraction of the nitrogen needed to maintain the observed Tampa Bay daily production rates can be supplied by internal loading. Nevertheless, both direct measurements and model estimates have shown that the long-term trends in both phytoplankton biomass and production have been controlled by the external supply of nitrogen (Johansson 1991; Janicki and Wade 1996; Wang et al. 1999). Other potentially important sources of nitrogen to Tampa Bay have also been recognized, including in-bay nitrogen fixation and nitrogen transport from the Gulf of Mexico, however, these sources have received little study to date.

Phytoplankton Production in Tampa Bay and Other Estuaries:

A comprehensive review and comparison of phytoplankton production rates for estuaries worldwide was published by Boynton et al. in 1982. A section of the comparison, addressing river-dominated estuaries, has been modified here to include the mean daily rates during the recent decade for the three Tampa Bay segments studied (Figure 11).

It is evident that the current production rates for the three Tampa Bay segments fall within the high range of the listed estuaries. The annual mean rates are 210gCm^{-2} for all listed estuaries excluding Tampa Bay, and 385gCm^{-2} for the three Tampa Bay segments. Further, the rates measured in Tampa Bay are similar to those measured in Mid-Chesapeake Bay and Apalachicola Bay.

The annual range of production varies greatly between estuaries, but the seasons of minimum and maximum rates are generally similar for most of the listed estuaries, including Tampa Bay. Maximum production rates usually occur during summer and minimum rates during winter. The association of maximum production during summer is a pattern common to most well-mixed

estuaries. It has been suggested that the high summer phytoplankton production rates are supported by the increased availability of recycled nutrients, from both the sediments and the water column, that occur during the warm season (e.g. Boynton et al. 1982).

Estimated Current Annual Carbon Production of Phytoplankton and Seagrass:

The contribution of carbon production by different plant groups to the Tampa Bay system has been discussed previously (Pomeroy 1960; Johansson et al. 1985; Jensen and Gibson 1986; Lewis and Estevez 1988; Johansson 2005). These comparisons have emphasized the importance of the pelagic phytoplankton community to supply organic matter to the bay ecosystem. The Tampa Bay seagrass community (with epiphytes) and other plant communities, including macro-algae and benthic micro-algae, appear to be of secondary importance in these comparisons.

An updated comparison of carbon production by the phytoplankton and seagrass communities for the three bay segments studied is provided in Table 3. Only phytoplankton and seagrass carbon production are considered in the comparison. The contributions by macro- and micro-algae are clearly important in the shallow portions of the bay, however, their contribution to the total Tampa Bay system appears low (Pomeroy 1960; Johansson et al. 1985). Further, the comparison of phytoplankton and seagrass is of specific interest because of Tampa Bay ecosystem management goals. These goals emphasize the maintenance of phytoplankton biomass at levels appropriate for restoring seagrass areal coverage to historical levels (Johansson and Greening 2000; Greening and Janicki 2006).

The results from the comparison indicate, similar to the earlier estimates, that phytoplankton carbon production dominates system production of organic matter in the three bay segments studied. Further, based on the relatively steady but slow rate of current seagrass expansion in Tampa Bay (Greening and Janicki 2006; Avery et al. 2010) it can be expected that the pelagic bay habitat will continue to dominate system carbon production in the foreseeable future.

CONCLUSION

Annual Tampa Bay phytoplankton production rates during the most recent decade are about 410gCm^{-2} for HB, 350gCm^{-2} for MTB, and 390gCm^{-2} for OTB. These rates are near half of the rates measured during the 1980-1985 period, though a comparison of production rates between numerous estuaries world-wide indicate that the current Tampa Bay rates fall in the high range. Reductions in biomass have been greater; current HB and MTB concentrations are near 70% lower than those measured during 1980-1985. Water column-averaged chlorophyll-a concentrations during the most recent decade are about 10mgm^{-3} for HB, 5.8mgm^{-3} for MTB, and 6.3mgm^{-3} for OTB. The decreases in production and biomass, and also reductions in phytoplankton abundance, followed a large reduction in external DIN loading to Tampa Bay 30 years ago.

The depth distribution of production in HB and MTB during the period of record has shifted from a large proportion of total water column production occurring near the surface to a more even distribution with depth. The shift is most likely a result of improvements in Tampa Bay light penetration and a deepening of the photic zone. Further, self-shading by the phytoplankton population appears to be important in determining the depth distribution of production.

Total water column production in HB peaks during a prolonged period in the summer and follows the seasonal variation in water temperature most closely. As expected, there is also a general agreement between seasonal production and solar radiation. The seasonal production pattern does not appear to be strongly associated with the pattern in external DIN loading. Temperature-regulated internal nutrient cycling in HB may be an important contributor to the high summer production.

Estimates of the amount of nitrogen needed to sustain the observed daily Tampa Bay production rates indicate that internal nitrogen recycling processes may be an important supply source. However, direct measurements and model estimates have indicated that long-term trends in phytoplankton production and biomass have been controlled by the external supply of nitrogen. Other potentially important sources of nitrogen to Tampa Bay have also been recognized, including in-bay nitrogen fixation and nitrogen transport from the Gulf of Mexico, however, these sources have received little study to date.

The strong dependence on nitrogen by the Tampa Bay phytoplankton community is evident from long-term nutrient addition experiments. Of 152 bioassay tests conducted to date, 149 have shown that nitrogen was the most limiting nutrient to phytoplankton growth. None of the tests have indicated that phosphorous was the most limiting nutrient.

Finally, a comparison of current Tampa Bay phytoplankton and seagrass production indicates that the pelagic phytoplankton community dominates the production of organic matter. Based on the relatively steady but slow rate of current seagrass expansion in Tampa Bay it can be expected that the pelagic bay habitat will continue to dominate system carbon production in the foreseeable future.

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Table 1. Average annual water column (A) and surface (B) phytoplankton production rates and chlorophyll-a concentrations measured in HB, MTB and OTB over two time periods.

A

Water Column Phytoplankton Production ($\text{gCm}^{-2} \text{yr}^{-1}$)			
Bay Segment	Period		Reduction %
	1980-1985	1999-2008	
HB	750	410	45
MTB	630	350	44
OTB		390 ¹	

Water Column Chlorophyll-a (mgm^{-2})			
Bay Segment	Period		Reduction %
	1980-1985	1999-2008	
HB	170	47	72
MTB	110	37	65
OTB		30 ¹	

¹ 2000-2008

B

Surface Phytoplankton Production ($\text{gCm}^{-3} \text{yr}^{-1}$)			
Bay Segment	Period		Reduction %
	1980-1985	1999-2008	
HB	520	260	50
MTB	220	110	47
OTB		190 ¹	

Surface Chlorophyll-a (mgm^{-3})			
Bay Segment	Period		Reduction %
	1980-1985	1999-2008	
HB	35	10	72
MTB	15	5.5	64
OTB	13 ²	8.6 ^{1,2}	31

¹ 2000-2008

² EPCHC Station 40 (mid depth samples) located close to COT 40 (Figure 1).

Table 2. Estimated amount of total nitrogen (TN) required to sustain phytoplankton production rates in HB, MTB and OTB on an annual basis for the periods listed. Also shown is the fraction of phytoplankton production rates that potentially could be sustained by estimated TN loading supplied from external sources.

Bay Segment	Phytoplankton Production (gCm ⁻² yr ⁻¹)	TN Load Required to Sustain Production (mtTNyr ⁻¹)	External TN Load (mtTNyr ⁻¹)	Percent of Production that can be Sustained by External TN Load
HB Worst Case ¹	750	10700	5010	47
HB Current ²	410	5900	1780	30
MTB Worst Case ¹	630	24700	1480	6
MTB Current ²	350	16000	640	4
OTB Current ³	390	9300	560	6

¹ Phytoplankton production 1980-85; external TN loading 1976 (Zarbock et al. 1994).

² Phytoplankton production 1999-08; external TN loading 2003-07 (TBNMC 2009).

³ Phytoplankton production 2000-08; external TN loading 2003-07 (TBNMC 2009).

Table 3. Estimated current annual phytoplankton and seagrass carbon production rates in HB, MTB and OTB.

Bay Segment	Phytoplankton ¹	Seagrass+Epiphytes	Relative Production by Phytoplankton (%)
	mtCyr ⁻¹	mtCyr ⁻¹	
HB	33000	54 ²	99.8
MTB	83200	6700 ³	92
OTB	58800	5900 ³	90

¹ Areas deeper than 2m.

² Based on 2006-2008 seagrass coverage (BSG estimate); shoal grass, annual production rate 90gCm⁻² (adapted from Neely 2000).

³ Based on 2008 seagrass coverage (Kaufman pers. comm.): mixed species, estimated annual production rate 250gCm⁻².

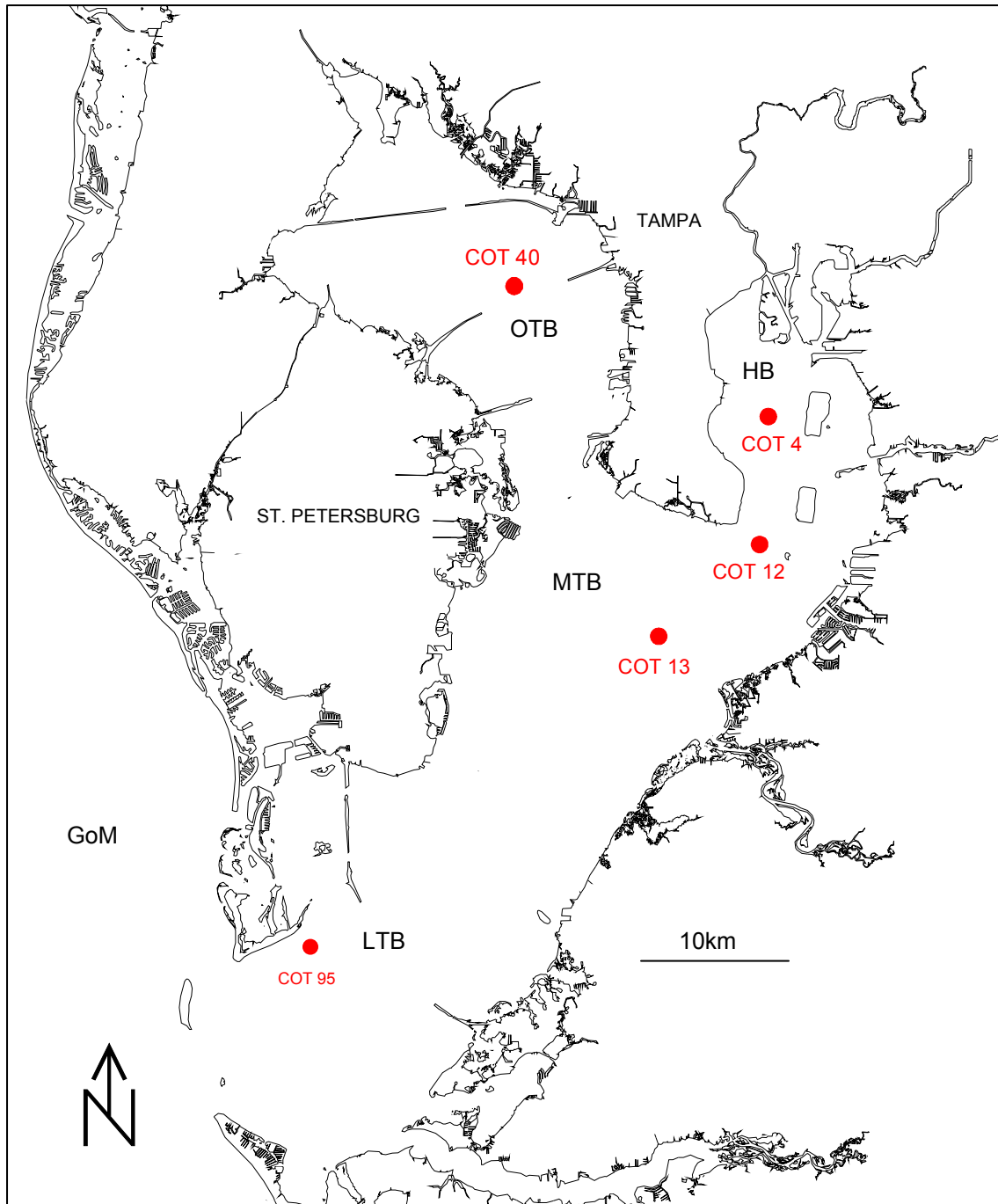


Figure 1. Phytoplankton monitoring locations in Tampa Bay maintained by the City of Tampa Bay Study Group. Phytoplankton production is measured at COT 4 (depth 3.8m), 12 (depth 5.4m), 13 (depth 6.4m) and 40 (depth 4.8m). Phytoplankton bioassay experiments are conducted on water collected at COT 4, 13, 40 and 95 (depth 8.6m).

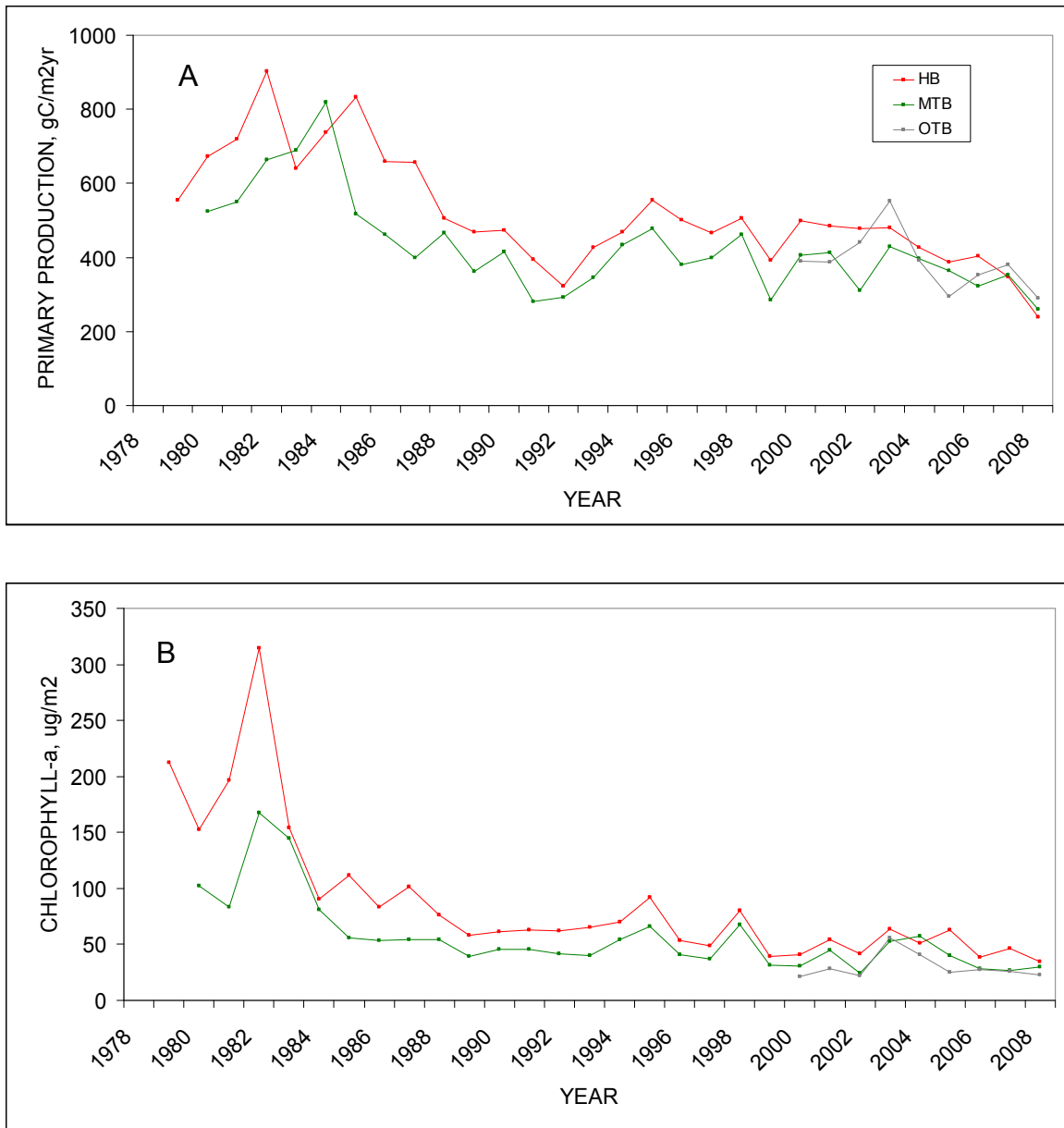


Figure 2. Long-term trends in water column phytoplankton production rates (A) and chlorophyll-a concentrations (B) in HB, MTB, and OTB.

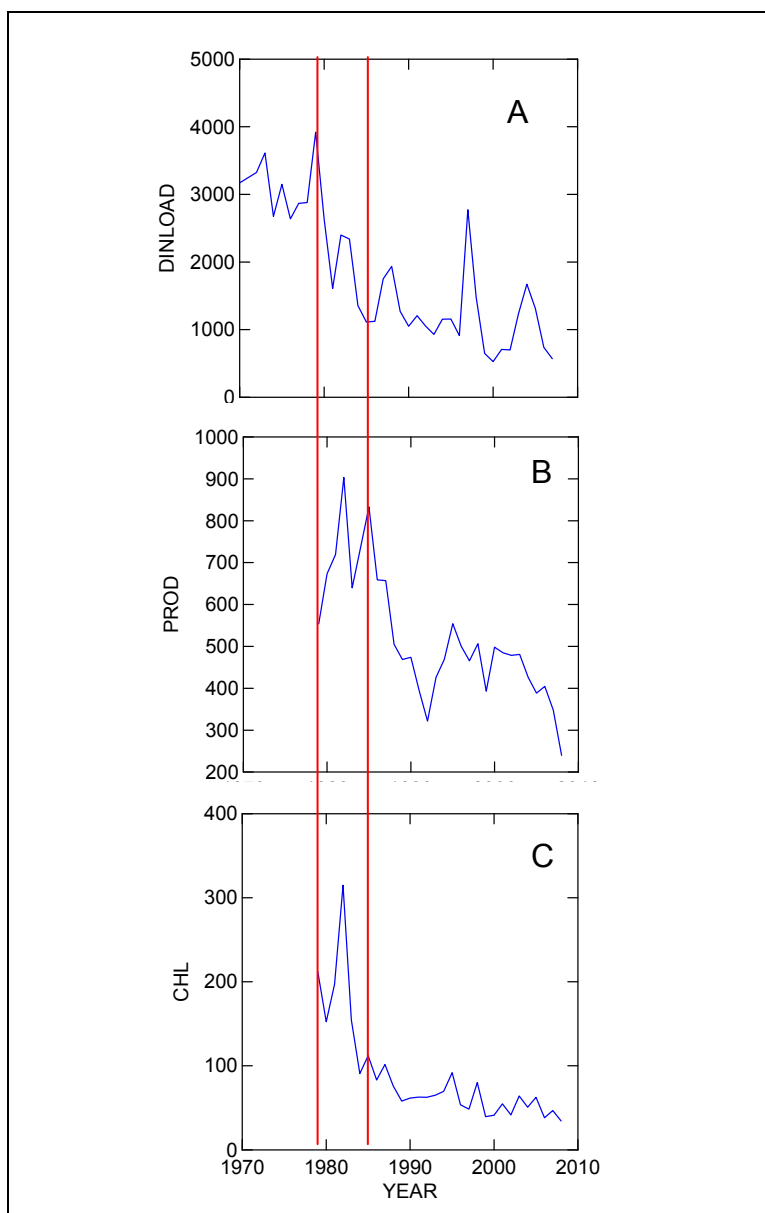


Figure 3. Long-term trends in (A) dissolved inorganic nitrogen loading (DIN) from external sources to HB (kgNday^{-1}), (B) HB phytoplankton production rates ($\text{gCm}^{-2}\text{yr}^{-1}$) and (C) HB chlorophyll-a concentrations (mgm^{-2}). The red lines denotes the period of major reductions in DIN loading.

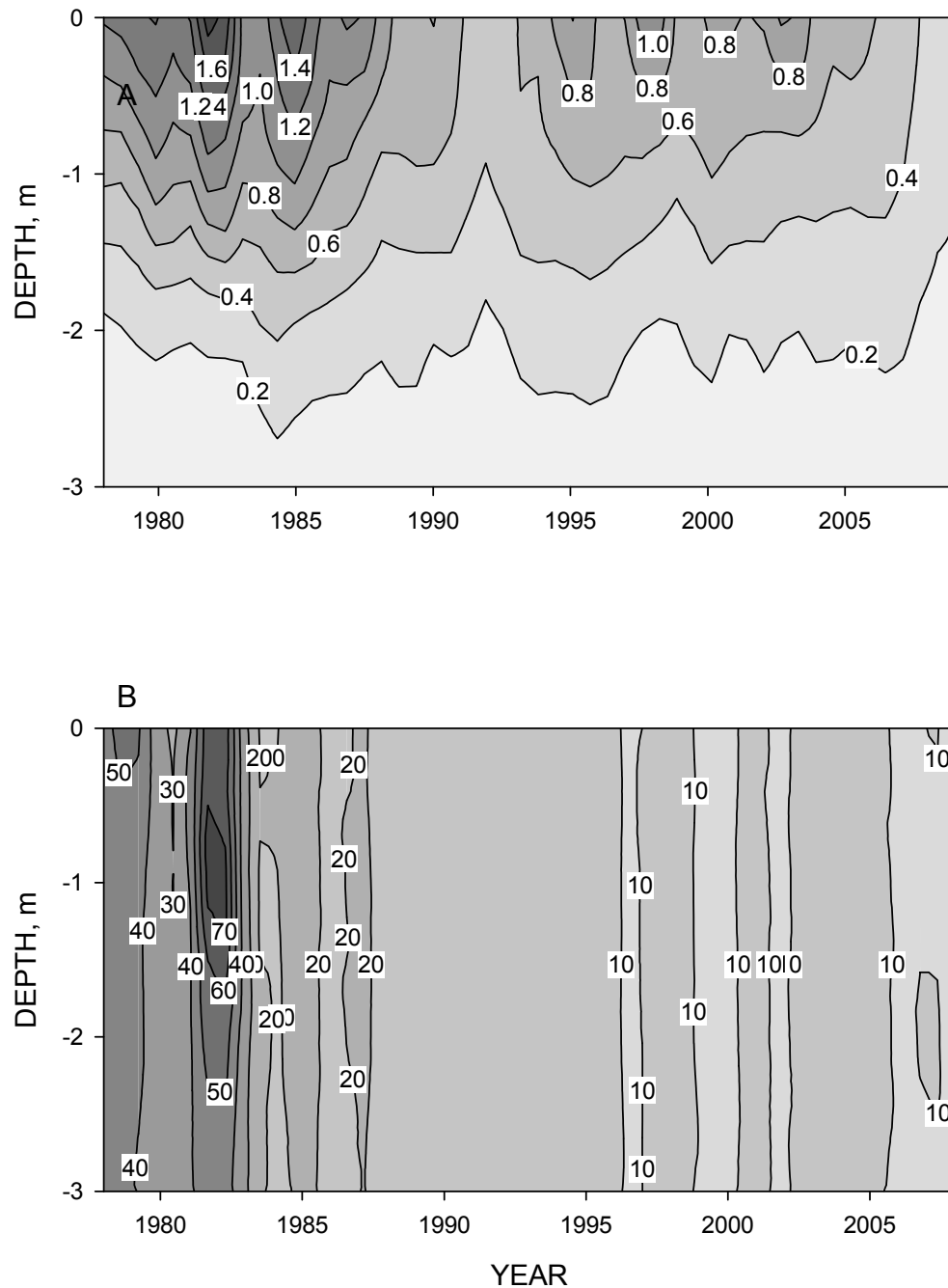


Figure 4. Long-term vertical distribution in (A) phytoplankton production rates ($\text{gCm}^{-3}\text{day}^{-1}$) and (B) chlorophyll-a concentrations (mgm^{-3}) in HB.

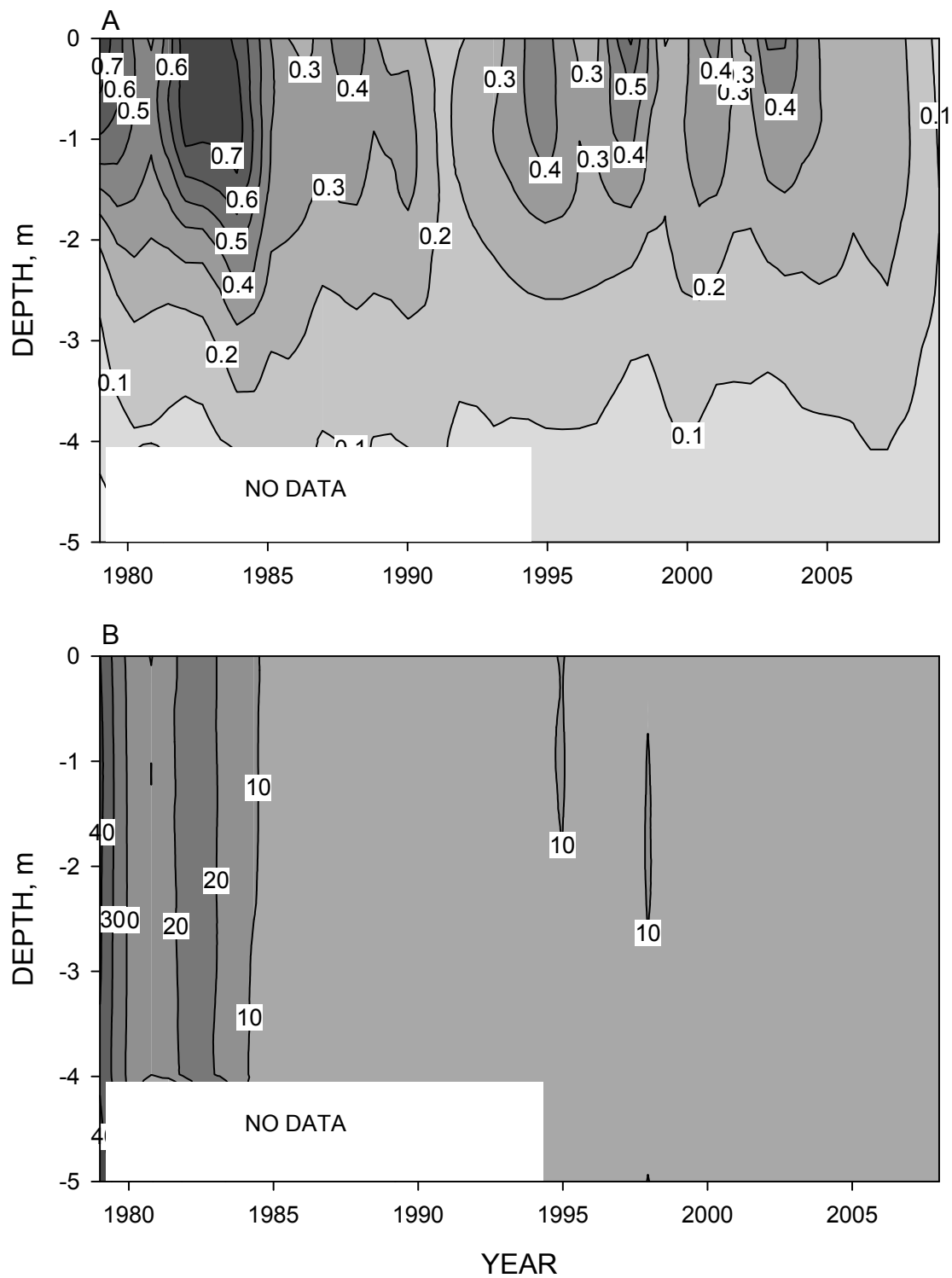


Figure 5. Long-term vertical distribution in (A) phytoplankton production rates ($\text{gCm}^{-3}\text{day}^{-1}$) and (B) chlorophyll-a concentrations (mgm^{-3}) in MTB.

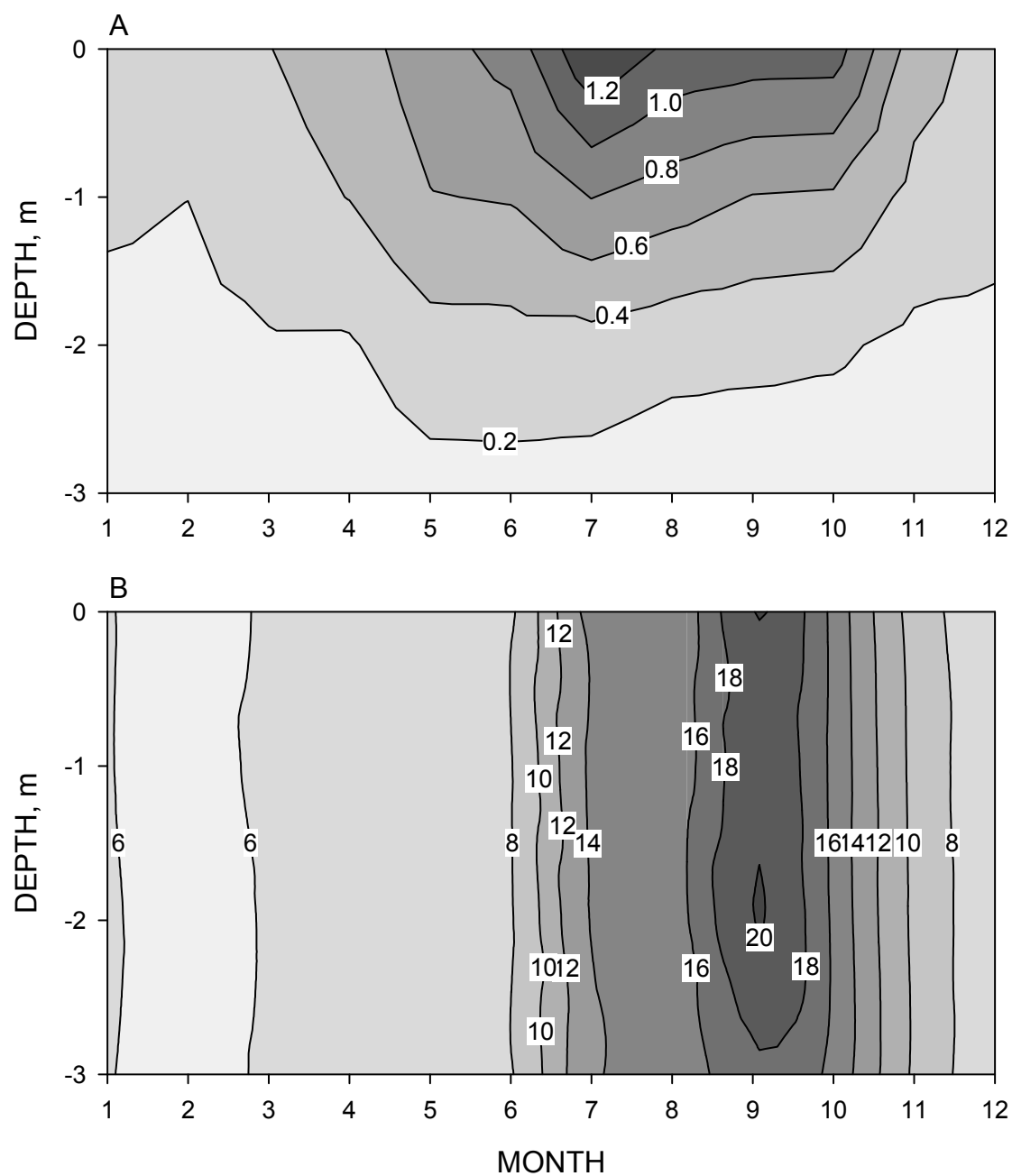


Figure 6. Seasonal vertical distribution for the period 1999-2008 in (A) phytoplankton production rates ($\text{gCm}^{-3}\text{day}^{-1}$) and (B) chlorophyll-a concentrations (mgm^{-3}) in HB.

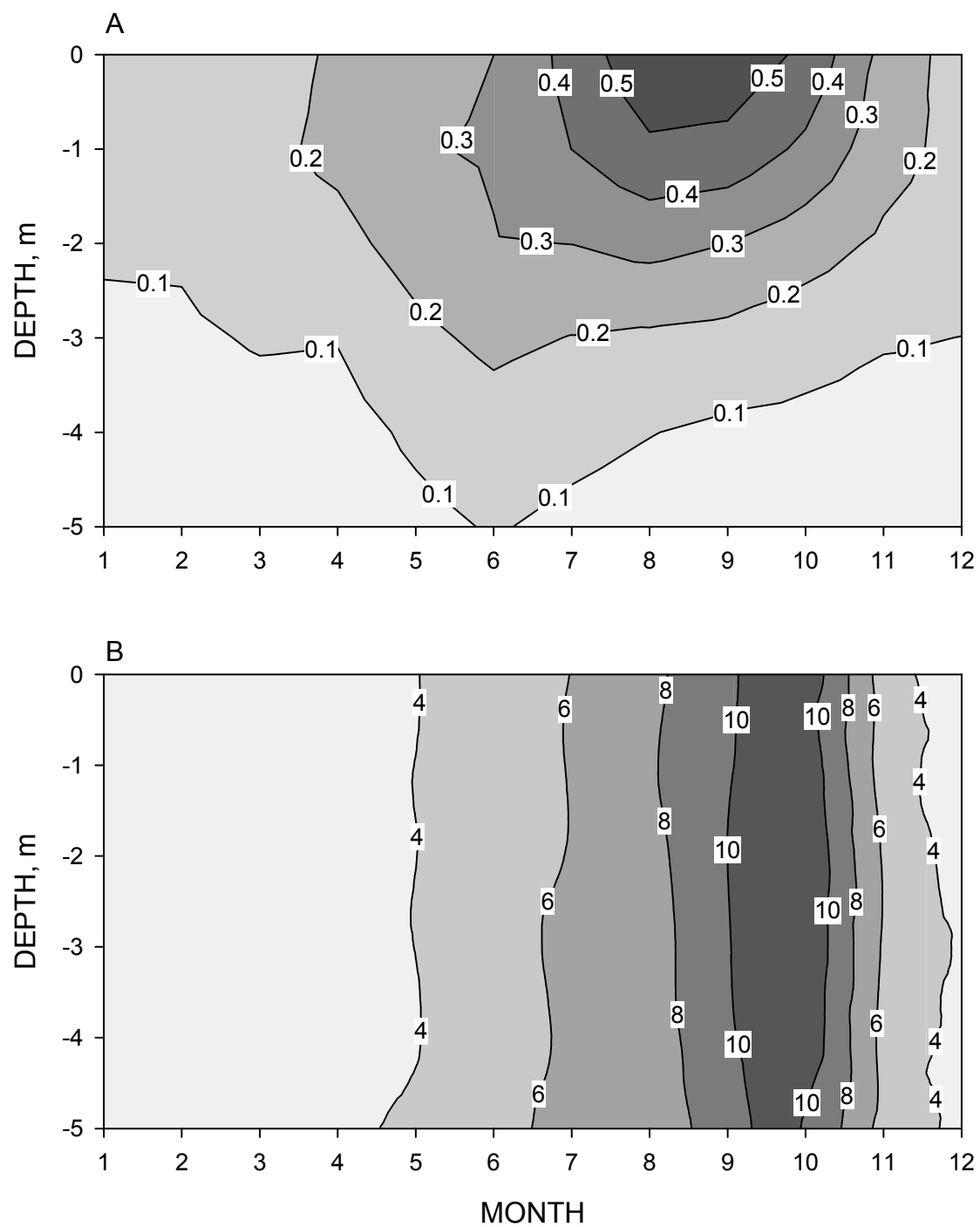


Figure 7. Seasonal vertical distribution for the period 1999-2008 in (A) phytoplankton production rates ($\text{gCm}^{-3}\text{day}^{-1}$) and (B) chlorophyll-a concentrations (mgm^{-3}) in MTB.

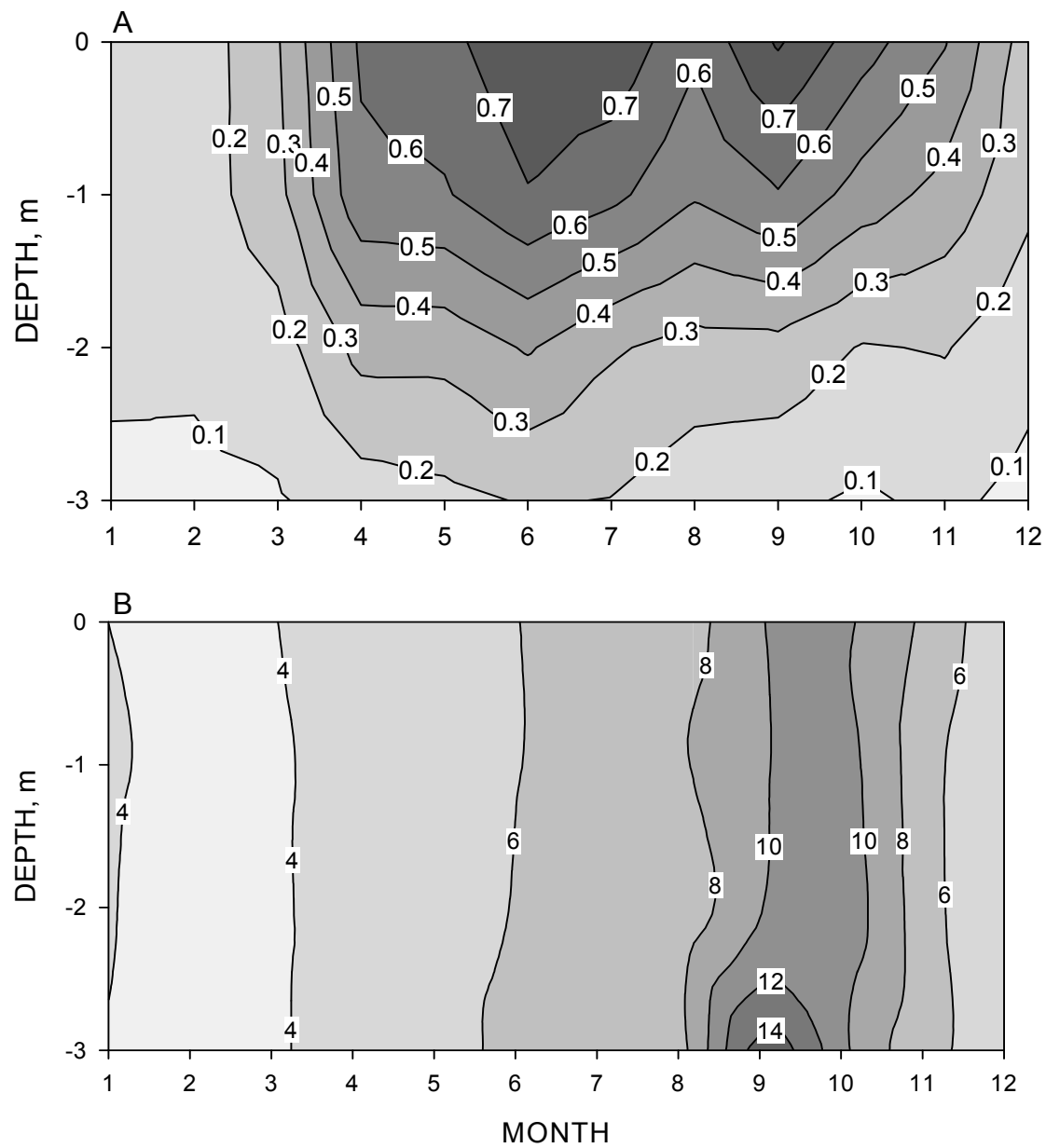


Figure 8. Seasonal vertical distribution for the period 2000-2008 in (A) phytoplankton production rates ($\text{gCm}^{-3}\text{day}^{-1}$) and (B) chlorophyll-a concentrations (mgm^{-3}) in OTB.

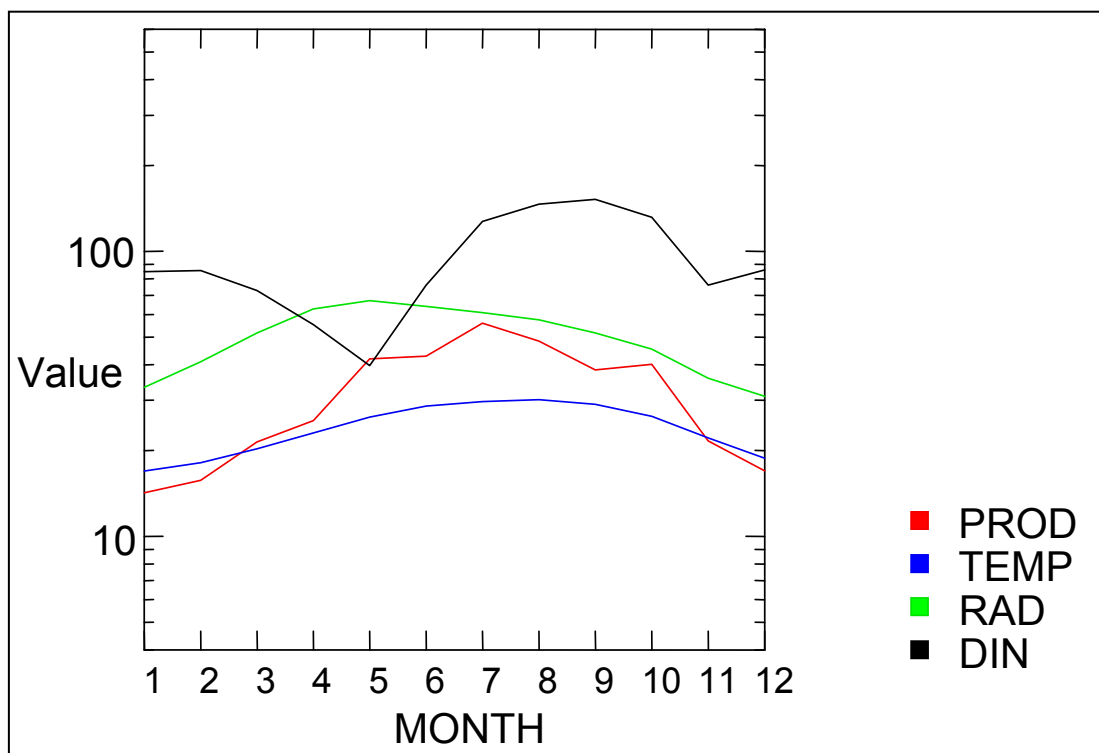


Figure 9. Seasonal averaged variations during the recent decade in HB in phytoplankton production, water temperature, solar radiation at Tampa and dissolved inorganic nitrogen loading (DIN) from external sources to HB. Values plotted are relative for illustration purposes.

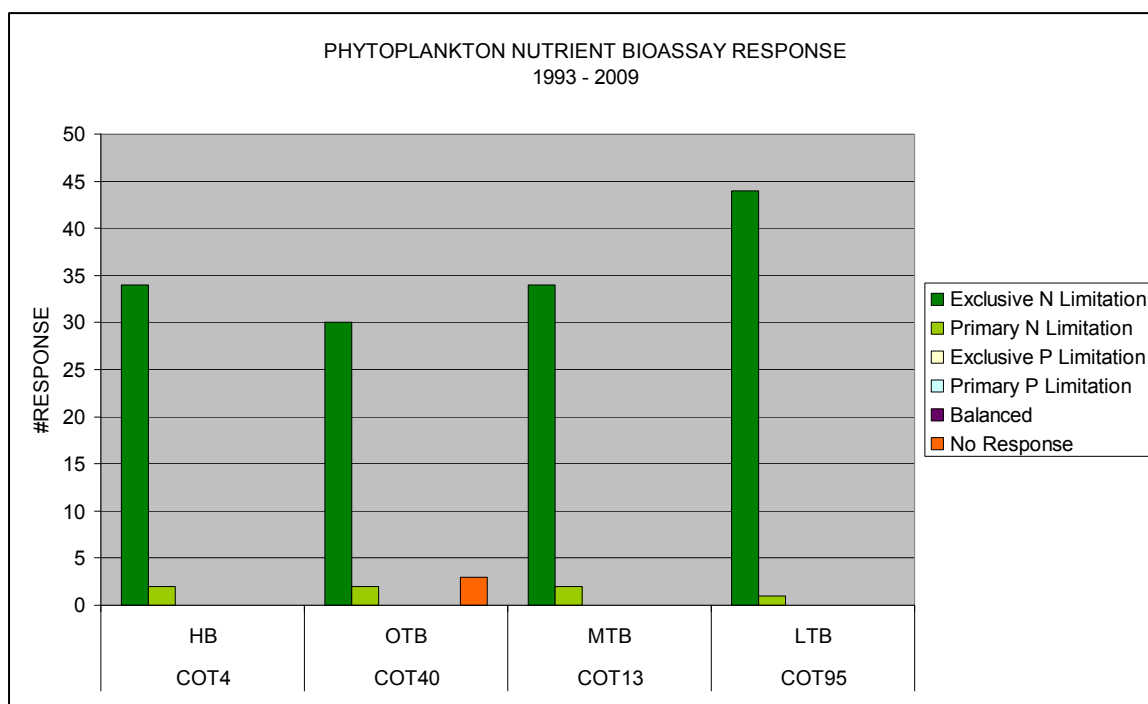


Figure 10. Results from natural phytoplankton nutrient enrichment bioassays in the four major subsections of Tampa Bay for the period 1993 – 2009 (see Figure 1 for station locations).

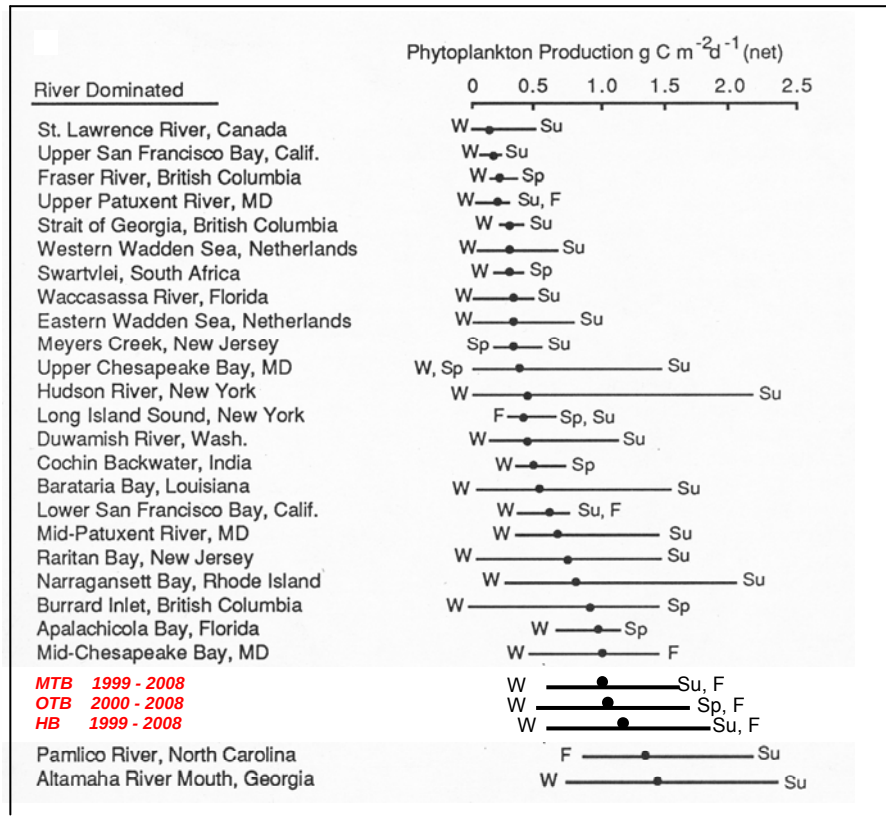


Figure 11. Average daily phytoplankton production rates (black dot) in HB, MTB, OTB and 25 other river-dominated estuarine systems. The horizontal bars indicate the annual range. The seasons when maximum and minimum rates occur are also indicated (W. winter; Sp. spring; Su. summer; F. fall). Figure modified from Boynton et al. (1982).

ABSTRACTS

Session 3: WATER QUALITY & PRIMARY PRODUCTION

PRESENTATIONS

Hydrodynamic and Water Quality Modeling of Hillsborough Bay, Florida. K. Hackett, R. Pribble, A.J. Janicki (Janicki Environmental, Inc.), and C. Tsokos (University of South Florida). Significant improvements in the water quality of Hillsborough Bay, Florida have been documented in the recent past as a result of the concerted efforts of the Nitrogen Management Consortium under the auspices of the Tampa Bay Estuary Program. With the projected increase in human population living in the watershed, it is important to identify the most cost-effective methods of controlling loadings. The implementation of these methods of controlling loadings should allow for the continued improvement of water quality in Hillsborough Bay and greater Tampa Bay. In order to better predict the results of future management actions, the Environmental Fluid Dynamics Code (EFDC) has been developed and employed to predict hydrodynamics and water quality in Hillsborough Bay and the estuarine portion of the three main tributaries to Hillsborough Bay: the lower Hillsborough River, the Tampa Bypass Canal/McKay Bay, and the lower Alafia River. A synopsis of hydrodynamic and water quality model calibration will be presented. [khackett@janickienviro.com]

Water Quality Monitoring in Tidal Tributaries. R. Boler (Environmental Protection Commission of Hillsborough County) and M.E. Flock (Pasco County Department of Environmental Management). Water quality is a fundamental component needed to assess tidal tributaries for environmental resources management decisions. The Environmental Protection Commission of Hillsborough County (EPC) has been measuring water quality in Tampa Bay and Hillsborough County's tributaries since 1974. All major tributaries are monitored monthly for a broad array of water quality variables. Other minor tributaries are monitored quarterly. We will present the locations, frequency and parameters for the tidal tributaries that EPC samples. We will seek to determine if there are spatial and temporal monitoring gaps around the Bay. We will seek to identify additional parameters for measurement that might improve our ability to understand and manage tidal tributaries. Additionally, Pinellas County Department of Environmental Management (DEM) has been measuring water quality in Tampa Bay and Pinellas County tributaries since 1991. Tributary water quality was monitored monthly from 1991-2002. In 2003, sampling frequency was reduced and a number of tributary stations were moved so volume discharge measurements could be added for TMDL purposes. We will present the locations, sampling frequency and water quality variables for DEM tributary stations from 1991-2002 and changes in the program since 2003. Water quality and load results will be summarized for the current set of stations. [rboler@epchc.org]

UNDERSTANDING THE UNDERWATER LIGHT FIELD AND ITS RELEVANCE TO SEAGRASS SUSTAINABILITY AND RESOURCE MANAGEMENT IN TAMPA BAY

C.J. Anastasiou, J. Kunzelman & J.O.R. Johansson

ABSTRACT

A major challenge in managing seagrass is setting appropriate minimum light targets, and in Tampa Bay, resource management agencies have adopted a bay-wide minimum light target of 20.5% of photosynthetically-available radiation (PAR). PAR is a broadband quantity and does not take into consideration the spectral properties of the light field, nor does it consider the specific absorption characteristics of the seagrass themselves. To address the need for better monitoring tools, a GIS-based modeling approach was used to couple high resolution bathymetry with the light attenuation coefficient to map the percent subsurface PAR(λ) reaching the bottom along the seagrass deep edge. Percent subsurface PAR(λ) was also calculated using direct measurements at the seagrass deep edge. PAR(λ) was grouped into blue, green, and red color bands whose wavelengths were based on the specific absorption characteristics of the seagrass species *Thalassia testudinum* (Banks ex. König) and *Halodule wrightii* (Asch.), both major species found in Tampa Bay. In all cases, the light field was depleted of blue light, accounting for as little as 5.3 % of the total PAR at the bottom. Based on annual average light attenuation coefficients, seagrass received 31.7 – 38.9 % of surface PAR and 13.6 – 18.1 % of surface blue light along the deep edge. During the rainy season, seagrass at the deep edge received as little as 2.51 % of surface blue light while still receiving 17.5 % of surface PAR, 19.1 % of surface green light, and 26.8 % of surface red light. Under the lowest light conditions measured in this study, seagrass were primarily dependent on red light and, to a lesser extent, on blue-green and yellow light.

INTRODUCTION

Seagrass are important primary producers in estuarine systems around the world and provide critical habitat and food for many commercially and recreationally important fish. Seagrass also play a key role in biogeochemical processes, sediment stability, as well as many other functions (Bortone 2000; Hemminga et al. 2000; Thayer et al. 1984). Because of the perceived importance of seagrass to maintaining a healthy estuary, resource management agencies have focused on restoring and protecting seagrass to the greatest extent possible. Light is the primary limiting factor for most seagrass ecosystems. For this reason, most management plans attempt to set minimum light targets along the deep edge, typically in terms of the photosynthetically available radiation (PAR).

It is important to think of PAR in terms of its flux of quanta as opposed to its energy state because once a quantum has been absorbed by a plant cell, its contribution to photosynthesis is the same regardless of its wavelength-specific energy (Kirk 1994; Mobley 1994). Of course, a given photon must first be absorbed by one of the wavelength-specific photosynthetic pigments. The usefulness of a given light field for photosynthesis is not simply a function of the total intensity of PAR, but how well the spectral distribution of PAR matches the absorption spectrum of a given aquatic macrophyte or phytoplankton (Kirk 1994).

Because PAR, by definition, is a broadband quantity, it does not take into account the spectral variability of light reaching the bottom. Spectral PAR (PAR(λ)) provides much more information on the shape of the incoming light field than simply measuring PAR. However, neither PAR nor PAR(λ) provide any indication that the photons available are actually being absorbed. Since seagrass are considered the “end users” of the incoming light field, it is important to express light quality in terms of the specific absorption characteristics of the light harvesting pigments found in

seagrass. Like all higher plants, seagrass are reliant mostly on light within the blue and red color bands, though the presence of accessory pigments can increase the operational window beyond the blue and red wavelengths (Figure 1).

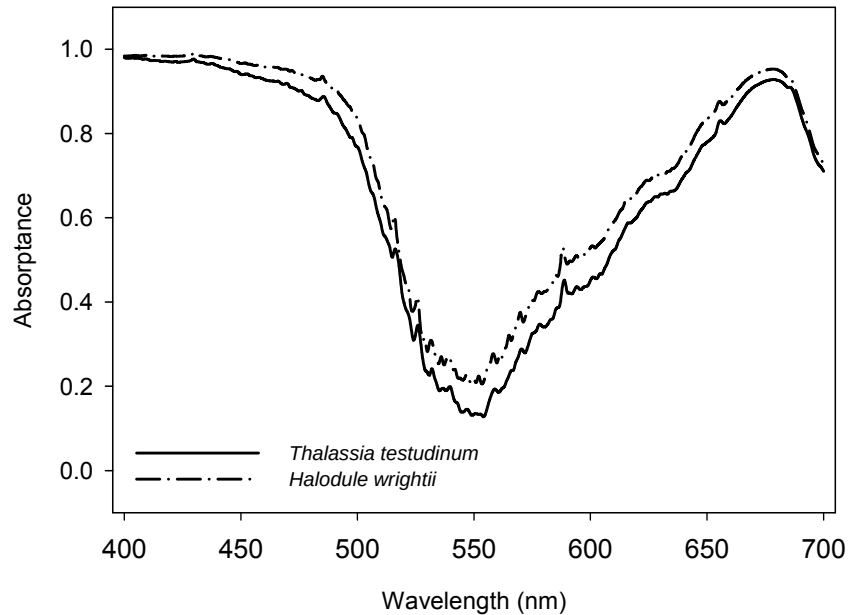


Figure 1. Absorbance spectra for two species of seagrass *Thalassia testudinum* (Banks ex. König) and *Halodule wrightii* (Asch.).

For resource managers, it is far too complicated to address light quality on a per nanometer basis. It is more practical both operationally and conceptually to combine wavelengths into broad color bands based on the absorption characteristics of the seagrass (Figure 2). In this study, three color regions or bands are defined (Table 1). These regions correspond to the absorption peaks associated with the chlorophylls and the carotenoids.

Table 1. Selected wavelength ranges of the color bands blue, green, and red and the pigments that are represented by each. Only the red algae, blue-green algae, and the cryptophytes contain billiproteins that allow them to harvest green light.

	Blue	Green	Red
Wavelength Range	400nm-490nm	490nm-600nm	640nm-690nm
Pigments	Chlorophyll/Carotenoid Protein Complex	Billiproteins	Chlorophylls
Found in Seagrass	Yes	No	Yes

MATERIALS AND METHODS

The Seagrass Management Area Concept—As part of the Tampa Bay Estuary Program's seagrass management plan re-evaluation, Tampa Bay was subdivided into 30 individual Seagrass Management Areas (SMA) (Figure 2) (EPCHC 2007). For this study, four SMAs were selected based on *a priori* knowledge of the optical properties and historical seagrass coverage. Within each SMA two fixed stations were established from which all *in-water* irradiance measurements were collected. For each SMA, except for Egmont Key, nearshore and offshore stations were established (Table 2).

Table 2. Fixed stations for each SMA where in-water irradiance measurements were collected. Meter marks correspond to the approximate distance from the shoreline. The seagrass *Halodule wrightii* disappeared from the offshore Wolf Branch and Kitchen sites mid-way through the study. Depth is relative to MSL.

SMA	Strata	Meter Mark	Depth (m)	Seagrass Species
Coffeeport Bayou	near	100	0.92	<i>Thalassia testudinum</i> , <i>Halodule wrightii</i>
	shore	900	1.18	<i>Thalassia testudinum</i>
	offshore			<i>Thalassia testudinum</i>
Wolf Branch	near	300	0.85	<i>Thalassia testudinum</i> , <i>Halodule wrightii</i>
	shore	1100	1.50	<i>Halodule wrightii</i> (disappeared)
	offshore			<i>Halodule wrightii</i> (disappeared)
Kitchen	near	600	0.88	<i>Halodule wrightii</i>
	shore	1300	1.16	<i>Halodule wrightii</i> (disappeared)
	offshore			
Egmont Key	near	100	1.59	<i>Thalassia testudinum</i>
	shore			

The nearshore sites were located within relatively healthy seagrass beds where light limitation was assumed not to be a factor. Originally the offshore sites were to be located at the deepest extent of seagrass growth, where light limitation was likely to be the primary limiting factor. In the Kitchen and Wolf Branch offshore areas, seagrass coverage was extremely sparse during the beginning of the study and had disappeared completely by the end of the study period. Because these SMAs were so shallow, the nearshore sites were actually located along the deep edge of the persistent seagrass bed.

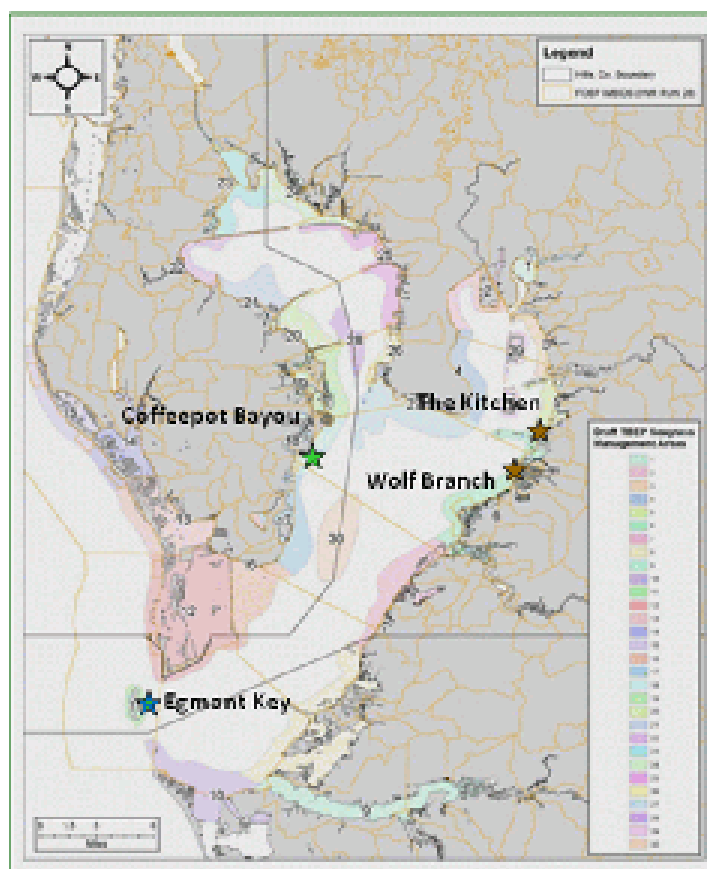


Figure 2. Seagrass Management Areas of Tampa Bay and selected areas for this study.

Measuring $E_d(\lambda)$, $PAR(\lambda)$, and $PAR-A$ —A planar irradiance cosine collector (Hobi Labs, Inc., Bellevue, WA) was mounted onto a PVC measuring rod. The cosine collector was then connected via a fiber optic cable to a portable spectrometer (HR2000, Ocean Optics, Dunedin, FL). A field laptop PC (Panasonic Toughbook, Panasonic Corporation, New York, NY), running the Ocean Optics program OOI Base32, provided the spectrometer's command and control. All data were stored on the PC. A planar irradiance cosine collector was chosen over a spherical sensor to remove any inherent bias caused by bottom reflectance. This makes it easier to compare different sites and provides a conservative estimate of $E_d(\lambda)$.

Mapping percent subsurface irradiance—A major limitation in understanding the seagrass light-depth relationship is accurate bathymetric data. Airborne laser bathymetry, also known as Light Detection and Ranging (LIDAR), can help overcome this limitation. LIDAR is a technique for measuring shallow waters using a pulsed laser beam from an airborne platform. This technology has been in use since the mid-1960s (Hickman and Hogg 1969). At that time, laser technology was brand new and used primarily for anti-submarine warfare by the U.S. Navy. Today airborne LIDAR is routinely used for hydrographic surveys and has evolved into an accurate operational technique.

In 2007, selected areas of Tampa Bay were mapped using NASA's Experimental Advanced Airborne Research LIDAR (EAARL).

Results and Discussion—Seagrass in Coffeepot Bayou are depth limited as evidenced by the relatively sharp shelf break at the offshore terminus. Most of the seagrass beds along the flat shelf behind this slope are classified on the 2006 seagrass map as continuous. Visually, these grasses

appear to be in good health and are persistent. Depths along the shelf break are between 1.25m – 1.50m MSL. The annual average $K_d(\text{PAR})$ of 1.51 m^{-1} was used to map the percent subsurface PAR for the Coffeepot Bayou SMA (Figure 3).

Extracting the percent subsurface PAR reaching the bottom along the mapped seagrass deep edge yielded an average of 31.7 ± 6.7 (Table 3). By contrast, the percent of blue light reaching the bottom was only 13.6% (Figure 3). The percent of green and red light reaching the bottom was 48.5% and 30.7%, respectively (Table 3).

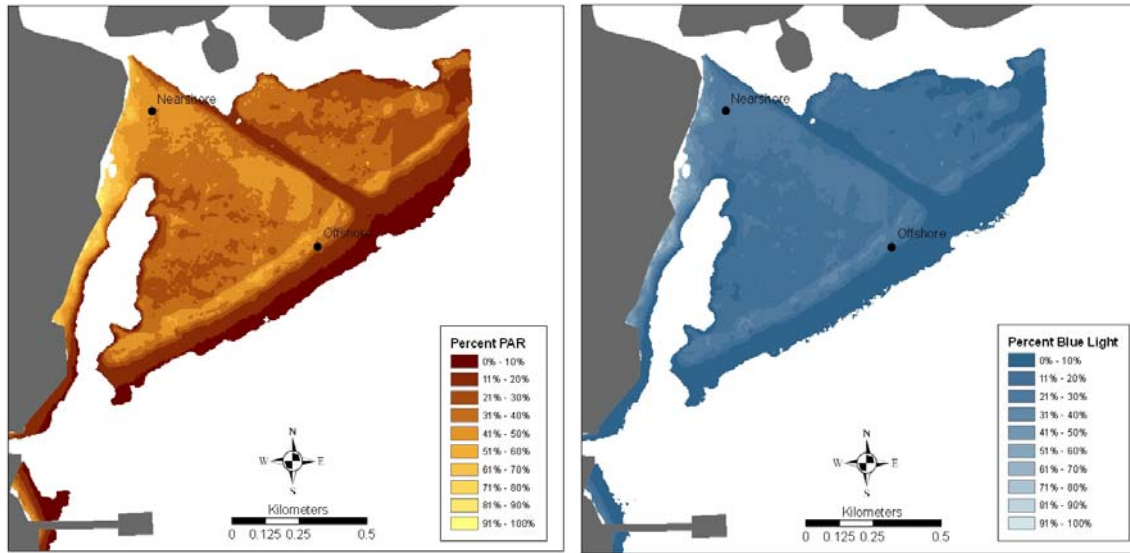


Figure 3. Percent subsurface PAR and blue light reaching the bottom for Coffeepot Bayou calculated using the annual average $K_d(\text{PAR})$ and LIDAR bathymetry.

Table 3. Total measured depth, percent subsurface blue light, and percent subsurface PAR relative to surface conditions along the mapped seagrass deep edge. These data were extracted using a sub-routine in GIS in which percent subsurface irradiance was collected for each pixel that fell along the seagrass deep edge.

	% Blue Light	% Green Light	% Red Light	%PAR	MSL Depth (m)
Coffeepot Bayou					
Mean $\pm$ standard deviation	13.6 ± 3.5	48.5 ± 5.1	30.7 ± 7.3	31.7 ± 6.7	1.21 ± 0.30
Median	13.8	48.9	31.5	32.7	1.14
Max	24.1	87.15	43.6	43.6	2.05
Min	4.5	26.26	13.6	15.5	0.82
Kitchen					
Mean $\pm$ standard deviation	17.3 ± 2.7	44.5 ± 3.2	40.7 ± 2.72	38.7 ± 3.25	0.87 ± 0.08
Median	17.4	44.8	40.3 51.8	38.9	0.86
Max	29.6	57.15	35.0	51.8	1.09
Min	11.0	36.26		30.3	0.60
Wolf Branch					
Mean $\pm$ standard deviation	18.1 ± 4.9	45.1 ± 5.6	40.7 ± 5.7	38.9 ± 5.8	0.97 ± 0.15
Median	18.0	45.4	40.9	39.1	0.95
Max	27.8	55.0	50.9	49.2	1.29
Min	18.0	34.4	29.8	28.1	0.72

Using the annual average $K_d(\text{PAR})$ of 0.985 m^{-1} for Wolf Branch yielded a percent subsurface PAR along the mapped seagrass deep edge of approximately 39% (Table 3). For Wolf Branch, the 30% - 40% range for subsurface PAR was located approximately 250m offshore (Figure 4) and corresponded to the mapped seagrass deep edge.

The modeled percent subsurface blue light at bottom was 17% along the mapped seagrass deep edge using a $K_d(\text{blue})$ of 1.78 m^{-1} (Figure 4). Near the offshore sample station, 1300m from the bank, is the approximate location of the minimum light target of 20.5% (Figure 4). Based on this target, there should be seagrass at this offshore station. While there were very sparse seagrass at the beginning of this study, after six months the few shoots that were there had disappeared. The percent subsurface blue light along this same 20.5% PAR line was 0% - 10%, further supporting the hypothesis that this is a blue-light limited environment. Using only percent subsurface PAR, without any information of the amount of blue light reaching the bottom, would lead to the conclusion that this area is meeting its minimum light requirement.

Spatial patterns in the Kitchen SMA were similar to Wolf Branch. Percent subsurface PAR at the deep edge was 38.9% and corresponded to an average annual $K_d(\text{PAR})$ of 2.07 m^{-1} . For the Kitchen, where there were persistent seagrass, the percent subsurface blue light had a range of 20% - 40% (Figure 5). Like Wolf Branch, at the beginning of the study, the offshore station had very sparse seagrass that disappeared during the rainy season. At this location, the percent subsurface blue light was approximately 10% (Figure 5) while the percent subsurface PAR was near 25%. Again, this further suggests that these areas are blue-light limited and that site suitability based solely on the minimum subsurface PAR could be misleading.

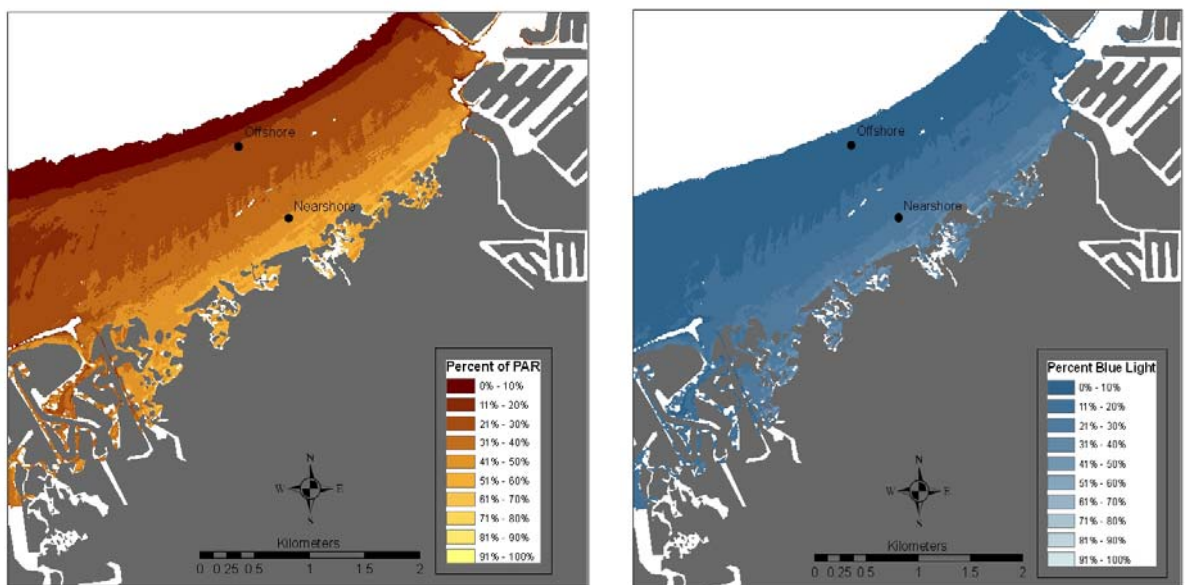


Figure 4. Percent subsurface PAR and blue light reaching the bottom for Wolf Branch calculated using the annual average $K_d(\text{PAR})$ and LIDAR bathymetry.

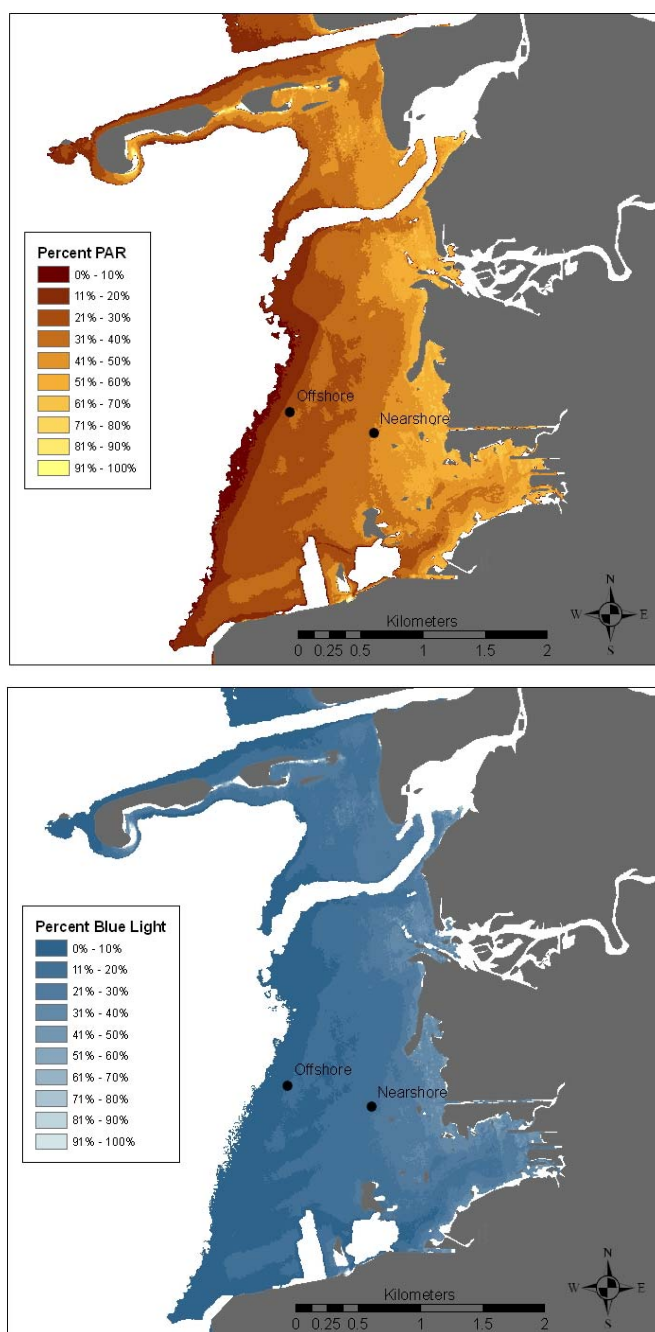


Figure 5. Percent subsurface PAR and blue light reaching the bottom for the Kitchen calculated using the annual average $K_d(\text{PAR})$ and LIDAR bathymetry.

CONCLUSIONS

In the context of the existing minimum light target of 20.5% PAR, the seagrass deep edge should not be light-limited. Based on the results presented here, this target may be too low and should be increased to 30% or higher. While this may seem like more than enough light, it is important to remember that much of the photosynthetically-usable blue light has been attenuated by the time the light reaches the bottom. Results presented here suggest that a target of 20% may actually be an appropriate blue light target. Because there were no significant differences in either $K_d(\text{blue})$

or $K_d(\text{PAR})$ for Coffeepot Bayou, Kitchen, and Wolf Branch, applying a bay-wide target may be appropriate and the need to develop Seagrass Management Area-specific targets unnecessary.

Having access to high resolution bathymetry, like that collected by NASA's EAARL system, greatly enhances the ability to model the spatial distribution of light. Unfortunately, such data are few and far between but none the less imperative for accurate model development. The GIS-based model presented here is an effective tool to quickly assess the status of the subsurface light field on an area-wide basis and does not and does not necessarily need such high resolution bathymetry. What is necessary is an accurate understanding of the spatial and temporal variability in $\text{Par}(\lambda)$ and $K_d(\lambda)$. The framework presented here is relatively straightforward and can quantify the spectral properties of the subsurface light field in a way that is cost effective and can be readily integrated into existing water quality monitoring programs. Because this system is designed to work in very shallow waters, it makes an ideal tool for seagrass applications. While understanding the spectral characteristics of the light field is a critical first step, it is not enough simply to know how much light there is at the bottom. It is equally important to understand the underlying causes of light attenuation and if these causes can be managed to facilitate improvements in water clarity, and ultimately to provide an environment suitable for seagrass recovery and growth.

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TRENDS IN NITROGEN LOADS TO TAMPA BAY FOR 1985 - 2007

P.A. Vaas & A. Janicki

ABSTRACT

Trend analyses of nitrogen loads to Tampa Bay are presented for the period 1985 to 2007 by bay segment and for the entire bay. Analyses of the total loads summed over all sources are presented. Results of the analyses of loads are presented for the Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay, and Lower Tampa Bay mainstem bay segments plus three additional segments: Boca Ciega Bay, Terra Ceia Bay, and the estuarine portion of the Manatee River. It is essential to detect trends in nitrogen loading to Tampa Bay to show progress toward meeting the standards set by the total maximum daily load (TMDL) limits as required by the USEPA and for Reasonable Assurance for the Florida DEP. Based on models developed for the Tampa Bay Estuary Program (TBEP), reductions in nutrient loads will reduce chlorophyll-*a* concentration and improve the light availability in the bay to allow the reestablishment of seagrasses, one of the primary goals of the TBEP. Total loads of nitrogen were estimated from nonpoint sources, domestic and industrial point sources, groundwater and springs, atmospheric deposition, and fertilizer handling losses. Methods for calculation of nitrogen loads are described in a report by Janicki Environmental, Inc. (2008). The trend analyses were conducted on the monthly load estimates using seasonal Kendall's Tau trend analysis, a set of nonparametric trend analysis techniques that make adjustments for seasonality and autocorrelation in the data.

INTRODUCTION

Trend analyses of nitrogen loads to Tampa Bay are presented for the period 1985 to 2007 by bay segment and for the entire bay. It is essential to detect trends in nitrogen loading to Tampa Bay to show progress toward meeting the standards set by the total maximum daily load (TMDL) limits as required by the USEPA and for Reasonable Assurance for the Florida DEP. Based on models developed for the Tampa Bay Estuary Program (TBEP), reductions in nutrient loads will reduce chlorophyll-*a* concentration and improve the light availability in the bay to allow the reestablishment of seagrasses, one of the primary goals of the TBEP.

According to census data, steady increases in population have occurred in the Tampa Bay area (Figure 1). Population has increased from less than a half-million people in the bay watershed in 1940 to over 2 million in 2000. In spite of these dramatic increases in population the water resources and aquatic ecosystems must be maintained in a healthy state.

APPROACH

Analyses of the total nitrogen loads summed over all sources are presented. Results are presented for the mainstem bay segments Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay, and Lower Tampa Bay. The loads to the LTB segment were calculated as the sum of the LTB plus segments Terra Ceia Bay, Boca Ciega bay and the estuarine portion of the Manatee River. A map showing the segments is presented in Figure 2.

Total loads of nitrogen (tons/month) were estimated from nonpoint sources, domestic and industrial point sources, groundwater and springs, atmospheric deposition, and fertilizer handling losses. Methods for calculation of nitrogen loads are described in a report by Janicki Environmental, Inc. (2008). A trend analysis of the total monthly freshwater inflow (millions of cubic meters per month, $10^6 \times \text{m}^3/\text{month}$) to Tampa Bay is also presented. The trend analyses were conducted on the monthly load estimates using seasonal Kendall's Tau trend analysis, a set of nonparametric trend analysis techniques that make adjustments for seasonality and

autocorrelation in the data. Programs used in the trend analysis are those developed by Reckhow et al. (1993).

The monthly data for total nitrogen (TN) and water loads are presented in the time series. For each time series, an ordinary least squares (OLS) regression analysis of load versus time was used to construct the OLS trend line that was superimposed over the time series of monthly data. Box and whisker plots, showing medians, means, ranges and quartiles were used to show the distribution of TN by year and by monthly seasons. The ends of the box whiskers show the minimum and maximum values, except where the whiskers end in an arrow, which indicates that the maximum values were outliers that went off the scale of the plot.

RESULTS

The time series of total freshwater inflow to Tampa Bay shows the typical seasonal patterns of inflow that occur in response to rainfall and runoff patterns common in Florida (Figures 3a-c). The monthly time series (Figure 3a) shows many extreme outliers in the summer months caused by hurricanes and tropical storms, followed by low inflow in the dry season that runs from November through May. The distributions of inflow by month (Figure 3b) show the typical seasonal pattern, with wet months including June through October, and the highest variation in September. The distributions of inflow by year (Figure 3c) show high variability across years, but no clear trend over the analyzed period of record, 1985 through 2007. The trend analyses indicated that the Kendall's Tau trend test statistics had to be adjusted for seasonality and autocorrelation. After adjusting the trend test statistics for seasonality and autocorrelation, no statistically significant trends were found at the 95% confidence level, either for the individual segments or in the time series of total inflow to the bay. The trend results are presented in Table 1.

The time series of total nitrogen loads to Tampa Bay (Figures 4-7) follow the same seasonal patterns seen in freshwater inflow, as the nitrogen loads are calculated as a function of concentration and inflow. The time series of TN loads to Old Tampa Bay (OTB) shows seasonal variability (Figure 4a), with highest loads during the wet season including June through October (Figure 4b), and variability across years due to climatic variation, but no overall trend (Figure 4c). The loads for OTB are generally lower than those of the mainstem segments that receive inflow from Hillsborough River, generally lower than 200 tons per month. The seasonal Kendall's Tau trend test indicated no significant trend in the OTB segment (Table 1).

The times series of monthly TN loads to Hillsborough Bay (HB) shows many peak loads greater than 500 tons per month during the summer months (Figure 5a). Seasonal variability and annual variation show similar patterns as described for OTB (Figures 5b-c). The seasonal Kendall's Tau trend test shows no significant trend in TN loads overall in Hillsborough Bay (Table 1).

The times series of monthly TN loads to Middle Tampa Bay (MTB) and Lower Tampa Bay (LTB) are shown in Figures 6 and 7, including seasonal and annual variation. Both segments show the same patterns as seen in the OTB and HB segments. The trend tests showed no significant overall trends in either the MTB or LTB segments (Table 1).

CONCLUSIONS

Statistical trend analyses presented here show that there is no significant trend in any of the time series of nitrogen loads in any segment in Tampa Bay over the period 1985 - 2007. According to census data, steady increases in population have occurred in the Tampa Bay area (Figure 1). Urbanization includes an increase in the area of impervious surfaces. The trend tests show no significant trends in the hydrological loads in any segment or in the time series of total freshwater

inflow summed bay-wide. In spite of increases in population and urban land use, the total nitrogen loads have remained constant over the long-term. The lack of any trends in the hydrological loads indicates that the lack of a significant trend in nitrogen loads can be attributed to effective nitrogen management programs in the Tampa Bay area.

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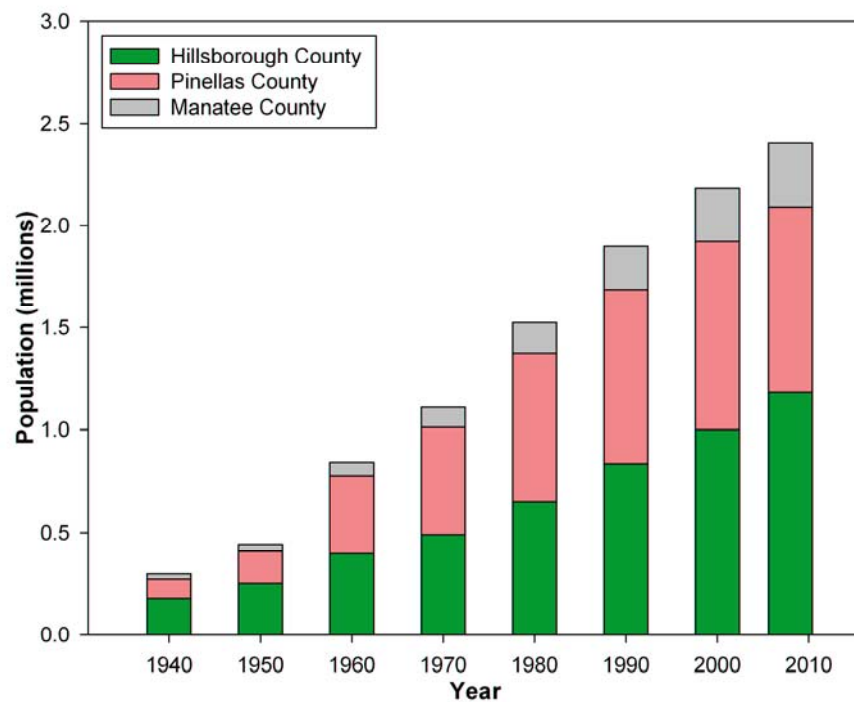


Figure 1. Census figures for population growth during 1940 – 2010 for the Tampa Bay watershed.
Source: U.S. Census.

Table 1. Results of trend analyses of Nitrogen and Hydrological Load.

Analysis period: 1985 – 2007		
Segment	Nitrogen Load	Hydrological Load
Old Tampa Bay	No trend	No trend
Hillsborough Bay	No trend	No trend
Middle Tampa Bay	No trend	No trend
Lower Tampa Bay	No trend	No trend
Bay-wide Total Load	--	No trend
Note: The 95% confidence level was used as the threshold for rejection of the null hypothesis of no trend.		

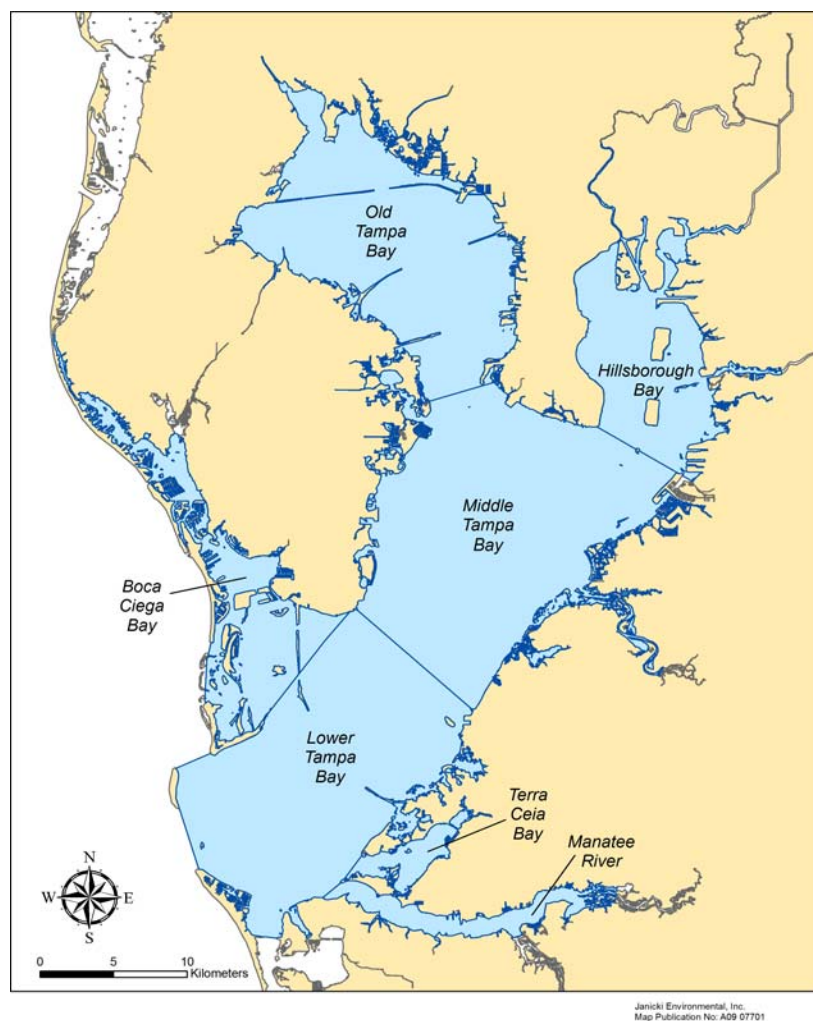
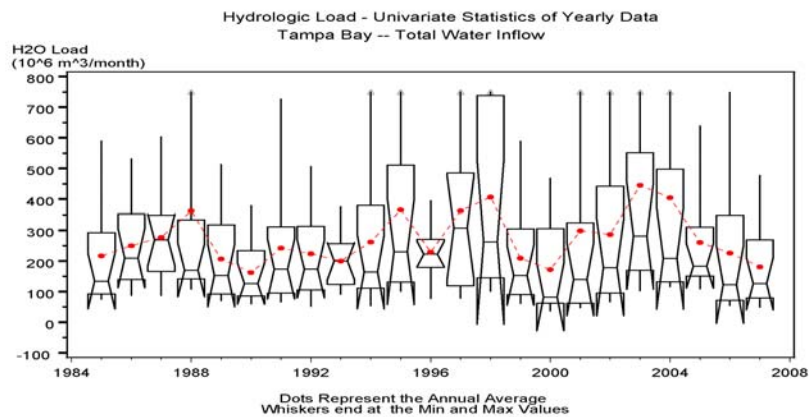
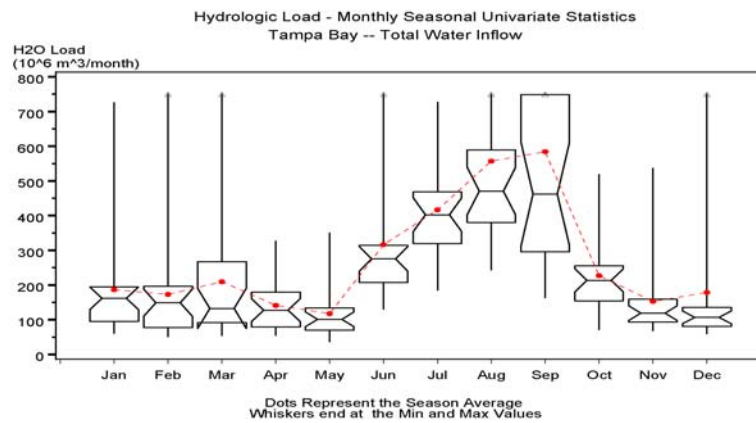
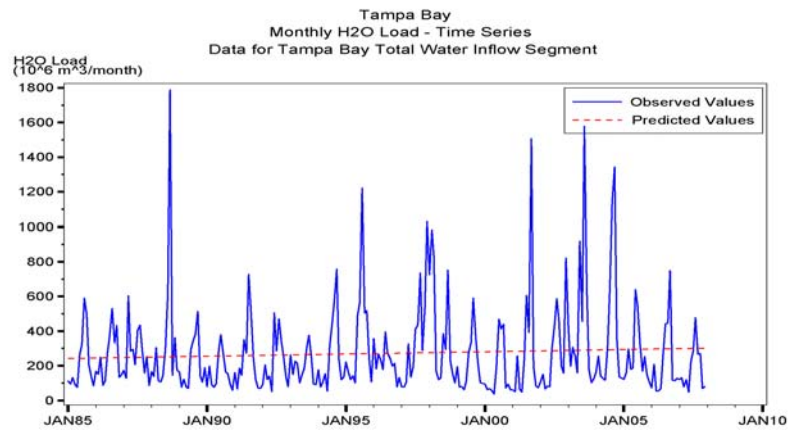
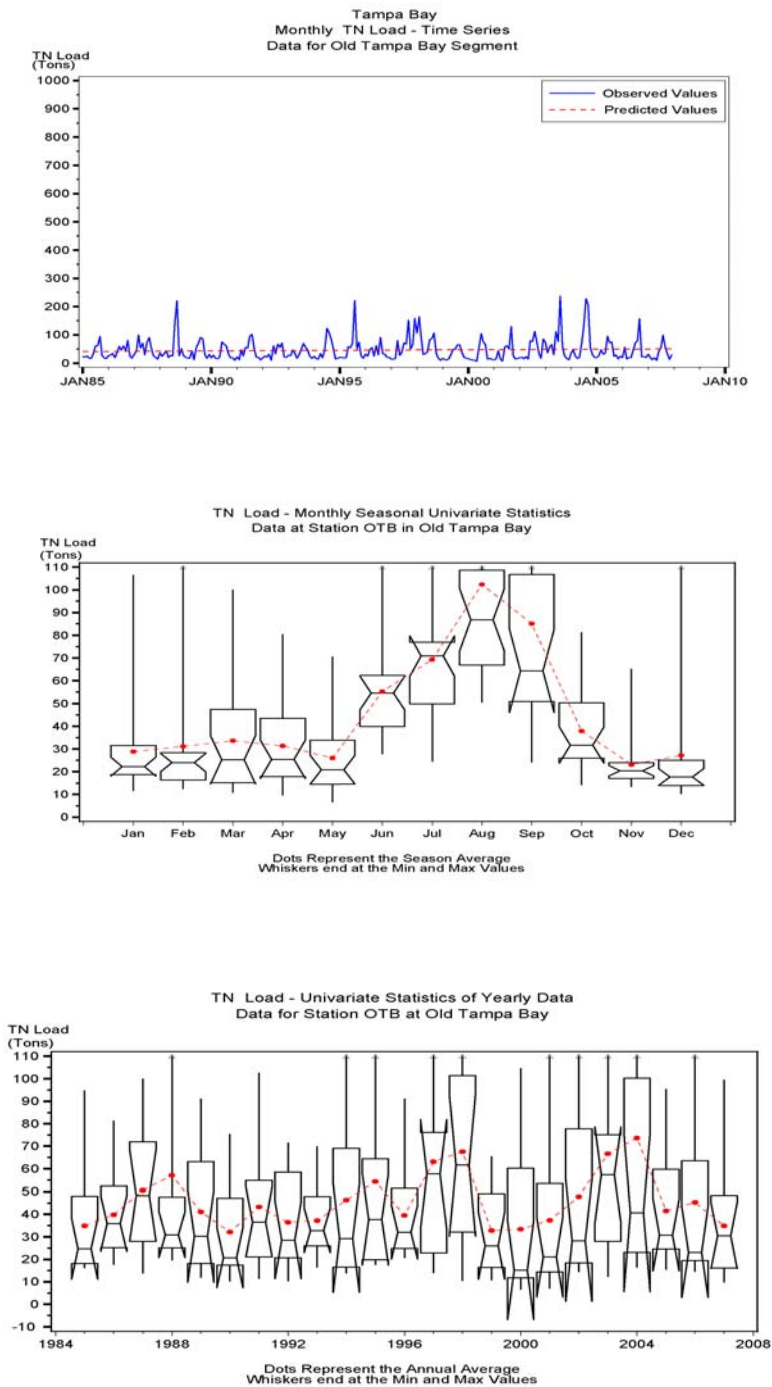


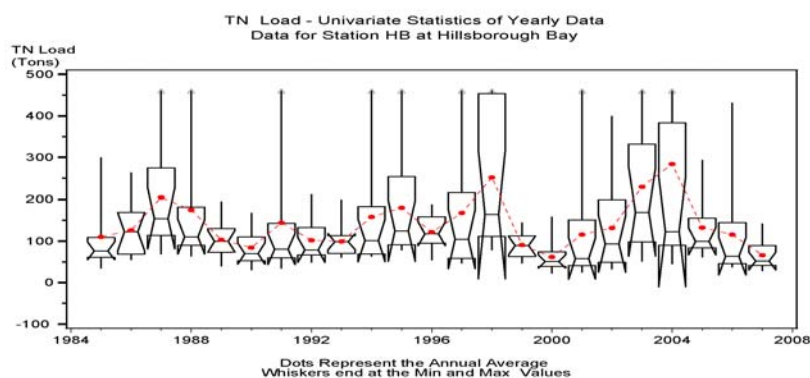
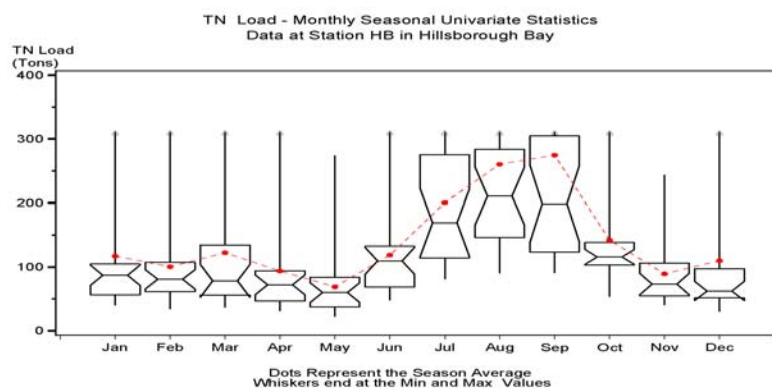
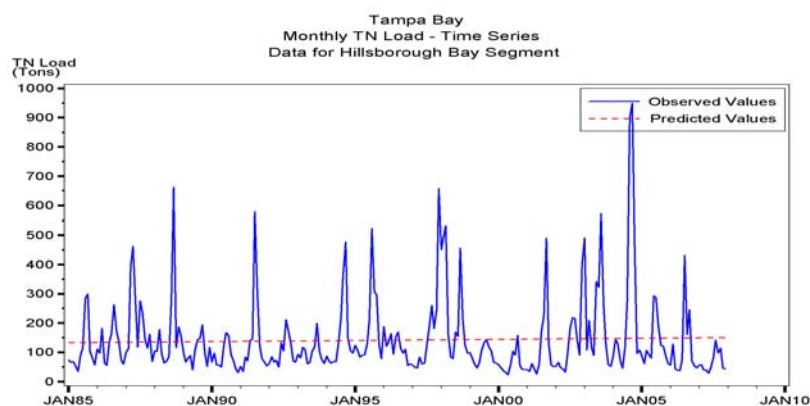
Figure 2. Map of Tampa Bay showing the segments used in an analysis of trends in TN load.



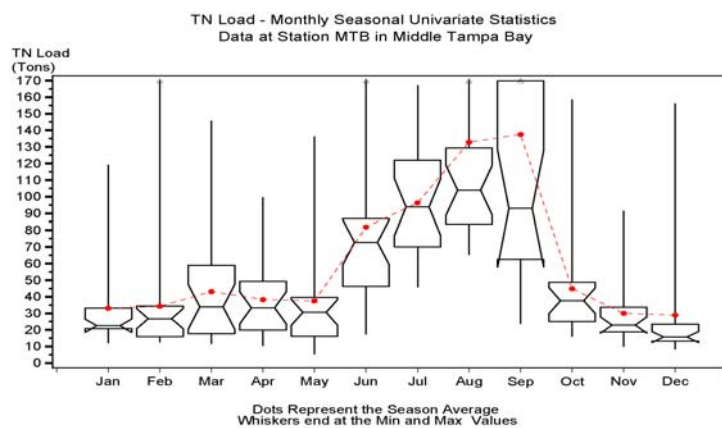
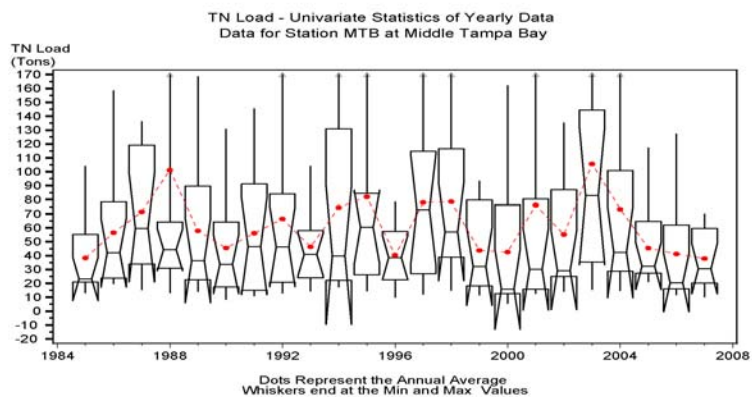
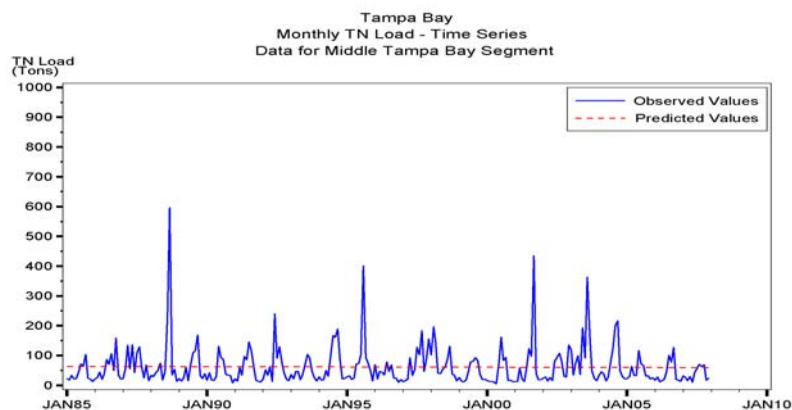
Figures 3a-c. a) Time series of monthly data for total freshwater inflow to Tampa Bay, with OLS regression line superimposed. b) Box plots summarizing distribution of flows for twelve monthly seasons, with time series of the average monthly flows superimposed. c) Box plots summarizing distribution of inflows by year, with time series of the annual averages superimposed.



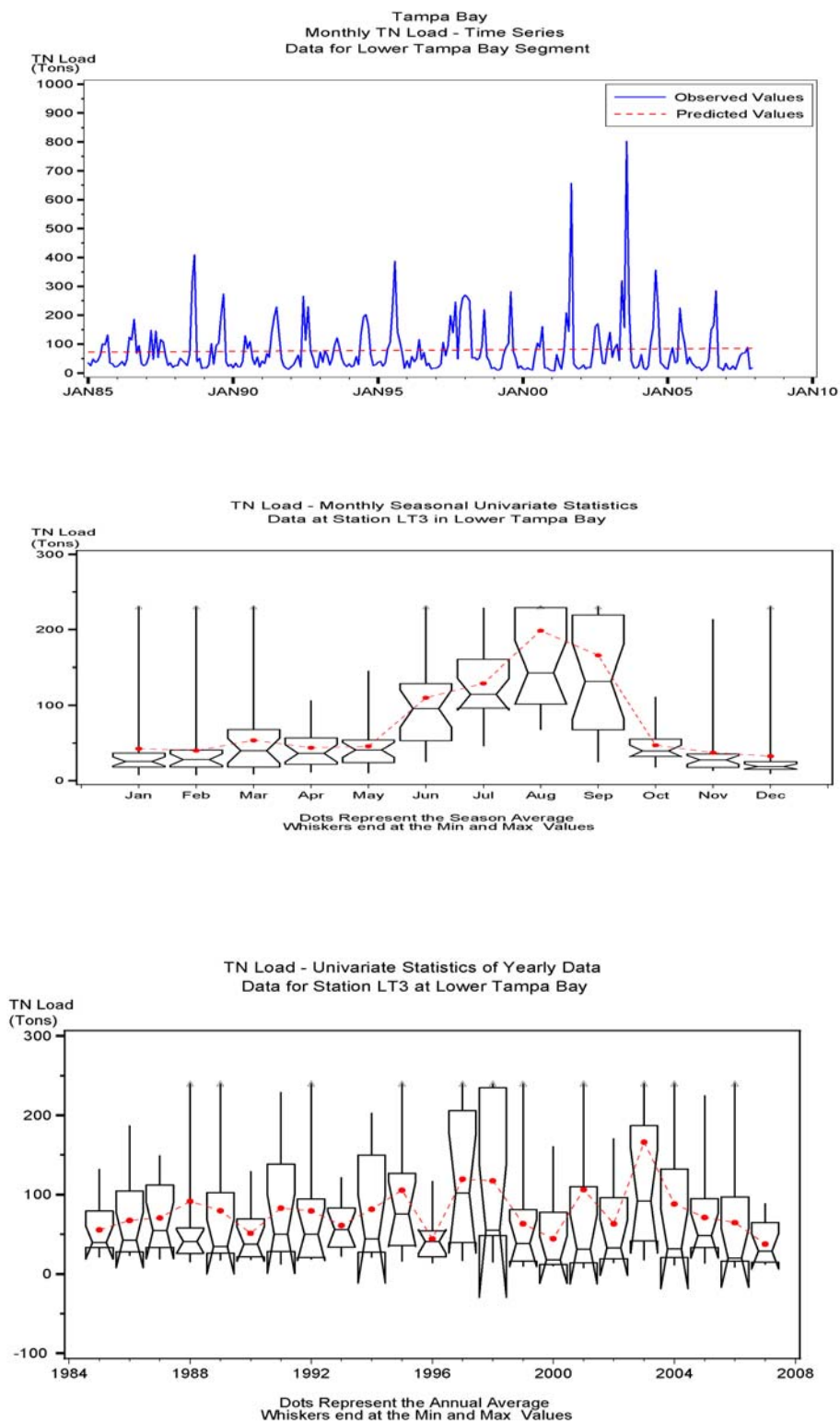
Figures 4a-c. a) Time series of monthly data for TN loads to Old Tampa Bay, with OLS regression line superimposed. b) Box plots summarizing distribution of TN load for twelve monthly seasons, with time series of the average monthly TN loads superimposed. c) Box plots summarizing distribution of TN load by year, with time series of the annual averages superimposed.



Figures 5a-c. a) Time series of monthly data for TN loads to Hillsborough Bay, with OLS regression line superimposed. b) Box plots summarizing distribution of TN load for twelve monthly seasons, with time series of the average monthly TN loads superimposed. c) Box plots summarizing distribution of TN load by year, with time series of the annual averages superimposed.



Figures 6a-c. a) Time series of monthly data for TN loads to Middle Tampa Bay, with OLS regression line superimposed. b) Box plots summarizing distribution of TN load for twelve monthly seasons, with time series of the average monthly TN loads superimposed. c) Box plots summarizing distribution of TN load by year, with time series of the annual averages superimposed.



Figures 7a-c. a) Time series of monthly data for TN loads to Lower Tampa Bay, with OLS regression line superimposed. b) Box plots summarizing distribution of TN load for twelve monthly seasons, with time series of the average monthly TN loads superimposed. c) Box plots summarizing distribution of TN load by year, with time series of the annual averages superimposed.

ABSTRACTS

Session 4: WATER QUALITY & ECOSYSTEM IMPLICATIONS

PRESENTATIONS

Chlorophyll-a Response in Tampa Bay to Varying Nitrogen and Hydrologic Loads: Implications to the Assessment of Water Quality Target Compliance. A.J. Janicki, R. Pribble, K. Hackett (Janicki Environmental, Inc.), H. Greening, and E.T. Sherwood (Tampa Bay Estuary Program). For more than 20 years, the management of water quality in Tampa Bay has focused on achieving nitrogen load reductions despite an ever-increasing population in the surrounding watershed and airshed. Specifically, the efforts of the Tampa Bay Nitrogen Management Consortium have been to maintain nitrogen loads to the four major bay segments at levels observed during the 1992-1994 period. The premise was that this “hold the line” strategy would maintain chlorophyll-a concentrations at levels conducive to the growth and reproduction of seagrasses in the bay.

Bay segment-specific chlorophyll-a targets were established in 1996 and proposed as appropriate indicators for the Reasonable Assurance accepted by the Florida Department of Environmental Protection in 2002. With few exceptions that were typically tied to anomalous meteorological conditions, these chlorophyll-a targets have been met since 1996. Over this same time period, the Consortium members identified and implemented a series of action plans that have cumulatively met the desired preclusion of 17 tons/year TN loads to the bay.

Over the past two years, the Consortium has been working with FDEP and the U.S. Environmental Protection Agency to update the 2007 Reasonable Assurance plan. There are two critical elements in this plan. First is the need to establish nitrogen load allocations for the entities that contribute to the nitrogen loading to the Bay. This is to ensure that the nitrogen loading targets are being managed at desired levels, i.e., the 1992-1994 average annual loads. Second is the framework for assessing compliance in both water quality (i.e., chlorophyll-a concentrations) and average annual TN loads over a 5-year period.

Examination of recent updates of TN loads to the bay showed that the TN load targets were exceeded in several years from 2003-2007. The increased loads were shown to have resulted primarily from higher than average hydrologic loads. Despite the increase in loads, chlorophyll-a targets were met in all four bay segments for the first time during both 2006 and 2007. Therefore, there was a “disconnect” between the management targets for the bay as the water quality targets were being met despite the elevated TN loads.

Further examination of the TN loading-chlorophyll-a relationship showed that there is an apparently significant influence of varying residence time due to varying hydrologic loads on this relationship. The observed response in chlorophyll depends in part on the amount of nitrogen entering the bay as well as the hydrologic inputs. Additional analysis of these loads shows that the amount of nitrogen load per unit of hydrologic load has been declining in all four major bay segments, which is in agreement with the observed bay water quality. We propose that this “nitrogen delivery rate” (TN load:hydrologic load ratio) is a defensible method for tracking compliance with the desired TN loads to Tampa Bay. [ajanicki@janickienviro.com]

Water Quality and Biology in Tampa Bay During Six Years of Desalination Facility Operation. K.L. Maki-Jenkins, R. Woithe, A. Willis (PBS&J), R. McConnell (Tampa Bay Water), M. Wessel, K. Hackett, and A.J. Janicki (Janicki Environmental, Inc.). The Tampa Bay Seawater Desalination Facility (Facility) has operated intermittently during its first six years and provides an opportunity to evaluate potential effects of desalination on water quality and biology under different operational conditions. Up to 44 mgd of the power plant’s 1.4 billion gallons per day of cooling water is withdrawn to produce 25 mgd of drinking water. The remaining 19 mgd of brine concentrate is mixed with cooling water and subsequently discharged. Models completed for permitting the Facility predicted minor salinity changes in the bay due to concentrate discharges. A hydrobiological monitoring program was implemented in 2002 to test model predictions and ensure that Facility operations did not adversely affect Tampa Bay. Continuous records of salinity from the intake and discharge canals were used to evaluate the effects of increasing production. Observed salinity differences at the discharge canal were less than predicted by the model, and comprised

five percent of the natural salinity variation in the vicinity of the Facility during the monitoring period. Data collected in four biological monitoring areas did not indicate any significant spatial or temporal changes in salinity or other water quality parameters. Dominant taxa, diversity, and community structure of benthic macroinvertebrate assemblages varied spatially, but variables not related to discharge from the facility explained the spatial heterogeneity observed. Seasonal and operational period patterns in fish community diversity in the vicinity of the facility were similar to patterns occurring elsewhere in Tampa Bay. Monitoring remains ongoing. [KLMJenkins@pbsj.com]

Management of Piney Point Phosphate Facility Closure and Effects on Water Quality in Bishop Harbor and Adjacent Regions of Tampa Bay. R. Pribble, A.J. Janicki (Janicki Environmental, Inc.), and R. Brown (Manatee County). FDEP assumed responsibility for the environmental security of the Piney Point phosphate processing facility in Manatee County near Bishop Harbor in 2001, including the untreated wastewater stored at the facility. Manatee County began collecting water quality data in October 2001 in Bishop Harbor. Unusually high rainfall in December 2002 caused concern that one of the wastewater containment berms might fail, resulting in a catastrophic release of highly-acidic and nutrient-laden untreated wastewater into Bishop Harbor. Treated wastewater was discharged to the harbor to alleviate this possibility. FDEP requested an assessment of the potential water quality responses to the discharge of treated wastewater into Bishop Harbor. Because of the need for timely technical guidance on the most appropriate discharge strategy, Janicki Environmental provided this assessment through a three-step approach, with each additional step more complex. These steps included development of empirical loading limits obtained by using a statistical model relating ammonia loads to chlorophyll, use of a two-dimensional water quality model to evaluate the findings of the empirical model, and use of a three-dimensional water quality model linked to the three-dimensional hydrodynamic model developed by USF for the bay and harbor. Discussion of the methodologies used to set loading limits using these steps is followed by examination of the resulting loadings to and water quality in Bishop Harbor. [rpribble@janickienviro.com]

Rapid Assessments of Lakes and River Stream Reaches in Hillsborough County, Florida. D. Eilers, J. Griffin (University of South Florida), J. McGee, and D. Glicksberg (Hillsborough County). Hillsborough County and the University of South Florida have over ten years of experience in the use of rapid lake assessments methods and have completed assessments and re-assessment on over 100 lakes in the County between 1998 and 2009. The objective of these assessments is to catalog the general health of small to medium size lakes by taking a snapshot of the lake's morphology, vegetation and water chemistry through an intense, one-day data collection effort. This method is now being adapted for rivers and streams.

In 2009 a large number of river reaches in the Hillsborough and Alafia Rivers will be assessed. These assessments will assist the University and Hillsborough County to better understand the general health of the resource and will allow estimates of plant species diversity, river volumes and general morphology and water chemistry. All these data will be made available for public view on the *Hillsborough County and City of Tampa Water Atlas* (www.hillsborough.wateratlas.org).

This report discusses changes to the assessment methodology and the expansion of the program to include stream reach assessments. The field methods that will be discussed include: (1) bathymetry to determine bottom contour; (2) bathymetry and the use of side-looking sonar imaging to determine submerged vegetation; (3) determination of percent volume infestation (PVI) and percent area coverage (PAC); (4) identification and mapping of invasive and non-native as well as native aquatic plants; (5) semi-quantitative assessment of submerged vegetation biomass and the parallel determination of nutrients held in vegetation; (5) physical profiling with the use of a multi-probe; (6) water quality sampling; and (7) the estimate of sediment volume. The general findings of this first year of stream assessments are also discussed. [eilers@arch.usf.edu]

ABSTRACT
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POSTER

Near-Shore Water Quality and Seagrass Depth Limits in Upper Tampa Bay, Florida. J.J. Pacowta, J.O.R. Johansson, W.M. Avery, and K.B. Hennenfent (City of Tampa, Bay Studies Group). Periodic setbacks occur in Tampa Bay seagrass coverage following periods of prolonged rainfall. The setbacks are not unexpected because the increased rainfall causes high tributary discharges of dissolved and particulate matter that affect the light climate of bay waters. However, several near-shore and shallow *Halodule wrightii* meadows in the upper areas of the bay, many bordering mangrove and salt marshes, have been stagnant or have shown very limited expansion for a decade or longer; a time which has included both dry and wet periods.

Light availability at the deep edge of these near-shore *H. wrightii* meadows, estimated from an optical model, appears to average about 50 to 60% of surface incident (I_o) photosynthetically available radiation (PAR). This light level would appear sufficient for the meadows to grow and expand into deeper waters. In contrast, deep edges of *H. wrightii* meadows that are temporarily established during extended dry periods, which are located near the offshore edge of the estuarine shelf, receive lower average light levels of about 30 to 40% I_o .

An ongoing study of shallow water quality in southeastern Hillsborough Bay shows that CDOM (colored dissolved organic matter) absorbance is consistently higher in waters above the near-shore seagrass bed than in waters above the offshore meadows. Chlorophyll and turbidity generally show increasing trends with distance from shore. Results from the study suggest that reductions of high energy blue light from relatively high CDOM absorbance, in addition to losses caused by phytoplankton, other particulate matter, and epiphytes may limit the near-shore meadows to the shallow depths they currently inhabit.
[john.pacowta@ci.tampa.fl.us]

LONGSHORE BARS ON THE SHORELINE OF TAMPA BAY AT MARIPOSA KEY

C.J. Hearn

ABSTRACT

This paper concerns a study of longshore bars on the coastlines of Tampa Bay. We have studied these bars in detail along the coast of Mariposa Key (Terra Ceia Aquatic Preserve). Modeling was performed of the nearshore processes with special regard to seagrass habitat. Together with historical data and aerial photographs, this has resolved the long term (decadal) migration of the bars as well as their alongshore wavy structure and its connection to the short term (sub-annual) oscillation of the bar structure due to alongshore current-wave instabilities. The long term shoreline budget of sedimentary material emphasizes the importance of anthropogenic influences on the structure of these longshore bars. The results are of general applicability to coastal studies in low-energy, strongly tidal, embayments which support substantial commercial shipping and recreational boating and are heavily urbanized.

INTRODUCTION

This is a report on the nearshore dynamics of Tampa Bay in the Mariposa Key region of the Terra Ceia Coast (Figure 1). Further details are given in Hearn (2010). The study area is just south of Bishop Harbor on the coast traditionally called Mariposa Key (Figure 1). The coastal terrain consists of karst ponds and mangroves. South of Mariposa Key is a series of low lying wetlands with well-defined coastal channels. To the north of Mariposa Key is the entrance channel to Bishop Harbor, beyond which is the entrance to Redfish Creek. Mariposa Key, in common with much of Tampa Bay, has a nearshore plateau a few hundred meters in width, with inter-tidal and sub-tidal longshore bars running roughly parallel to the coast. Such structures may be important to the stability of suitable habitats for sea grass meadows (Lewis 2005). Our study of the longshore bars at Mariposa is based on existing aerial photography, LIDAR data, boat-based sonar surveys (made between 2006 and 2008) plus simulation and synthesis modeling (Hearn 2008).

Dynamics of Tampa Bay

Tampa Bay is a semi-tropical, shallow, tidal estuary on the southwest coast of Florida. General circulation in the Bay is driven by wind, tides and buoyancy. These driving forces act on different time scales and it is now established (Hearn 2010) that the long term circulation is a mixture of the annual average of processes associated with all three forces. There are also much shorter term events that result from any one of the forcing factors. The long term average circulation consists of an inward surface wind-driven flow across the entire Bay. This penetrates into the water column only a few centimeters and is generally the result of fetch-limitation of winds which gives preferential weight to south-westerlies although the prevailing wind direction is from the northeast. Below this level is an outward flow of fresher water that derives from river flow and runoff. Finally there is bottom inward flow of marine water although this only occurs in the deeper central parts of the Bay. The strength of the inward bottom flow, and the outward flow above it, are dependent on the annual rainfall (minus evaporation and transpiration) which feeds river flow and runoff into the Bay. Hearn (2010) gives definitive estimates of water fluxes and sediment transport.

Waves in Tampa Bay are mainly driven by local winds. There is also some evidence for occasional swell waves close to the mouth and these may episodically penetrate further into the Bay. Wind waves are strongly fetch-limited so that their direction varies greatly between the many coasts in the Bay. Consequently, the littoral drift driven by waves is changeable in direction around the Bay and also affected by coastal constructions.

Dynamics of coastline, nearshore region and longshore bars

Longshore bars are common on coasts worldwide and generally considered to be formed by wave action. The dynamics of the bars (Hearn 2010) makes the bars roughly symmetrical but steeper near the crest and this is found to be true at Mariposa. The height of a bar depends on the amount of internal friction which is controlled by sediment grain size (and composition) and vegetative roots. Figure 2 shows a model simulation of the longshore bars on the outer part of the plateau at Mariposa with increased friction (broken line). The bathymetry is based on the existing bathymetry for transect 1 (full line) which is the most southerly transect shown in Figure 5. The increased friction could be due to coarser sediment or stabilization through seagrass roots. This emphasizes the possible impact of anthropogenic factors on the bars, including loss of seagrasses and changes in sediment sources (from coarser marine sediments to softer terrestrial sediments). We did not collect seagrass data at Mariposa but Figure 3 shows seagrass and bathymetric data transposed from a neighboring site (Avery and Johansson 2004), together with bathymetric data at Mariposa; this illustrates the effect of bars on seagrass habitat. Figure 3 shows the cross-shore transect 1 from Figure 5 (dotted curve) and the bathymetry along a nearby transect which is called S4T2 by the Interagency Seagrass Monitoring Program in Tampa Bay (full curve) (Avery and Johansson 2004). The green and red shading represents areal coverage by *Halodule wrightii* and *Thalassia testudinum*, respectively, based on data for 2003. The heights of the colored areas are proportional to coverage with maximum coverage at about 85%. Figures 2 and 3 are supportive of a symbiotic link between longshore bars and seagrass.

Dynamics of coastline, nearshore region and longshore bars

The basic mechanism for bar formation is that a small perturbation in the height of the seabed causes a convergence of sediment transport around the perturbation. This means that the perturbation tends to grow in height. Any such perturbation also tends to disperse under the influence of gravity so that sediment slides down its sides. On many coasts, wave-formed bars are the dominant stable coastal morphology; i.e. the coast is ‘barred’. On a low energy coast, such as Mariposa, we would expect the bars to show short-time variability (months) superimposed on much longer term variability (years to decades). As at Mariposa, longshore bars can exist in the inter-tidal zone and below the low water line (subtidal).

The general character of barred shores can be characterized in terms of their *large scale morphology*, the *overall bar structure* and the *internal bar structure*:

Large scale morphology—Apart from severe catastrophic events (such as hurricanes), or anthropogenic influences, the natural time scale on which the *large scale morphology* changes is related to the characteristic length scale. For Tampa Bay, treated as a whole, the observed general morphology has overall length scales of tenths of kilometers and so would be expected to change on time scales of order centuries prior to anthropogenic shoreline construction. Bathymetry charts of Tampa Bay are available for 1879, 1928, 1930, 1935, 1943/4, 1959, 1969, 1978 and 1988/9. Bathymetry data for some of these years are available in digital form from the author. These charts, which show only the coastline (they do not show submerged *nearshore* structure), indicate no major changes in the *coastline* at Mariposa and this is confirmed by *T* sheets of the Tampa Bay coastline.

Overall structure—This is simply the overall shape and form of the bars within the nearshore zone. It is basically the sort of structure that we might see from the window of an airplane as it ascends over the ocean from a coastal airport. These images have been captured in aerial photographs dating from the early decades of the 20th century. That photography reached reasonable quality by the 1950s and from that era we can estimate the width of the whole nearshore system and perhaps the width and number of bars and their approximate orientation to

the coast. We would expect the *natural timescales* for change in this overall structure to be measured in multiple decades or centuries, in the absence of human interference.

Internal structure—The *internal structure* of the bar system involves much more local detail of individual bars, i.e. their height and width, and this can change on a variety of much shorter time scales from months to years. This is due to the basic processes of wave and sediment dynamics not reaching a perfect equilibrium, or rather that the equilibrium is not a perfect steady state. Short term dynamics are controlled by wave-current instabilities, whereas the longer term changes in the internal structure are linked to the movement of sediment through littoral drift, onshore and offshore sediment transport (by currents and waves), and the supply of terrestrial sediment by erosion and runoff.

Bathymetry of Mariposa Coast

The nearshore structure at Mariposa is shown in Figures 4 to 6. Figure 4 shows the Mariposa coast from a 2003 U.S. Geological Survey (USGS) aerial photograph illustrating the longshore bars which extend out a distance of about 200 m on the outer nearshore plateau. The photograph shows alongshore periodicity of the bars which contributes to their short term instability and is of the order of about 100 m. Figure 5 shows the four detailed transects used in this study. The transects are numbered 1 to 4, moving along the coast (from left to right, in Figure 5). LIDAR data collected in 2003/4 is used as a background in Figure 5 with the colors red to blue spanning the depth 0 to -3 meters (relative to NAVD88). The coordinates are in meters based on UTM zone 17 (Hearn 2010).

Based on LIDAR (used in Figure 6), the nearshore region at Mariposa is seen to consist of a nearshore subtidal plateau which has a width of a few hundred meters and extends offshore from the coastline proper (defined at mean lower low water which is at -0.5 meters) to a depth of around -1.7 meters (NAVD88); the plateau has a depth range of about a meter providing a mean slope of 2 parts in a thousand, i.e., 0.002. The outer edge of the plateau has a relatively steep downward slope of about 0.02. The small longshore bars seen in Figure 4 are on the outer, subtidal, part of the plateau and have typical upward and downward slopes of between 0.005 and 0.02. The small bars are about 30 to 40 meters wide (crest to crest). They provide changes in water depth of a few tens of centimeters from the tops of their crests to bottom of the troughs. The mean astronomical tidal range at Mariposa is about 0.6 m with mean sea level (MSL) at -0.1 m (relative to NAVD88), mean lower low water (MLLW) at -0.5 m and mean higher high water (MHHW) at 0.1 m. At low water, the crest of the outer small longshore bar is about one meter below water level.

Figure 6 shows the positions of the still water level (SWL) at mean highest high water (MHHW) in blue, mean lowest low water (MLLW) in red and lowest low water (LLW) in green (see Table 1). Surface gravity waves are generally considered to break at a still water depth which is about 25% greater than the height of the waves. The short vertical lines on the distance axis between 0 and 200 m show the positions at which average wave breaking starts for the three water levels MHHW, MLLW and LLW (using the same color code as for the water level). This is based on waves of height 0.61 m (two feet) which is typical of Tampa Bay. Waves of greater height would break further offshore and those of lesser height further inshore. Figure 6 shows that the small longshore bars span the zone where wave breaking is likely to occur between low and high water for typical wind waves in the Bay. The edge of the plateau itself corresponds to the breaking point of these waves at lowest low water (LLW).

Nearshore of the three, or four, well-defined small longshore bars, at Mariposa, there are wider ridges on the 'back region' of the plateau. These ridges are not due to the normal action of wind waves and lack any real alongshore coherence (compare Figure 5). The variation in depth over

this back region is still large with gullies and pools down to -2 meters and crests rising to -0.5 m. Moving further onshore, this 'ridge and gully' topography merges with the coastal terrain and the developing shoreline.

Short term oscillations

Boat based sonar surveys were conducted at Mariposa on over the outer plateau to investigate short term variations in the longshore bars. Sonar coupled to a GPS system was used to determine cross-shore bathymetry along the four transect lines at Mariposa (Figure 5). Three one-day surveys (at approximately high tide) were made from 2006 to 2008. A MAKO 171 Center Console boat was outfitted with a Marimotech E-Sea Sound echo sounder (model MP35) and an Airmar transducer (model SSB203). Water depth was measured by transmitting a 200 kHz tone burst of acoustic energy with a narrow beam width (9° at -3db). Positional information was collected with a Thales choke ring antenna (model 701945-01 Rev E) mounted directly above the transducer and logged with an Ashtech Z-Xtreme™ GPS receiver.

Data were collected at 1 Hz which provided about 1 meter spatial resolution along the boat track. This resolution is controlled by boat speed, footprint of the sonar beam and the sampling rate of the GPS. The system provides absolute height of the sea bottom, i.e., it does not rely on tidal height of the water level. Differential corrections were applied to the GPS data using Waypoint GrafNav software which improved the accuracy to 0.5 m in the horizontal and less than 0.05 m in the vertical which was not yet sufficient for the present purposes. The horizontal and vertical data were referenced to the North American Datum of 1983 (NAD83) and North American Vertical Datum of 1988 (NAVD88) geodetic models, respectively, using the GEOID03 model. Waves in Tampa Bay are essentially wind driven with height and direction varying greatly with location due to the variability of fetch and coastal sheltering. Wave period is centered around two seconds, i.e., a frequency of 0.5 Hz. Since the GPS and sonar in the surveys are not synchronized, there is aliasing of depth data from wind waves which (even under the best of conditions used for the surveys) are significant. For this reason, and to improve the spatial resolution (mentioned above), it was necessary to make multiple passes along the cross-shore transects on each survey and then filter and average the data. All field surveys were conducted during single day campaigns and chosen for optimal tide conditions and acceptable sea states. This report is based on just three one-day field campaigns plus existing LIDAR data.

Figure 7 shows bathymetric data for transect 1 (the first transect on the left in Figure 5, i.e., the most southerly transect). The inset in Figure 5 shows the actual transect path set against the LIDAR data from 2003/4. The curves in Figure 5 show the depth of the seabed as a function of distance along the transect. Distance is shown on the lower axis of each panel and is measured in meters along the transect. The starting point of each transect is chosen to be at a water depth just in excess of 2 m, i.e., the edge of the coastal plateau. This starting point is maintained for each transect consistently throughout the surveys. The other axis is depth in meters relative to the NAVD88 height datum. That datum is very close to mean sea level (which is at -0.12 m). There are four curves for bottom bathymetry. Three of those come from the sonar surveys, and the fourth is derived by projecting the LIDAR data onto the transect line. Along each transect, the longshore bars are numbered starting from the outer edge of the transect. Similar data exist for the other three transects but are omitted here for brevity. All of the four data sets in Figure 7 show three well defined longshore bars which have been numbered. It is evident that the bars are part of a structure which has overall stability in the sense that the bars retain their identities over the period of the survey. Within that overall structure, the individual bars do show cross-shore movement, and erosion/accretion between surveys which can be as much as 10 m in the horizontal and 0.2 m in the vertical. The bars are evidently exhibiting short term oscillations due to wave current instabilities.

Longer term bar migration at Mariposa

A set of aerial digital images taken by various agencies of the Mariposa coast between 2004 and 2006 were geo-rectified and have been used here to create a time series of a small part of the Mariposa coast using a synthesis model of the nearshore region (Figures 8 and 9). The study area is at the northern end of the Mariposa coast (north of the area in Figure 5) and the heights were calibrated against a coastal point (outside of the scene shown in Figure 8). The time series was used to construct a movie sequence (this movie file is available from the author) of which two frames are shown in Figure 8. Notice that the frames have been rotated relative to those in earlier Figures in order to fit more conveniently on the printed page (so that the approximate direction of north is right to left). This synthesis model shows 3 or 4 discernable longshore bars (Figures 8 and 9). The color map in Figure 8 provides depth (blue is deeper water, moving through green/yellow to red and then brown for shallow water). The frames show the bathymetry for 2004 (left) and 2006 (right). The straight white line in each frame is a cross-shore transect. The synthesized bathymetry along this transect, for the two frames, is shown in Figure 9. The longshore bars are numbered (with arrows) in both frames starting offshore and moving inshore along the transect.

The time series shows two important features. The first is that the longshore bars show instabilities in the alongshore direction and appear to *wiggle* with time. This is evident by comparing the frames in the movie file. The other feature is that the outermost bar is dispersing and moving off the side of the plateau. This can be seen through a visual comparison of the outer edge of the plateau in the movie frames and in Figure 9. In the two-year period, longshore bar 1 has clearly dispersed and descended from the edge of the plateau. Longshore bar 3 has grown slightly at the expense of the bar 4 (the innermost longshore bar).

CONCLUSIONS

We have detailed data for bottom sediments in Tampa Bay from USGS and other agencies which is summarized in Figure 10. These are presented in Hearn (2008). Near the coast of the Bay, bottom sediments consist of two groups, i.e., the grain size distribution is bipolar. The larger grains are sand with a mean fineness (ϕ) of about 4, i.e., a grain diameter of about 60 μm . The small grains have a fineness of about 8, i.e., a grain diameter of about 4 μm . The relative proportion of these fine and coarser grains does vary over relatively short distances along the coast. The finer sediments are likely to be of terrestrial origin and sediment cores suggest that these sediments have been increasing in relative magnitude over the last few decades. This change is probably due to increasing urbanization of the Bay. Long term changes in sediment sources could be responsible for changes in the internal structure of the bars as well as having a direct influence on their suitability as seagrass habitat. For example, a change from coarse sand to fine sediment might create bars that have less height, as illustrated in Figure 2, with important consequences for seagrass habitat as emphasized by Figure 3. It is likely that increased seagrass coverage of the bars will dampen short term variations in the bars that we found in our sonar and aerial data.

The decadal variability seen at Mariposa, and which is often observed in longshore bar schemes, involves a slow offshore migration with bars eventually dispersing at the edge of the plateau. This process carries offshore the sediment that constituted the bar. To maintain the longer-term overall structure, new bars must form at the coast from sediment that may come from four sources:

- onshore movement of suspended sediment from the Bay induced by waves,
- coastal erosion and transport of erosion products,
- runoff from the coast bringing terrestrial sediment to the nearshore zone,
- littoral drift of sediment along the coast which could bring new sediments from the Bay or possibly from the Gulf coast.

It is possible, although not proven, that the stabilization by seagrasses would reduce the net decadal offshore movement of bars that we found in Figures 8 and 9.

A parametric wave model was run for the Mariposa coastline and indicated that the direction of littoral drift is inward, i.e., from left to right in Figure 5. This directionality is clearly substantiated by the observed accumulation of sediment south of the exit channel from Bishop Harbor (which is just north of Mariposa Key) as shown in Figure 11, and the bank of sediment on the other side of the channel. Notice also the form of the longshore bars south of the entrance to Redfish Creek to the north. South of Mariposa Key the longshore bars are diminished and Figure 11 shows strong depletion of sediment. The channels to the wetland areas south of Mariposa Key cut through these areas of sediment depletion, and generally run directly offshore, which is in contrast to the Bishop Harbor channel; this runs to the north indicating strong littoral drift in that direction. This northward set of the Bishop Harbor channel is seen in the earliest aerial photographs; the channel is only drawn symbolically on the 1879 map and so we have no direct evidence as to its historical orientation.

The littoral drift commences to the south of Mariposa and is strong enough to cause a northward swing of the channel to Bishop Harbor. The littoral drift has a role in the offshore migration of the longshore bars south of the channel to Bishop Harbor. The drift carries enough sediment across the mouth of that channel to contribute to offshore migration of the longshore bars north of the channel and to the accumulation south of Redfish Creek. There must either be a source for this sediment, or alternatively, the southerly boundary of the longshore bars could be migrating northward. A present source for the sediment could possibly be terrestrial including outflowing material from wetlands. An older source could have come from the mouth of the Bay and there is much anecdotal evidence that sediment supplies have declined on the lower part of the eastern side of Tampa Bay (an example is the beach at the De Soto Memorial on which Hernando de Soto splashed ashore at Tampa Bay in May 1539 which must now be rebuilt on the occasions that the landing is re-enacted). This decline, and perhaps change in the nature of the sediments, could have replaced some of the courser sediments found in the lower Bay with finer sediments and silts. This would contribute to the conjectured reduction in height of the offshore bars illustrated in Figure 2.

RECOMMENDATIONS

The present study has laid important foundations for understanding longshore bars and seagrass habitat at Mariposa Key. It can be furthered in value by a more intensive, and frequent, program of bathymetric surveys, especially if these are linked to simultaneous seagrass data collection. Collection and analysis of sediment cores at Mariposa is essential. It is also suggested that the sediment transport budget be further refined both by increased physical data collection and modeling.

ACKNOWLEDGEMENTS

The author wishes to thank the U.S. Geological Survey at the Florida Integrated Science Center in St. Petersburg, Florida for bathymetry data collected at Mariposa Key. All of the numerical modeling was performed by staff of Working Science Consultancies, LLC, in their St. Petersburg office using SPECIES software. Many thanks are due to Gregg Brooks for very considerable help with the understanding of sediments in Tampa Bay. Robin Lewis and Holly Greening provided much help and motivation for this study.

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Figure 1. Locale map of Mariposa Key in Tampa Bay.

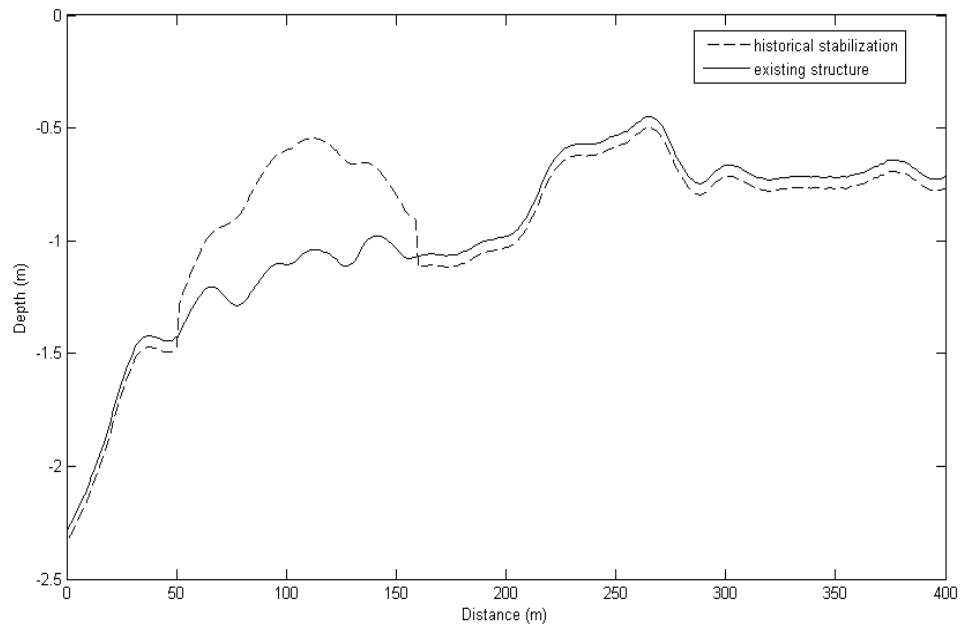


Figure 2. Simulation of the longshore bars at Mariposa with increased friction on the outer part of the plateau (broken line). From Hearn (2008) with permission.

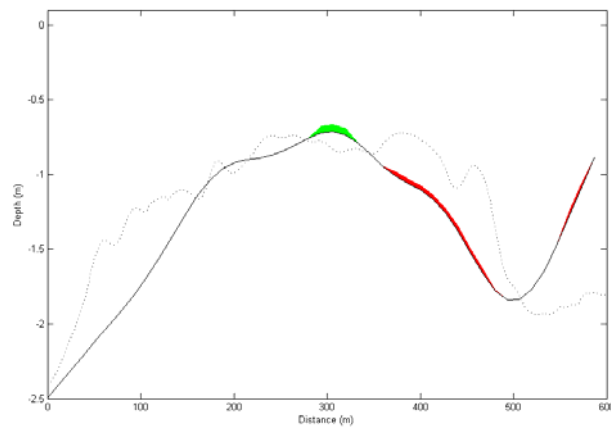


Figure 3. The cross-shore transect 1 and the bathymetry along a nearby transect called S4T2 used by the interagency seagrass monitoring program in Tampa Bay (full curve). The green and red shading represent area coverage by *Halodule wrightii* and *Thalassia testudinum*, respectively.



Figure 4. The Mariposa coast from a 2003 USGS aerial.

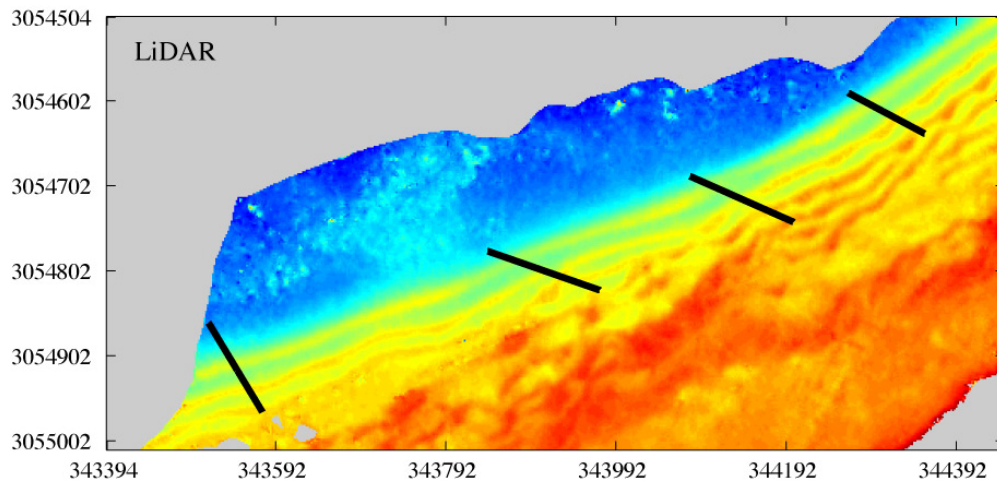


Figure 5. The four transects used in this study. The transects are numbered 1 to 4 moving along the coast from left to right. LIDAR data collected in 2003/4 is used as a background with the colors red to blue spanning the depth 0 to – 3 meters (relative to NAVD88). The coordinates are in meters based on UTM zone 17.

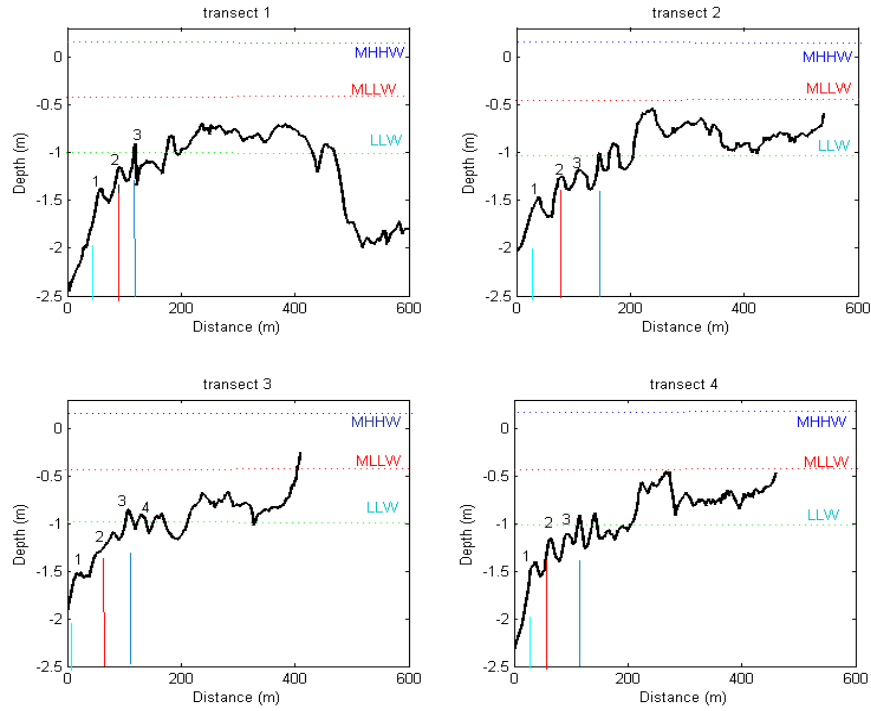


Figure 6. LIDAR bathymetry along the transects 1 to 4 shown in Figure 5 including MHHW, MLLW and LLW. The short vertical lines on the distance axis between 0 and 200 m show the positions at which average wave breaking starts for the three water levels using the same color code as for the water level. From Hearn (2008) with permission.

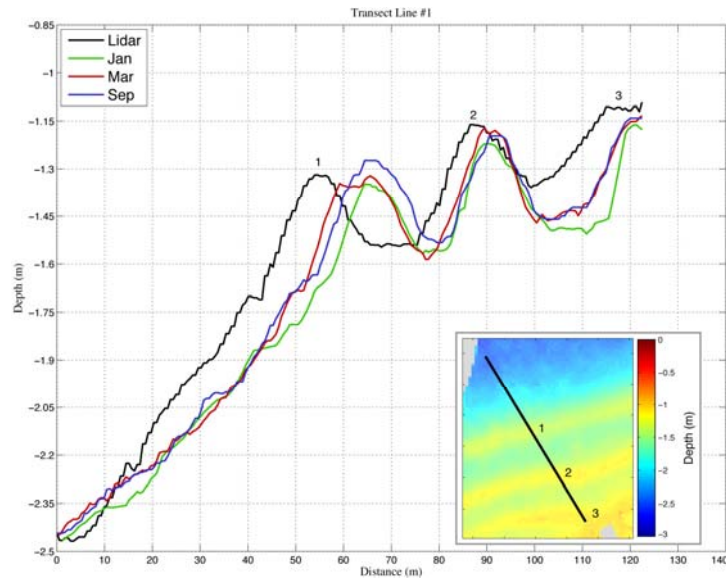


Figure 7. Bathymetric data for transects 1 and 4 (the first transect on the left and the last transect on the right in Figure 5). The inset in the Figure shows the actual transect path set against the LIDAR data from 2003/4.

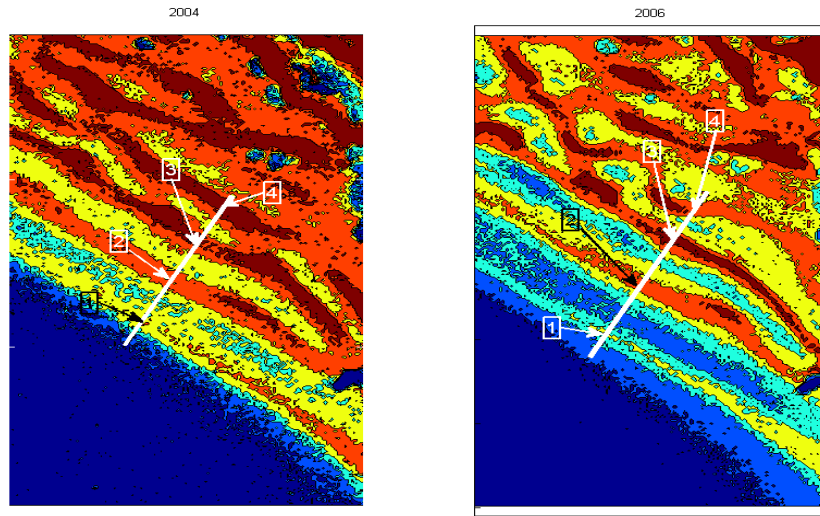


Figure 8. Two frames from the synthesis model of aerial data. From Hearn (2008) with permission.

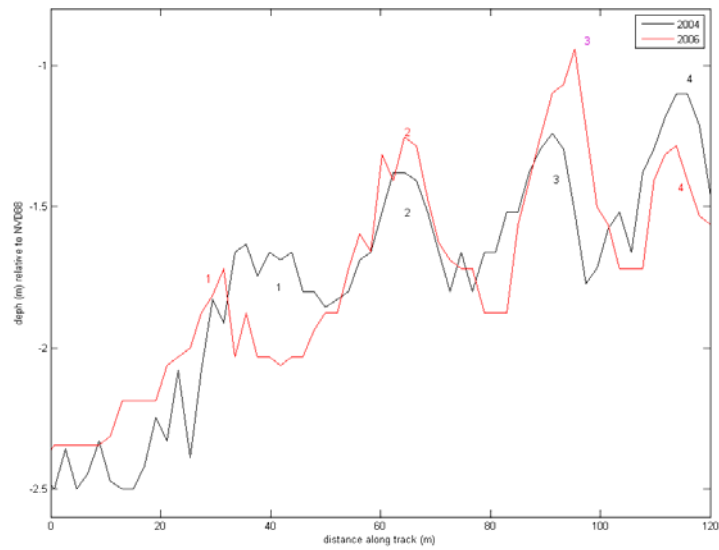


Figure 9. Synopsized model bathymetry for 2004 and 2006 along the transect shown in the two frames in Figure 8. From Hearn (2008) with permission.

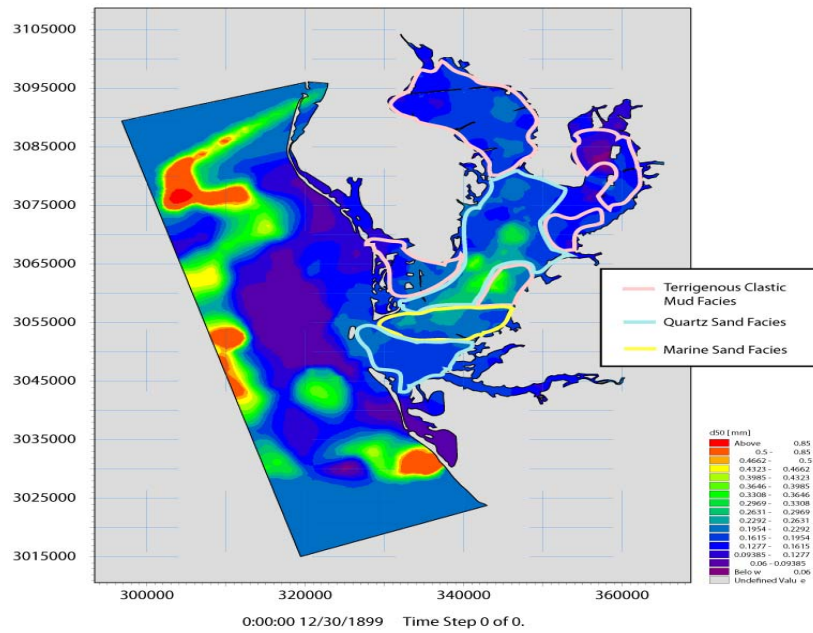


Figure 10. Sediment grain size and facies data for Tampa Bay (based partially on data from Brooks and Doyle 1998). From Hearn (2008) with permission.

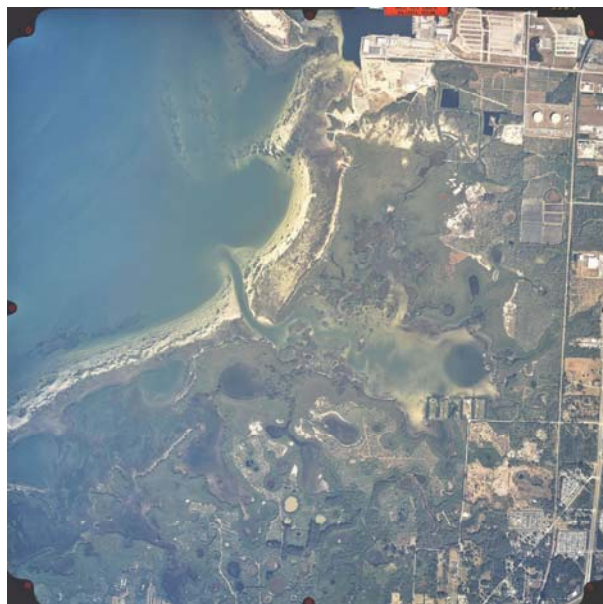


Figure 11. Google Earth view of Mariposa Key showing the entrance channel to Bishop Harbor and Redfish Creek to the north. Notice the sediment accumulations which are due to the northward littoral drift.

TWENTY YEARS OF TAMPA BAY SEAGRASS MAPPING AND ANALYSES

K.A. Kaufman & M.G. Heyl

ABSTRACT

The Southwest Florida Water Management District (District) boundaries contain five contiguous estuaries, including Tampa Bay, necessitating a large scale approach to monitoring the status of seagrass resources. Seagrass extent and distribution are surveyed biannually using aerial mapping. The objective of the mapping effort is to produce spatially- and thematically-accurate GIS coverages used to estimate total seagrass area. The program has documented temporal and spatial changes throughout the Tampa Bay study area over a twenty-year period. The 2008 seagrass estimates for Tampa Bay's seven bay segments totaled 11,996 ha. This is a 27 percent increase in Tampa Bay seagrass coverage compared to estimates from the District's first mapping effort in 1988. A review of changes in the program's use of mapping technologies and data analysis techniques in relation to mapping results over the last twenty years will be examined. Bay segment data was normalized using z-scores for detection of trends. The Tampa Bay estuary and various individual bay segments have significant positive correlations with time.

INTRODUCTION

Seagrass aerial mapping is a large-scale interdisciplinary approach for quantifying the status of seagrass resources. The approach combines two mapping techniques 1) aerial survey, the act of collecting the aerial photography source information, and 2) photogrammetry, defined as the art, science, and technology of obtaining reliable information about physical objects and the environment through processes of recording, measuring, and interpreting photographic images (American Society for Photogrammetry and Remote Sensing 2009). These elements must be combined into an approach that considers the constraints involved in capturing relevant benthic data as well as a biological and ecological understanding of the targeted feature to map: seagrass.

The Southwest Florida Water Management District (District) completed its first seagrass mapping effort in 1988, in order to document the status of the resource. An ongoing effort was then established to monitor seagrass as an indicator of the overall health of estuaries within the agency's jurisdiction. Since 1988, this seagrass mapping program has completed mapping products every two years for a total of ten mapping iterations. The program seeks to produce spatially and thematically accurate measures of seagrass areal extent and distribution.

CURRENT METHODS & METHODOLOGICAL ADVANCEMENTS

Seagrass aerial mapping is a multi-step process consisting of imagery acquisition, imagery processing, photointerpretation and benthic feature delineation, groundtruthing for signature verification, and accuracy field checks. The program has integrated new technologies for the acquisition of aerial imagery and advancements to geographic information systems (GIS) as necessary. Programmatic protocols have also been enhanced or adjusted as needed for improvement of mapping products. Highlights of recent enhancements incorporated into the District's program will be briefly discussed within subsequent sections.

Data Collection: Acquisition and Post Processing—The program conducts project-specific imagery acquisition. Strict data-capture criteria that must be satisfied in order to collect aerial photography include: atmospheric conditions, sun angle, water level, sea surface height, and water transparency conditions (Table 1). An additional criterion related to tidal cycle is currently under consideration. Capturing photography during outgoing tides may reduce incidences of reduced clarity in photography which occasionally occurred along inlets and passes, where exchange with the Gulf of Mexico suspends sediments and increases turbidity. In order to

include a tidal cycle criterion, new strategies for flight planning will be required and are under investigation.

Advancements in aerial photography acquisition allowed the program to transition in 2006 from true color film media to digital imagery. Digital imagery acquisition is conducted using a Z/I Digital Mapping Camera airborne imaging sensor. After acquisition, raw imagery undergoes post-processing and editing to reduce radiometric resolution (amount of information captured per pixel) from 12 to 8 bit, to remove distortion, and to register images to ground locations. The processing of raw imagery generates 0.3 m resolution digital orthophotographs. Imagery tiles derived from the 0.3 m digital orthophotographs are then reviewed for photographic signatures of benthic features and interpreted.

Photointerpretation and Signature Verification—Photointerpretation, or image interpretation, is the process of identifying and describing features in a photograph based on tone, color, contrast, and texture (Finkbeiner et al. 2001). Photointerpretation is currently conducted using heads up (interpreting from imagery on a computer screen) photogrammetric techniques and digital photogrammetric workstations are utilized for direct capture of digital data in an ArcGIS geodatabase (Photo Science 2008). Table 2 provides a summary of the techniques used prior to 2008. Photogrammetric and mapping techniques have also improved throughout the history of the program. Detailed descriptions for program methods prior to 2008 can be found in Kurz et al. (2000) and Tomasko (2005).

Once identified in the photography, benthic features are classified using modified codes from the Florida Land Use Land Cover Classification System (FLUCCS), see Table 3. A classification of either 9113 (patchy seagrass) or 9116 (continuous seagrass) is applied to each polygon delineating seagrass. In order to ensure consistency in delineation, the following classification conventions are used for patchy and continuous seagrass. Patchy seagrass is defined as greater than 25 percent unvegetated bottom visible within the seagrass polygon (Kurz et al. 2000). This category appears as singular, isolated patches of seagrass or extensive areas of patch strands mixed with open bottom. Typically these areas appear as rounded clumps, or as elongated strands (Photo Science 2008). Continuous seagrass is defined as less than 25 percent unvegetated bottom visible within a polygon (Kurz et al. 2000). These areas exhibit a continuous and uniform signature. Small, (less than 0.1 hectare) sandy bottom features may be interspersed within the bed, but these areas are not dominant (Photo Science 2008).

Visiting locations in the field allows photointerpreters to correlate the photographic signatures seen in the imagery with field conditions. Photographic signatures are evaluated and verified to successfully differentiate seagrass from features that may have similar appearances in photographs such as macroalgae and oysters.

Post Map Production Thematic Classification Accuracy Assessment—Thematic accuracy assessments are completed on the final products of each mapping effort. A random selection of points within Tampa Bay and the other four estuaries are field checked to verify the map has correctly identified the presence of seagrass. A 90 percent accuracy rate must be achieved for acceptance of a final map deliverable by the District.

Analyses—The vector data model is analyzed using spatial calculations in GIS to create areal estimates of seagrass. Coverage data is presented for Tampa Bay and each of the seven bay segments, as described by Lewis and Whitman (1985). The seven segments are: Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay, Boca Ceiga Bay, Lower Tampa Bay, Terra Ceia Bay, and Manatee River.

The District has completed 10 map iterations ($n=10$). The data were analyzed for temporal trends at two scales: estuary-wide and by individual bay segment. Evaluation of the data was completed for three different classifications: patchy seagrass, continuous seagrass, and total seagrass cover (sum of patchy and continuous). The areal measures of mapped polygons were used as the metric of investigation. The data were normalized by z-score in order to complete inter-segment comparisons on a common scale. The z-score is a measure of how much a bay segment deviates from its distribution's mean. By definition, the normalized distribution of each bay segment results in a mean of zero and a standard deviation of one. Years with seagrass acreages less than the long-term average will have negative z-scores (Kaufman 2006). The ratio of continuous seagrass area to patchy seagrass area was determined for all segments and estuary-wide as a means for investigating whether the types of seagrass classifications within an area had changed over time. The normalized data were then analyzed for correlations using Kendall's Tau and significance was determined from a look up-table for $n \leq 10$ (Kendall 1975).

RESULTS

Tampa Bay has experienced a 27 percent increase in seagrass coverage between 1988 and 2008. Data is presented in Figure 1 as a time series. Tampa Bay total seagrass cover was documented as 11,996 ha in 2008.

Each seagrass class (patchy, continuous, and total cover) for the Tampa Bay estuary was correlated with time, to determine if the estuary has exhibited significant changes in overall seagrass areal cover (Figure 2). Total seagrass area was significantly correlated ($p \leq 0.05$) with time, displaying a positive monotonic association. A positive monotonic correlation was also found for the estuary's continuous areal cover ($p \leq 0.05$). Spatial trends in estuary-wide seagrass coverage changes were investigated. The Tampa Bay estuary and Sarasota Bay estuary had a significant positive correlation for total seagrass area ($p \leq 0.05$) and continuous seagrass area ($p \leq 0.05$) but not patchy seagrass area.

Changes in seagrass cover at the smaller scale of individual bay segments were also tested. Total seagrass area in five of the seven bay segments had significant correlations with time (Table 4). The areas of continuous seagrass in four of these five segments were also positively correlated with time (Table 4), while only two segments, Hillsborough Bay and Manatee River, have positive monotonic trends in patchy seagrass. To evaluate changes in the nature of seagrass coverage, the distribution of the two seagrass cover types (the ratio of continuous seagrass area to patchy seagrass area) was correlated with time. No trend was found in the ratio of continuous to patchy seagrass for the Tampa Bay estuary and five of the seven segments (Table 5).

DISCUSSION

After completion of each mapping effort the data is reviewed to determine the status of Tampa Bay seagrass cover and for detection of trends. Results for the District's mapping program have most often been presented as comparisons of data from multiple time periods or subsequent data years. Percent change (new observation – old observation / old observation) or percent difference (new observation / old observation) are most often reported (Kurz et al. 2000, Tomasko 2002, Tomasko 2005). As an update to previous accounts of this program (Kurz et al. 2000, Tomasko 2002, Tomasko 2005, Tomasko et al. 2005, Kaufman 2006) the authors sought to examine the data using alternative forms of analysis. Analyses focused on reviewing changes documented by the program over the twenty-year period 1988 to 2008.

The Tampa Bay estuary experienced significant increases in seagrass coverage for two of the three seagrass classes. Total seagrass area and continuous seagrass area increased over time for the entire bay. Tampa and Sarasota Bays were found to be responding similarly, with overall increases in seagrass cover. Throughout the individual bay segments, a somewhat consistent

pattern of change was displayed, similar to the change estuary-wide for Tampa Bay. Two segments, Old Tampa Bay and Terra Ceia Bay, showed no trend in seagrass cover (for any class) over time. Both segments displayed more cyclical patterns that were not significant using Kendall's Tau test for correlation. Future analysis could include investigation of these segments' association with water quality parameters that may display similar variability and cyclical patterns like salinity.

Only two segments, Hillsborough Bay and Manatee River had significant increases in patchy seagrass area. Interesting to note, Hillsborough Bay was the only bay segment to experience gains in all three seagrass classes (patchy, continuous, and total). However; these increases occurred without a significant change in the ratio of continuous to patchy coverage, meaning both classes were increasing but the segment does not show a change in the nature of its coverage. The Manatee River segment experienced significant increases in patchy and total seagrass coverage and was the only bay segment to have a significant decrease in its ratio of continuous to patchy seagrass area. This indicates the Manatee River segment experienced expansions and shifts to more patchy seagrass coverage. Middle Tampa Bay was the only segment showing a positive change to continuous seagrass. Overall, there was no indication of significant shifts in the relative amount of either class of seagrass. A dynamic equilibrium may be occurring where the loss and gain of seagrasses within and among the two different classes are balanced at the scales investigated (Walker et al. 2006). Alternative techniques for the detection of change in the nature of mapped seagrass will be investigated in future work using GIS spatial analysis.

Results from the District's twenty-year seagrass mapping program show significant increases in the gross seagrass coverage of Tampa Bay. On a whole, most portions of the bay have documented increases in seagrass cover over time. However; the slight differences in trends over time for coverage and changes in seagrass classes may indicate variations of responses are occurring for seagrass in those portions of the bay. The determinations of status and trends in bay-wide seagrass coverage and most importantly the documentation of Tampa Bay's large scale recovery have been possible due to the District's efforts in monitoring seagrass resources.

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Table 1: Criteria required for imagery acquisition.

Environmental Consideration	Specification Required for Data Capture
Atmospheric Conditions: Cloud Cover	0% allowable
Atmospheric Conditions: Wind	≤ 10 knots
Sea Surface: Wave Height	≤ 61 cm (2 ft)
Sun Angle	30° - 45°
Tidal Stage	≤ Mean tide level
Water Transparency	2 m
Tidal Cycle (phase)*	Ebb tide cycle

* proposed criteria

Table 2: Summary of mapping techniques utilized for each mapping effort.

Year	Media	True Scale	Mapping Scale	Photogrammetric and Mapping Techniques Employed
1988	Natural Color Film	1:24,000	1:24,000	Manual cartography
1990	Natural Color Film	1:24,000	1:24,000	Manual cartography
1992	Natural Color Film	1:24,000	1:24,000	Manual cartography
1994	Natural Color Film	1:24,000	1:24,000	Manual cartography
1996	Natural Color Film	1:24,000	1:24,000	Manual cartography
1999	Natural Color Film	1:24,000	1:12,000	Analytical Stereoplotters
2002	Natural Color Film	1:24,000	1:12,000	Analytical Stereoplotters
2004	Natural Color Film (Scanned)	1:24,000	1:12,000	Soft Copy, Stereoscopic
2006	Digital True Color Imagery	1:24,000	1:12,000	Stereoscopic
2008	Digital True Color Imagery	1:24,000	1:12,000	Monoscopic

Table 3: Modified Florida Land Use Land Cover Classes used in seagrass mapping.

FLUCCS Code	Definition/Description
0	Not Classified (Land)
5400	Bays and Estuaries
5700	Marine (Gulf of Mexico)
7100	Beach, other than swimming
6510	Tidal Flats, in waters no more than 2 m, periodically exposed
9113	Seagrass, Patchy
9116	Seagrass, Continuous
9121	Attached Algae, pure stand and field verified to Caulerpa genus

Table 4: Results of bay segment correlations with time for the three seagrass classes.

Tampa Bay Segments	Total Area	Patchy Area	Continuous Area
Old Tampa Bay	ns	ns	ns
Hillsborough Bay	**Positive	**Positive	**Positive
Middle Tampa Bay	*Positive	ns	**Positive
Boca Ceiga Bay	**Positive	ns	**Positive
Lower Tampa Bay	*Positive	ns	*Positive
Manatee River	**Positive	**Positive	ns
Terra Ceia	ns	ns	ns

* $p \leq 0.1$; ** $p \leq 0.05$; ns = not significant

Table 5: Results for ratios of continuous seagrass to patchy seagrass correlations with time.

Waterbody (Estuary or Bay Segment)	Continuous Seagrass Area / Patchy Seagrass Area
Tampa Bay Estuary	ns
Old Tampa Bay	ns
Hillsborough Bay	ns
Middle Tampa Bay	* Positive
Boca Ceiga Bay	ns
Lower Tampa Bay	ns
Manatee River	**Negative
Terra Ceia Bay	ns

* $p \leq 0.1$; ** $p \leq 0.05$; ns = not significant

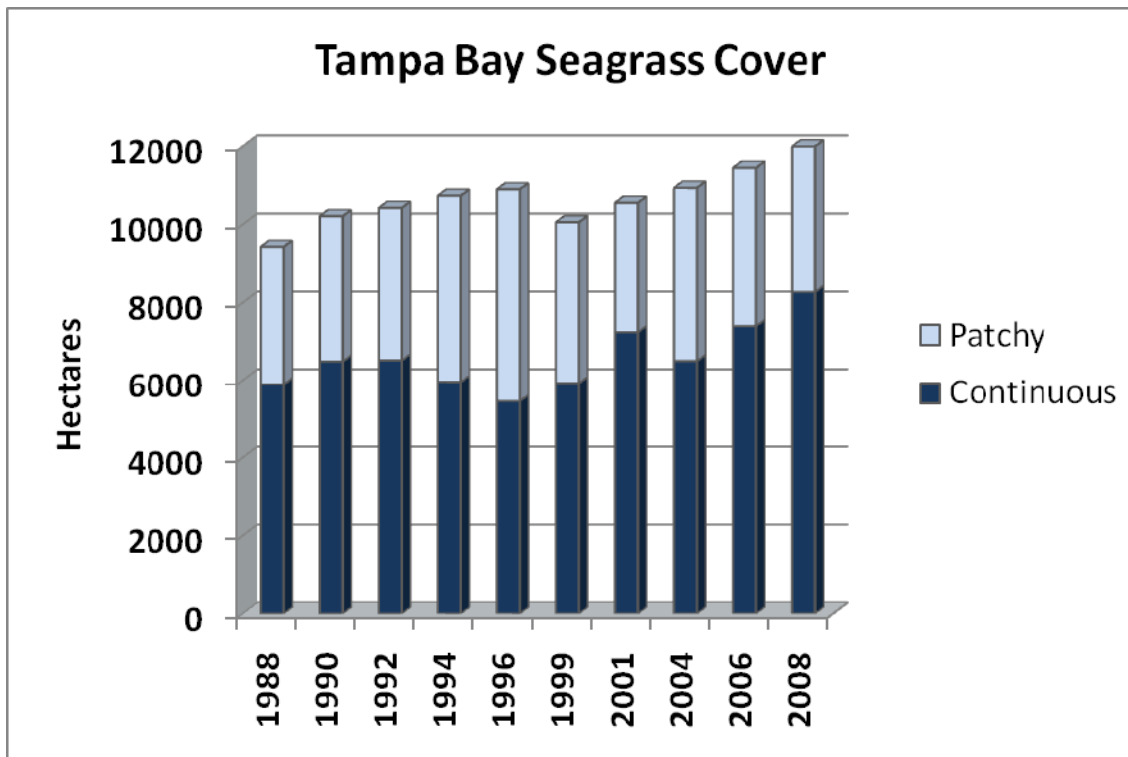


Figure 1: Results for total, patchy and continuous seagrass in Tampa Bay from 1988 to 2008.

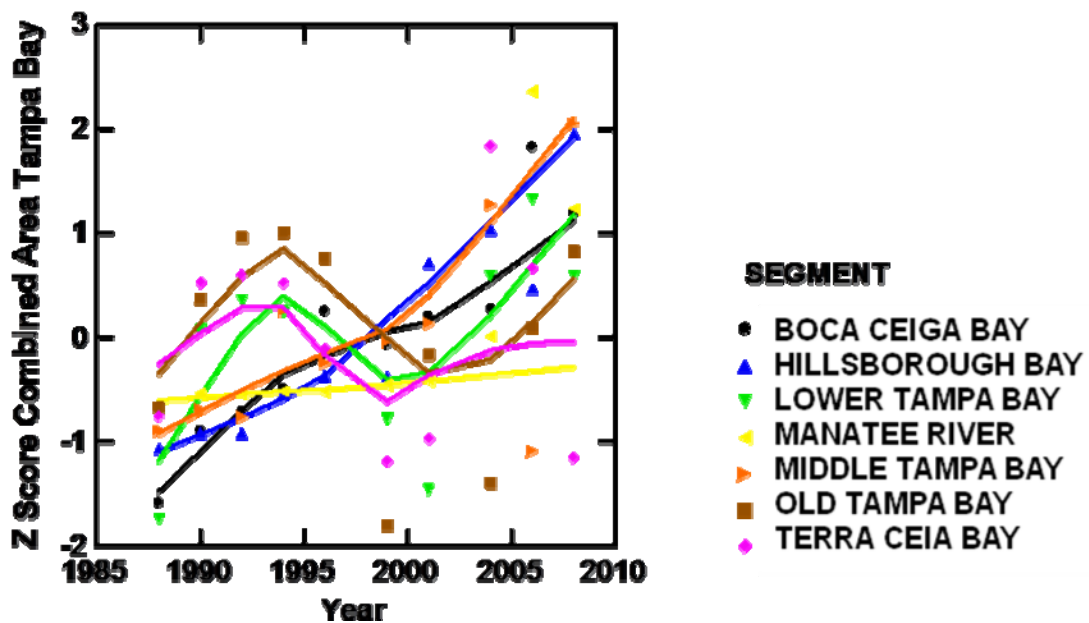


Figure 2: Total (combined) seagrass cover for each bay segment over time. Lowess smooth lines are plotted as a tool for visually discerning trends.

TRENDS IN TAMPA BAY SEAGRASS: 1998-2008

W.M. Avery, J.O.R. Johansson, K.B. Hennenfent & J.J. Pacowta

ABSTRACT

Under the auspices of the Tampa Bay Estuary Program, ca 60 fixed transects have been monitored since 1998 to document changes in seagrass species composition, abundance, and zonation throughout Tampa Bay. *Halodule wrightii* has been the most common species, found at twice the frequency of *Thalassia testudinum*. Also, *Syringodium filiforme* has been a significant contributor to seagrass composition while *Halophila engelmanni* and *Ruppia maritima*, and the attached alga *Caulerpa* (primarily *C. prolifera*) have been minor constituents. Seagrass frequency was stable between 1998 and 2004 with nearly 50 percent of meter square placements containing seagrass; however, since that time, the frequency has increased to about 60 percent, primarily due to *H. wrightii* expansion. Bay-wide abundance as determined by the Braun-Blanquet class coverage assignments has been relatively stable. The greatest abundance variability has been seen with Hillsborough Bay *H. wrightii*.

INTRODUCTION

In 1996, the Tampa Bay National Estuary Program (now Tampa Bay Estuary Program or TBEP) adopted a resource-based management strategy to protect and restore Tampa Bay (Tampa Bay National Estuary Program 1996). Subsequently, a Tampa Bay seagrass monitoring program was designed following many recommendations outlined by the Program's Technical Advisory Committee (Squires et al. 1993).

In 1997, the City of Tampa, Bay Study Group (BSG) began a fixed seagrass transect monitoring program in Hillsborough Bay, the northeastern subsection of Tampa Bay. This effort was expanded in 1998, under the auspices of the TBEP, to encompass the entirety of Tampa Bay. In all, seven agencies have participated in the Tampa Bay Interagency Seagrass Monitoring Program (TBISP) and include: Environmental Protection Commission of Hillsborough County, Florida Fish & Wildlife Conservation Commission - Florida Marine Research Institute, Hillsborough County Cockroach Bay Aquatic Preserve, Manatee County Environmental Management Department (MCEMD), Pinellas County Department of Environmental Management, Tampa Bay Watch, Inc., and the BSG.

The objective of this paper is to present trends of submerged aquatic vegetation (SAV) by bay segment and Tampa Bay as a whole. Two SAV elements are presented: abundance as determined by the Braun-Blanquet coverage class assessments (Braun-Blanquet 1965) and the percent frequency of occurrence (PFOC).

METHODS

The number of Tampa Bay transects assessed has ranged from 57 to 62 due to a number of issues including safety concerns (dredging, vessel traffic), manpower constraints, and deletion/addition of sites to produce a more robust and representative data set. Figure 1 illustrates the 62 site locations of the Tampa Bay fixed seagrass transects during 2008.

Transect monitoring is conducted each fall along a predetermined line usually starting near shore, traversing a potential seagrass area, and terminating near a specific depth depending on bay subsection. With one exception at Egmont Key (Figure 1), all transects have a terminus deeper than existing seagrass coverage. Seagrass abundance is assessed at specific intervals along each transect using the Braun-Blanquet class coverage assessment (BB) in which a SAV coverage percentage within a square meter is assigned a numerical rating of 1-5. For a more detailed methodology, please see Avery and Johansson (2001).

Percent frequency of occurrence (PFOC) was determined for each Tampa Bay subsection and the entirety of Tampa Bay. PFOC is defined as the number of times a SAV species was found within a specific data set of meter square placements divided by the total number of meter square placements within that data set.

The number of meter square placements for SAV assessment has varied between Tampa Bay subsections and has been driven by transect length and placement intervals. For example, the average number of meter square placements assessed annually has ranged from 500 in Old Tampa Bay (OTB) to 140 in Boca Ciega Bay (BCB). As a result, a SAV change in a certain number of meter square placements within the greater data set has had a smaller impact on PFOC than a similar change in the same number of meter square placements within a smaller data set. Therefore, PFOC differences between bay subsections may not reflect a similar magnitude of change in SAV.

RESULTS

Old Tampa Bay—Five seagrass species, *Halodule wrightii*, *Halophila engelmanni*, *Ruppia maritima*, *Syringodium filiforme* and *Thalassia testudinum* have been documented in OTB (Figure 1) since 1998. *H. wrightii* has dominated OTB seagrass composition with the PFOC over twice that of other OTB SAV components (Figure 2A). In addition, the PFOC for this species increased from ca 30% in 1999 to over 45% during 2008. Further, the average *H. wrightii* BB abundance increased substantially since the beginning of the program. *S. filiforme* and *T. testudinum* have been major components of OTB seagrass meadows (Figure 2 D-E). The PFOC for *S. filiforme* doubled between 2001 and 2008 with a similar increase observed for *T. testudinum*; however, the PFOC for each species has increased only slightly since 2004. Although the average BB abundance for *S. filiforme* has fluctuated somewhat, there was little net change over eleven years. In contrast, *T. testudinum* BB abundance decreased during 1998-2004 before stabilizing through 2008.

Compared to the three species previously discussed, *H. engelmanni* and *R. maritima* have been minor OTB seagrass constituents (Figure 2B-C). The PFOC for each species has been <10% except during 2005 when *R. maritima* became a prevalent inshore species along two upper OTB transects (Avery and Johansson 2006a). The average BB abundance for each species has been widely erratic with no discernable pattern.

The net seagrass PFOC and BB abundance in OTB during 1998-2008 has been positive (Figure 3A). Following an initial seagrass PFOC decrease, the trend has generally increased, driven by *H. wrightii*, *S. filiforme*, and *T. testudinum* PFOC increase. In addition, BB abundance, driven by a substantial increase since 2005, has also increased. However, only *H. wrightii* and *S. filiforme* have been responsible for the abundance increase.

The attached alga, *C. prolifera*, has been a notable contributor to OTB SAV (Figure 2F). This alga's PFOC generally has been above 5% during the course of the study and was over 10% in 2008. Similar to *H. engelmanni* and *R. maritima*, the average *C. prolifera* BB abundance has been erratic with no discernable pattern.

Hillsborough Bay—*H. wrightii* has been the dominant of the two seagrass species found in Hillsborough Bay (HB; Figure 1). The HB *H. wrightii* PFOC (Figure 4A) increased 11% during 1997-2002 and then declined to 1997 levels between 2003 and 2005. However, during 2005-2008, the PFOC increased 11%. The average BB abundance generally follows the increasing/decreasing pattern of the PFOC; however, there has been a net decrease overall. The second HB seagrass species, *R. maritima*, has been minor SAV component. The PFOC for this

species has been <5%, with the maximum found during 2005 (Figure 4B). The average BB abundance has been erratic with no apparent trend.

The net seagrass PFOC in Hillsborough Bay during 1997-2008, driven largely by *H. wrightii*, has been positive (Figure 3B). In contrast to OTB where BB abundance has increased since 1998, the BB abundance in HB has decreased considerably since the first assessment.

C. prolifera was absent in HB during the early portion of the study (Figure 4C), but became a prevalent SAV component after 2003. This alga was found in 13% of meter square placements during 2007, the highest value reported during the monitoring period.

Middle Tampa Bay—Except for the absence of *H. engelmanni*, Middle Tampa Bay (MTB; Figure 1) SAV composition was similar to that seen in OTB. *H. wrightii* (Figure 5A) was found at twice the frequency of two other prominent MTB seagrass species, *S. filiforme* and *T. testudinum*. *H. wrightii* PFOC increased 9% between 2000 and 2008. *H. wrightii* BB abundance reached a maximum and minimum during 2000 and 2004, respectively. However, following the 2004 minimum, the BB abundance increased in each subsequent year. *S. filiforme* and *T. testudinum* have been major components of MTB seagrass meadows, however, *S. filiforme* was found at slightly higher frequency than *T. testudinum* (Figure 5C-D). *T. testudinum* PFOC has not changed appreciably during the course of the program, averaging about 10%, while *S. filiforme* PFOC has increased, attaining over 20% during 2006-2007. *S. filiforme* BB abundance has been erratic with no apparent trend while *T. testudinum* BB abundance, following a decrease during 1998-1999, has increased slightly since that period. *R. maritima* has been a minor MTB component (Figure 5B) generally confined to upper MTB. Similar to the trend seen in OTB and HB, the PFOC for this species increased during 2005 before declining in subsequent years. The *R. maritima* BB abundance has been erratic with no apparent trend.

The net seagrass PFOC increase in MTB was miniscule during 1998-2008 (Figure 3C). A slight *H. wrightii* PFOC loss was offset by PFOC gains for *S. filiforme* and *T. testudinum*. In addition, BB abundance increased somewhat, driven by *H. wrightii* and *S. filiforme*.

In yet another similarity to OTB and HB, the *C. prolifera* PFOC peaked during 2005 (Figure 5E). However, in contrast to the two upper Tampa Bay subsections, the MTB *C. prolifera* PFOC dropped precipitously during 2005-2006. The BB abundance for this species has been erratic with no apparent trend.

Lower Tampa Bay—Each of the five Tampa Bay seagrass species and *C. prolifera* has been found in Lower Tampa Bay (LTB; Figure 1). In contrast to the *H. wrightii* dominance seen in the upper portions of Tampa Bay, this bay subsection has been *T. testudinum*-dominated. *H. wrightii* PFOC has been fairly stable (Figure 6A). After a slight decrease early in the study, *H. wrightii* PFOC has increased somewhat since 2003. Similar trends were seen for *H. wrightii* BB abundance. *T. testudinum* PFOC has been generally twice that seen for *H. wrightii* (Figure 6E). After a *T. testudinum* PFOC decline during 1998-2002, this species appears to have stabilized. *T. testudinum* BB abundance also decreased early in the study, but has generally increased since 2001. *S. filiforme* has been another prominent LTB seagrass component. This species has been relatively stable with a PFOC hovering near 10% (Figure 6D). *S. filiforme* BB abundance has been erratic with no apparent trend. *R. maritima*, *H. engelmanni*, and *C. prolifera* have been minor SAV species. Each species has been ephemeral in regards to PFOC (Figure 6B-C, F) and BB abundance trends have been driven by the presence/absence of each species.

LTB seagrass trends (Figure 3D) have been driven by *T. testudinum*. Seagrass PFOC in LTB declined considerably during 1998-2004, primarily due to the loss of *T. testudinum* and to a lesser degree, by *H. wrightii*. However, seagrass PFOC, driven by *T. testudinum*, increased during 2005-2006 before the trend leveled during 2006-2008. Finally, the net seagrass BB abundance was unchanged during the course of the program.

Boca Ciega Bay—Three seagrass species, *H. wrightii*, *S. filiforme* and *T. testudinum* have been found in BCB (Figure 1). *C. prolifera* has been a rare SAV constituent (Figure 7D) in this Tampa Bay subsection. *H. wrightii* has had the greatest PFOC of the three seagrass species, present in nearly 65% of meter placements (Figure 7A). Following a decrease during 1998-2002, *H. wrightii* BB abundance increased since 2003. Though *T. testudinum* PFOC (Figure 7C) has been slightly less than *H. wrightii*, it may be considered a co-dominant species. Although slight increases and decreases have been noted, *T. testudinum* PFOC and BB abundance have been generally stable. Finally, *S. filiforme* PFOC has increased steadily since 1999 (Figure 7B), however, *S. filiforme* BB abundance has been erratic with no apparent trend.

BCB seagrass trends have been driven by *H. wrightii*, *S. filiforme* and *T. testudinum*. Decreased *H. wrightii* and *T. testudinum* PFOC was responsible for a decline in seagrass PFOC during 1998-2000. However, BCB seagrass PFOC increased substantially during 2004-2007 due to increased PFOC in each of the three species. Overall, there was a slight seagrass PFOC increase during 1998-2008 (Figure 3E). There was a slight PFOC drop during 2008 in BCB as *T. testudinum* PFOC decreased. BB abundance increased slightly in BCB during 2004-2008, primarily due to *H. wrightii*.

Tampa Bay—Of the five Tampa Bay seagrass species, *H. wrightii* has been the most frequently encountered with the PFOC increasing from 34% to 40% during 1998-2008 (Figure 8A). *T. testudinum* and *S. filiforme* are the next most common seagrass species; however, *T. testudinum* PFOC has remained just under 20% (Figure 8E) while *S. filiforme* PFOC increased from 5% to 12% (Figure 8D). Although there were some interannual BB abundance variations, the *H. wrightii* and *T. testudinum* values remained stable while *S. filiforme* BB abundance increased. *R. maritima* and *H. engelmanni* were minor seagrass species (Figure 8B-C), found in less than 2% of meter square placements in all years except 2005 when *R. maritima* PFOC reached 5%. BB abundance for the latter two species was lower than the three dominant Tampa Bay species.

C. prolifera PFOC (Figure 8F) was greater than *R. maritima* and *H. engelmanni* but lower than *H. wrightii*, *S. filiforme*, and *T. testudinum*. The PFOC, which increased somewhat during 1998-2008, peaked to 10% during 2005 and was followed by a decline in 2006. Although the BB abundance has been erratic, there has been a general decline during the study.

Seagrass trends in TB changed little during 1998-2004. Following this period, TB seagrass PFOC (Figure 3F) increased in 2005 and then decreased slightly during 2006-2008. Each TB bay segment except for HB contributed to the increased PFOC during 2005, however, only in HB did the PFOC increase during 2006-2008. BB abundance in TB followed a similar pattern as the PFOC, remaining relatively unchanged until increasing in 2006. In contrast to the PFOC, the BB abundance changed little during 2006-2008.

DISCUSSION

Each year since 1998, the TBISP has assessed SAV along ca 60 transects. Annual data has been collected from an average of nearly 1700 meter square placements along 34ha of Tampa Bay substrate. The data have shown Tampa Bay to be a *H. wrightii*-dominated estuary, with all bay subsections except LTB having *H. wrightii* as the most common seagrass species. LTB seagrass meadows have been dominated by *T. testudinum*.

The greatest seagrass PFOC increases in Tampa Bay have been due to *H. wrightii* and *S. filiforme*. The increased *H. wrightii* PFOC has been driven by gains in OTB, HB and to a lesser extent, BCB. The positive trend seen for *S. filiforme* has been due to increases in OTB, MTB, and BCB. The *S. filiforme* changes in BCB were largely due to a species shift from *T. testudinum* to *S. filiforme* at one transect.

The decrease of *T. testudinum* PFOC in LTB is of concern. Several areas from Bishops Harbor to the Manatee River lost low density, offshore coverage between 1998 and 2002 (Avery and Johansson 2006a). In addition, a large area of dense offshore *T. testudinum* just south of the Skyway Bridge present during 2007 was not seen during 2008. The reasons for this loss are not clear and warrant further investigation.

The Southwest Florida Water Management District, Surface Water Improvement and Management Program (SWIM) has conducted SAV estimates from aerial photography interpretations on a biennial basis since 1988. Tampa Bay seagrass coverage during 1999 was determined at 24,841 acres (Kaufman 2006) and increased 19% to 29,630 acres in 2008 (Kaufman and Heyl 2010). In spite of differences in each program's methods and objectives, it is interesting to note that TBISP data indicate a 10% PFOC increase during the same period (Figure 3F). If the SAV PFOC, which includes *C. prolifera* as well as seagrass, is used to gauge the percent change, the PFOC change increases to 13% during 1999-2008, closer to the SWIM areal coverage trend.

Each Tampa Bay subsection, except for Middle Tampa Bay, had a seagrass PFOC decrease following the initial assessment (Figure 3A-E). This decrease followed the 1997-1998 "El Nino" event when greater than average rainfall resulted in a large freshwater influx to Tampa Bay. Except for LTB, seagrass PFOC began to increase in subsequent years until 2003-2004 when rainfall from several hurricanes again contributed to a large freshwater influx to the bay. During this period, seagrass PFOC in Tampa Bay subsections remained unchanged or decreased. Contributions by a large freshwater inflow may include depressed salinity, changes in non-chlorophyll turbidity (suspended detritus), greater chromophoric dissolved organic matter, and increased nutrient availability that may drive phytoplankton growth. The first comprehensive Tampa Bay study to assess the potential impact of these parameters on the spectral light quality level needed for seagrass maintenance is discussed by Anastasiou et al. (2010).

Other factors that may affect seagrass recolonization and warrant further study are sediment transport and nutrient-enriched submarine groundwater discharge. The loss of Hillsborough Bay seagrass was documented during and following the movement of a sand bedform through an area of developing *H. wrightii* (Avery and Johansson 2006b). Further *H. wrightii* loss was documented on a shallow Hillsborough Bay sandflat during 2001, a drought year, and evaluation of this area revealed buried drift macroalgae (W. Avery pers. ob.) suggesting sediment transport may have impacted seagrass. In addition, observations made by MCEMD in the Bishops Harbor area have led them to postulate that sediment transport may play a role in offshore seagrass loss along a TBISP transect (K. Harkinson pers. comm.). Therefore, a periodic collection of seabed elevation data along seagrass transects needs to be expanded. To date, baseline depth data has been collected along Hillsborough Bay transects and selected transects in Middle and Upper Tampa Bay. Finally, recent work by the USGS (Kroeger et al. 2007) indicates that groundwater seepage in Tampa Bay, already containing dissolved nitrogen and phosphorus, may increase nutrient flux from sediments. The impact of these nutrient sources on seagrass or factors impacting seagrass such as epiphyte loading is not known.

CONCLUSION

The TBISP program has evolved since its inception in 1998. As data was collected and assessed, it became apparent that several transects, due to location, substrate, and SAV composition, did not provide sufficient information to characterize seagrass trends. Therefore, during the first few years of the program, several transects were deleted while new transects were added in areas of interest. This flexibility in the program has resulted in a more robust database targeting the annual status of seagrass species stratified over depth gradients. In the future, assessments of the effectiveness of the program must include the same flexibility in regards to quality and perhaps the frequency of data collection. Finally, an annual training exercise to generate statistical comparisons within data collections of participating agencies must be continued.

Independent assessments of Tampa Bay SAV coverage by TBISP and SWIM agree that, over the past decade, there has been a net increase in SAV. The SWIM assessment of percent increased seagrass coverage is twice that of the increase of TBISP PFOC. The TBISP protocols allow the portioning of SAV into species of seagrass and attached algae and the addition of *C. prolifera* to the seagrass assessment allows the net PFOC gain to increase from 10% to 13%. Therefore, it would seem prudent for bay managers to consider results from TBISP while assessing SAV areal coverage trends.

Continued investigations into factors affecting seagrass health are imperative. A more in-depth understanding of seagrass biology, including how each species reacts to changes in light quality due to epiphytic loading and changes in water column constituents, is needed. Further, studies into the factors that drive these impacts need to be identified and managed if possible. Finally, physical impacts such as sediment transport on seagrass meadows or within potential seagrass habitats warrant examination.

ACKNOWLEDGEMENTS

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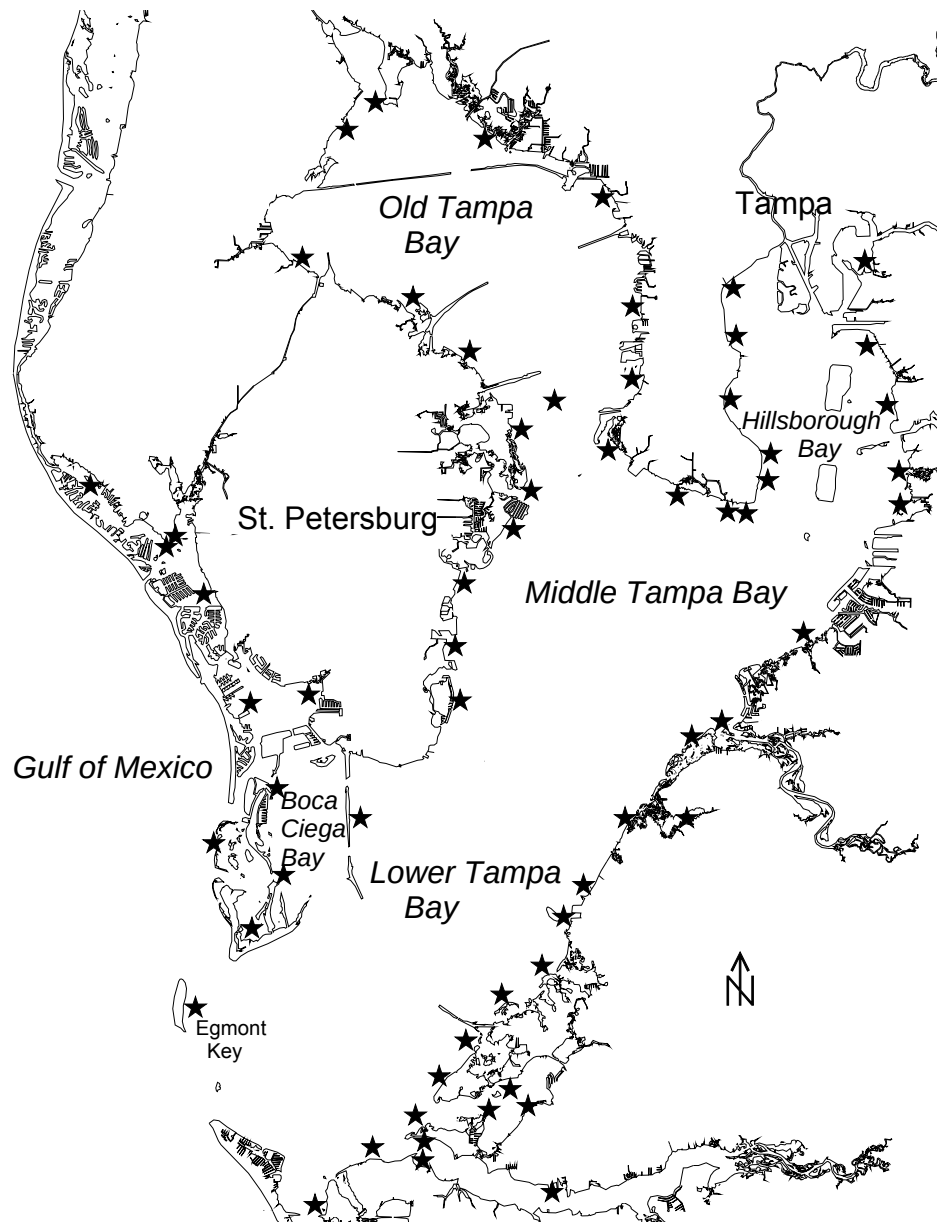


Figure 1. Location of the 62 fixed seagrass monitoring transects (★) in Tampa Bay.

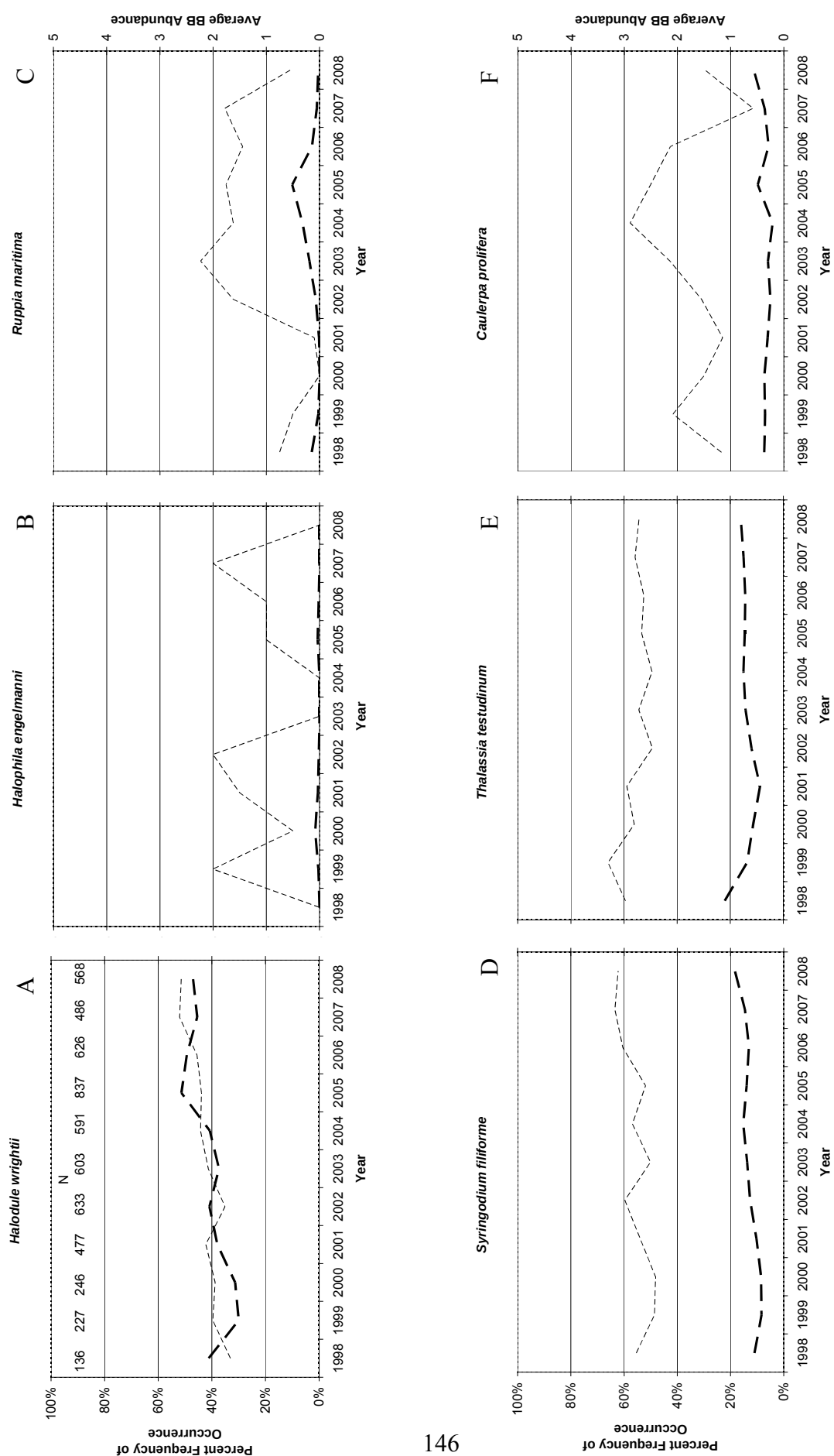


Figure 2. Percent frequency of occurrence (----) and average Braun-Blanquet (BB) abundance (-----) for Old Tampa Bay seagrass and the alga. *Caulerpa prolifera* from 1998-2008. N in graphic A represents the annual meter square placements for Old Tampa Bay

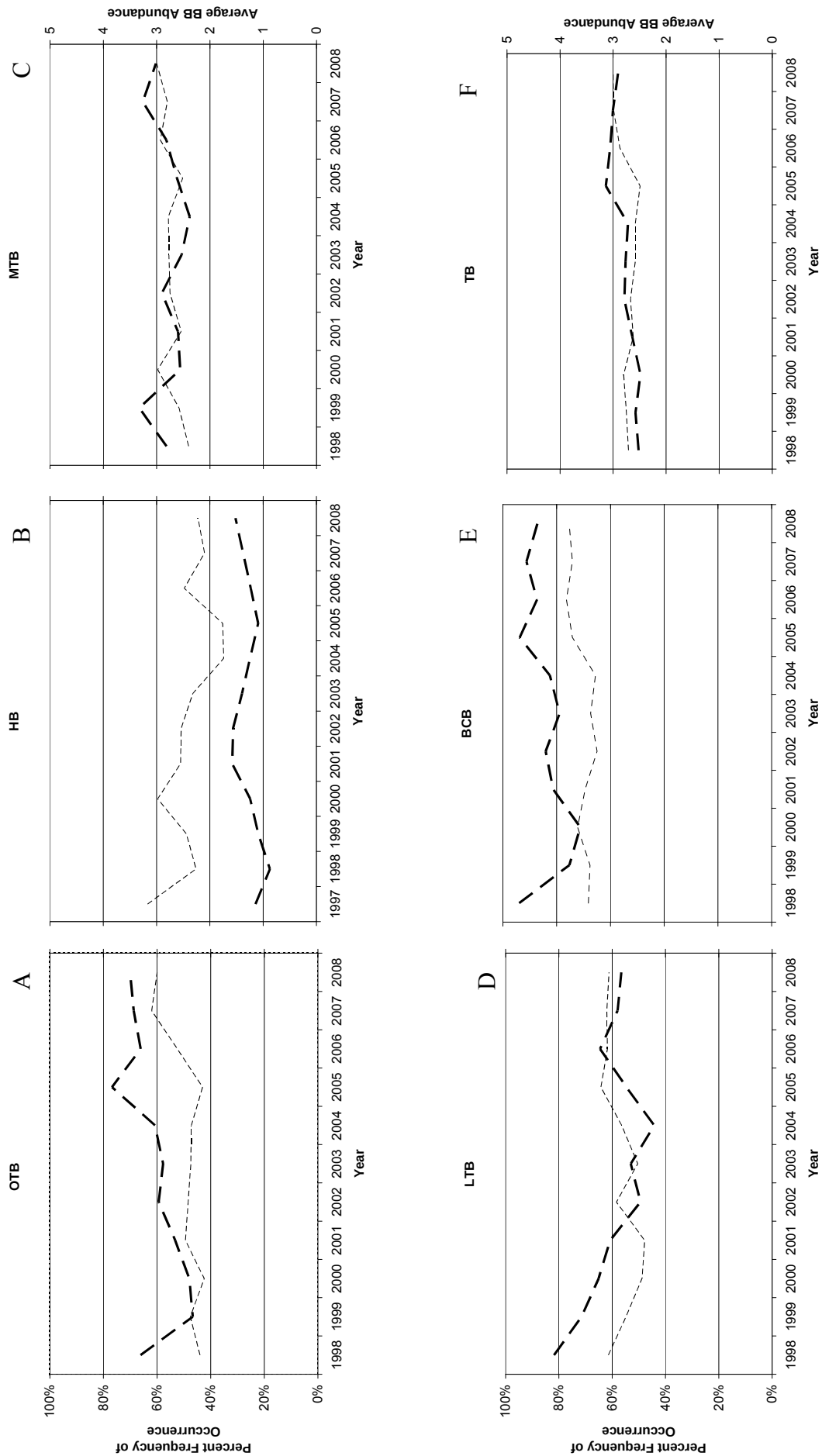


Figure 3. Percent frequency of occurrence (-----) and average Braun-Blanquet (BB) abundance (-----) for Old Tampa Bay (OTB), Hillsborough Bay (HB), Middle Tampa Bay (MTB), Lower Tampa Bay (LTB), Boca Ciega Bay (BCB), and Tampa Bay (TB) seagrass from 1998-2008.

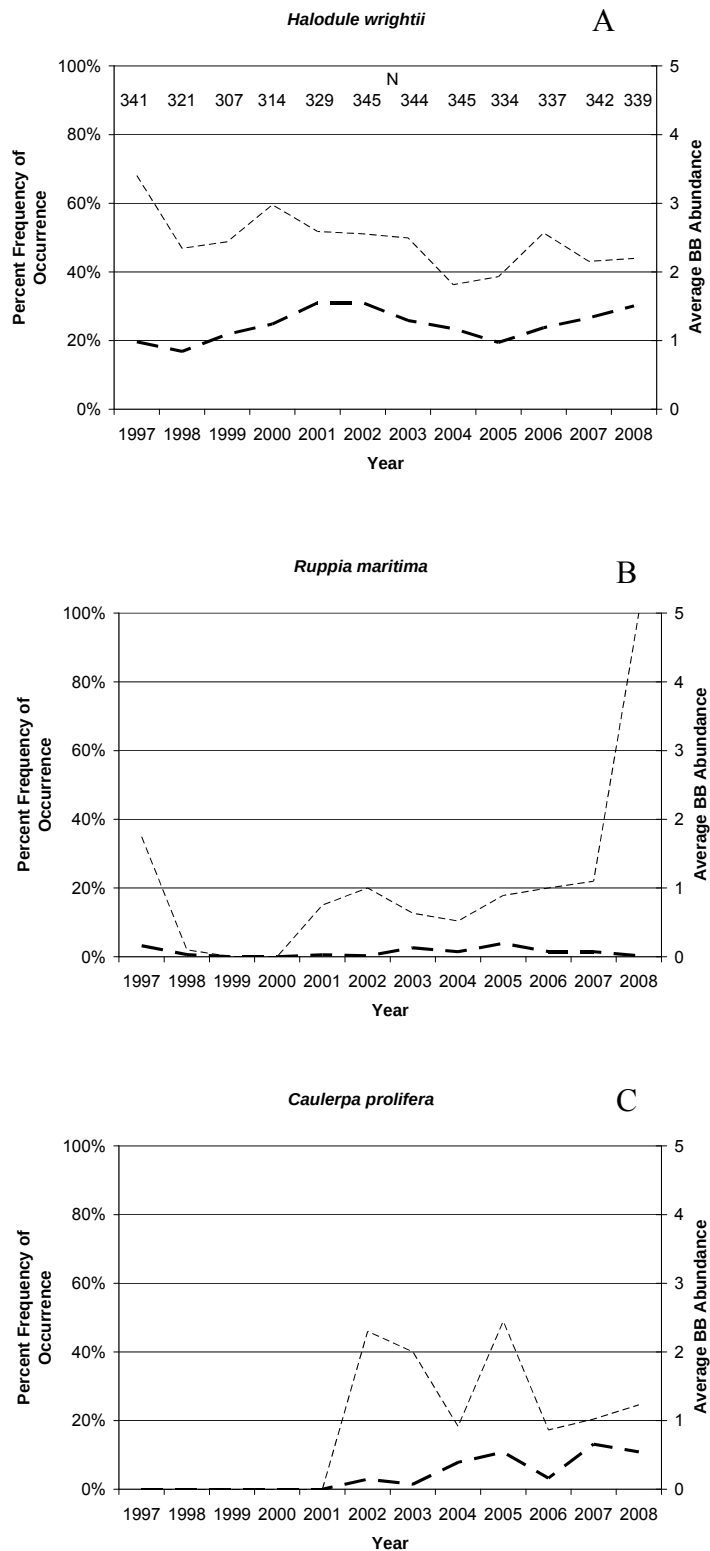


Figure 4. Percent frequency of occurrence (----) and average Braun-Blanquet (BB) abundance (-----) for Hillsborough Bay seagrass and the alga *Caulerpa prolifera* from 1997-2008. N in graphic A represents the annual meter square placements for Hillsborough Bay.

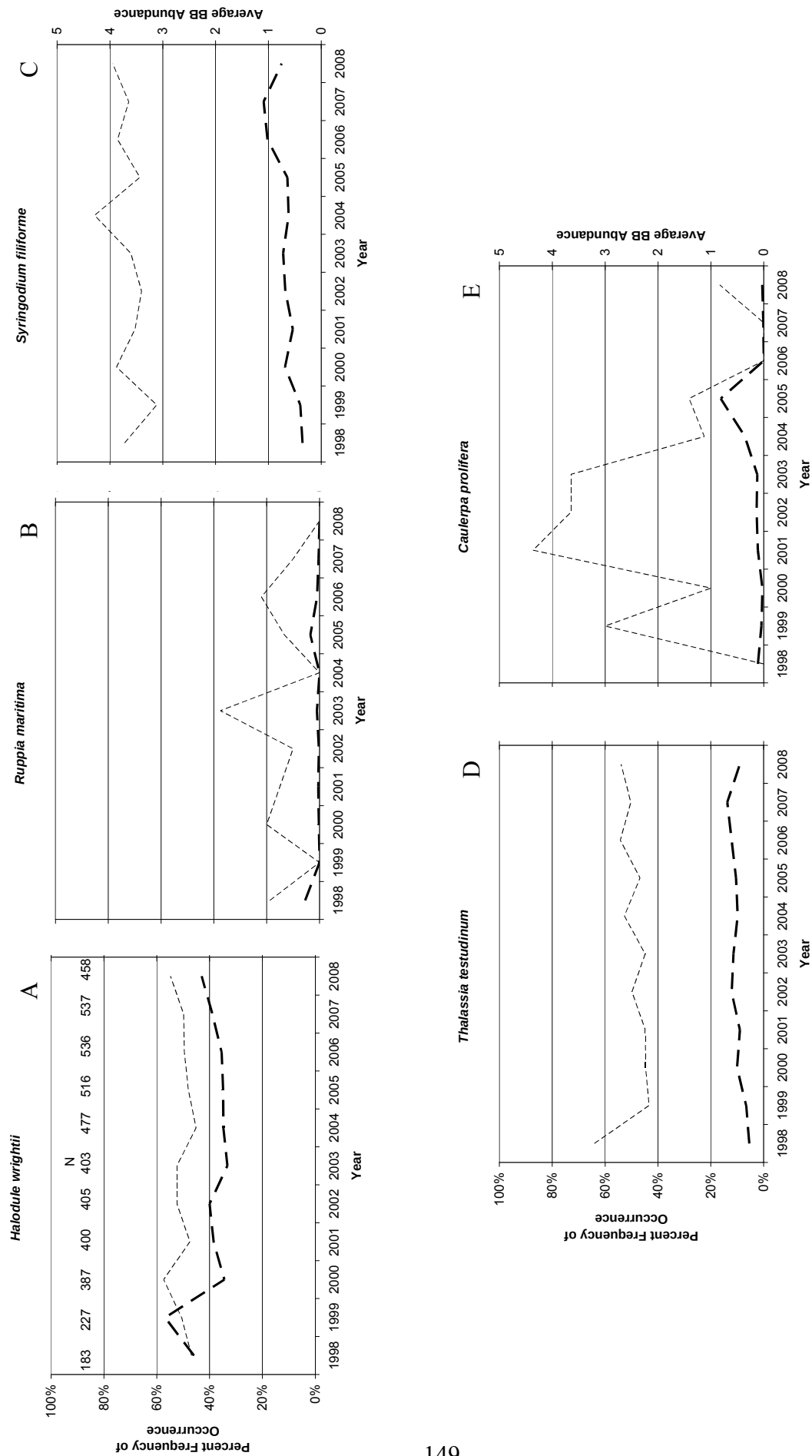


Figure 5. Percent frequency of occurrence (-----) and average Braun-Blanquet (BB) abundance (-----) for Middle Tampa Bay seagrass and the alga *Caulerpa prolifera* from 1998-2008. N in graphic A represents the annual meter square placements for Middle Tampa Bay.

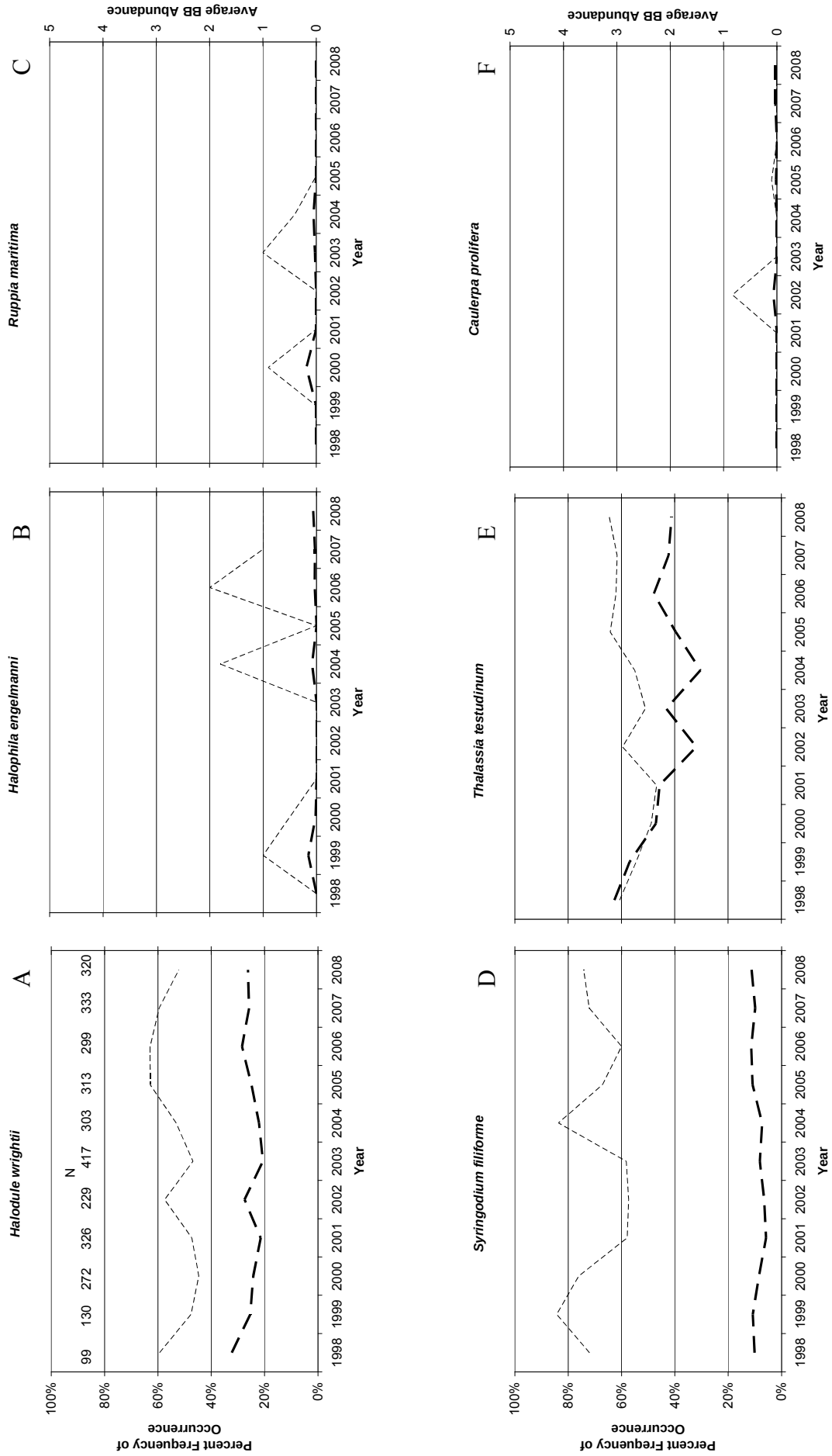


Figure 6. Percent frequency of occurrence (-----) and average Braun-Blanquet (BB) abundance (-----) for Lower Tampa Bay seagrass and the alga *Caulerpa prolifera* from 1998-2008. N in graphic A represents the annual meter square placements for Lower Tampa Bay.

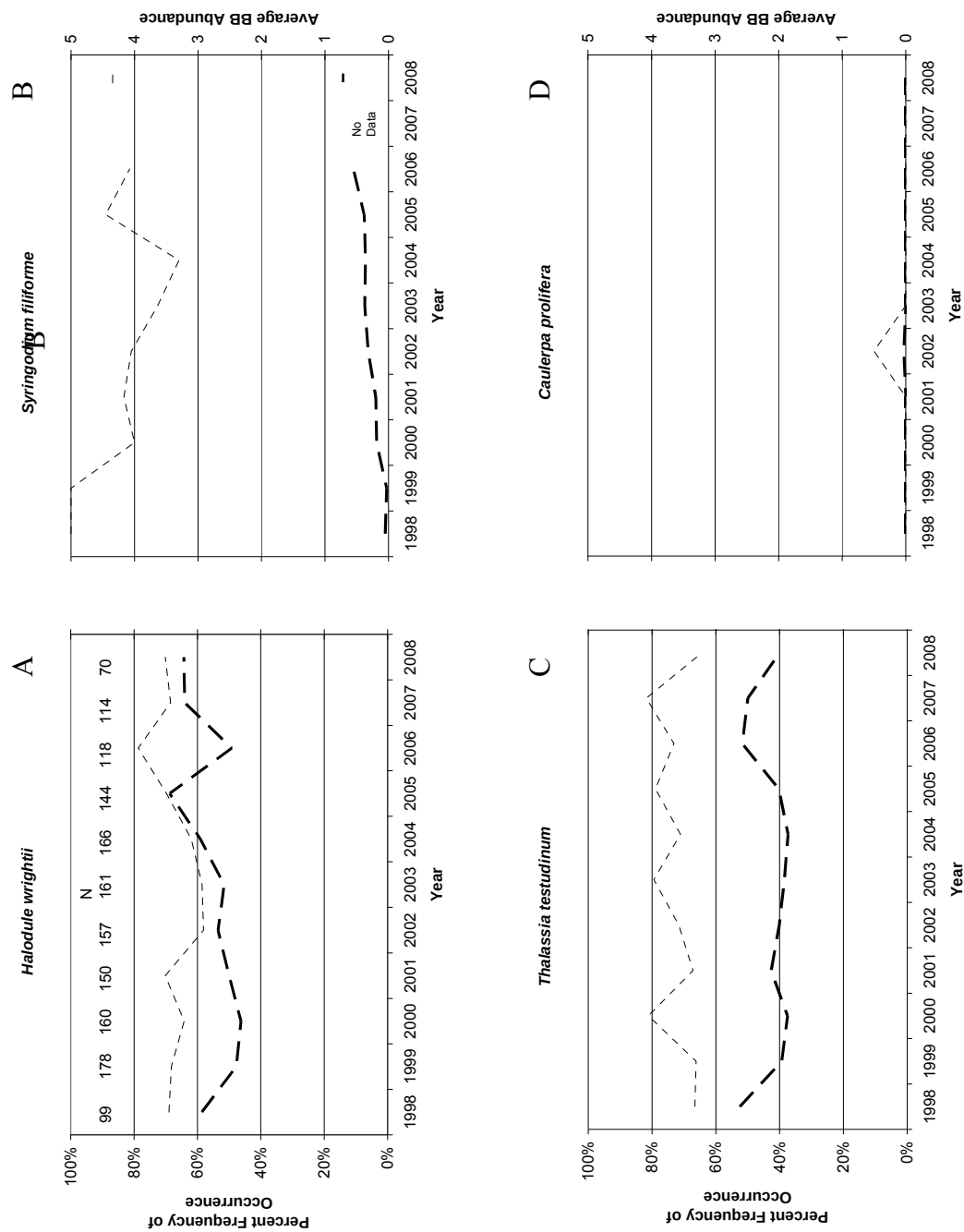


Figure 7. Percent frequency of occurrence (—) and average Braun-Blanquet (BB) abundance (-----) for Boca Ciega Bay seagrass and the alga *Caulerpa prolifera* from 1998-2008. N in graphic A represents the annual meter square placements for Boca Ciega Bay.

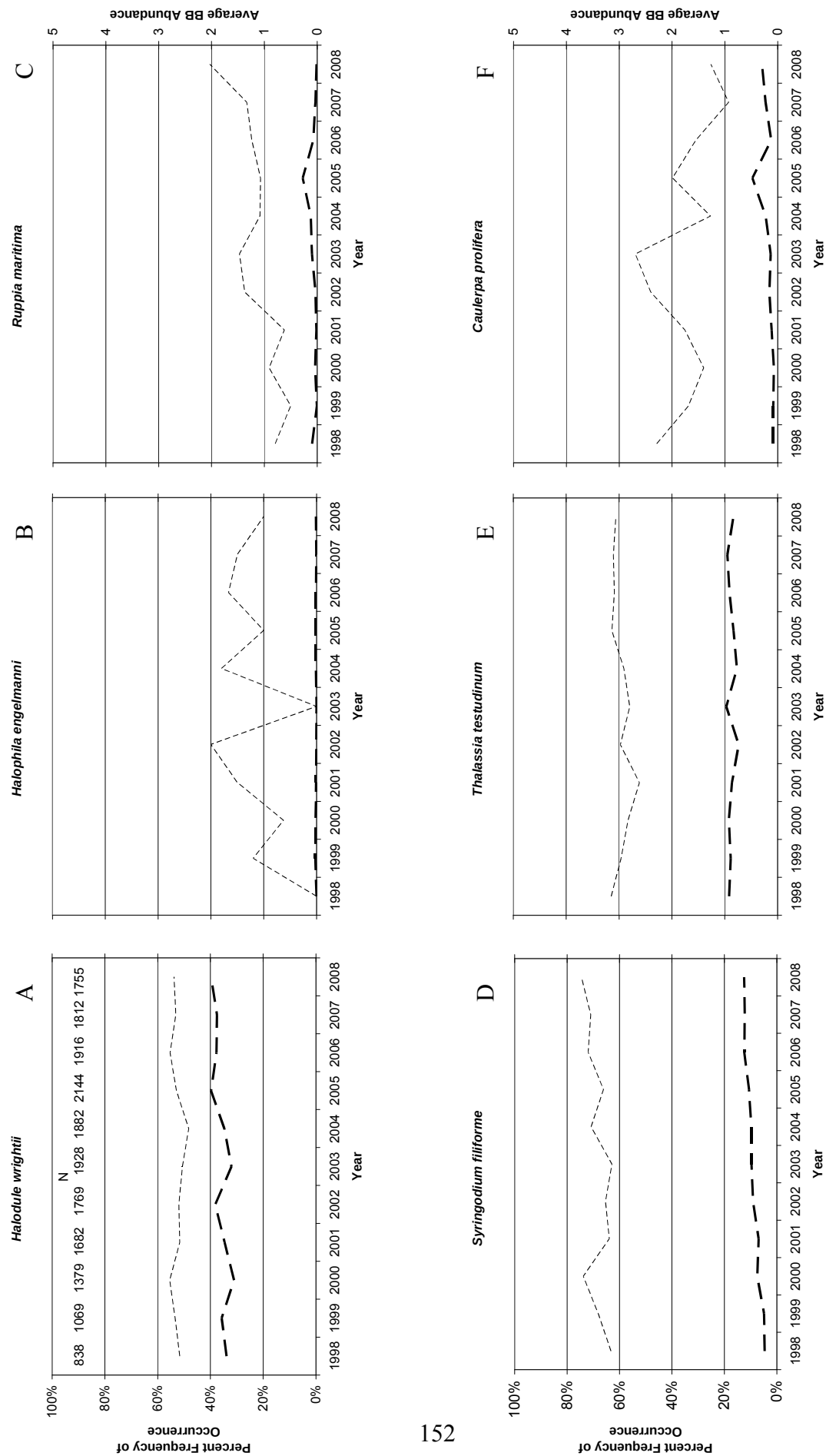


Figure 8. Percent frequency of occurrence (-----) and average Braun-Blanquet (BB) abundance (-----) for Tampa Bay seagrass and the alga *Caulerpa prolifera* from 1998-2008. N in graphic A represents the annual meter square placements for Tampa Bay.

RESTORATION OF SEAGRASS HABITAT IN TAMPA BAY USING LARGE MANATEE GRASS (*SYRINGODIUM FILIFORME*) SOD UNITS AND A DISCUSSION OF PLANTING SITE SEDIMENT ELEVATION DYNAMICS

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ABSTRACT

Extensive losses of seagrass meadows occurred during the 1960s in the off-shore portions of the shallow estuarine shelf in the upper sections of Tampa Bay. The losses most likely resulted from declining water quality and prevalent dredging activities. Considerable sediment erosion has also occurred in many areas of the bay with large seagrass losses. Over the last 25 years, water quality has improved substantially as a result of anthropogenic oligotrophication and many areas have been re-colonized by shoal grass. Re-colonization by manatee grass has generally been lacking and a planting project was initiated to restore this species to an area where it historically likely was present. Planting area bathymetry was also monitored using kinetic DGPS to evaluate sediment dynamics and to test the ability of the planted manatee grass to accumulate sediments and alleviate sediment losses. Approximately 1200 manatee grass sod units with a total area of about 48m² were harvested in summer 2006 from a permitted donor area in western Old Tampa Bay and planted in six 10x20m plots located on the offshore portion of the estuarine shelf in northern Middle Tampa Bay. After two years, the total planted manatee grass ground cover was 1340m². Growth and coalescence had occurred in all plots and several restored meadows were expanding in area coverage at rates similar to natural manatee grass meadows. Further, the above-ground biomass of the restored grass in several plots was similar to, or exceeded, that of the donor grass at the time of harvest. The two-year study did not clearly demonstrate that substantial sediment accretion was associated with the restored manatee grass meadows. However, relative fast-acting negative impacts on manatee grass persistence were clearly evident in areas of substantial sediment erosion and accretion, demonstrating that areas with more stable sediments provided the most favorable planting locations. These findings will assist in identifying suitable planting locations for future manatee grass restoration efforts in this area of Tampa Bay.

INTRODUCTION

The onset of rapidly declining water quality in Tampa Bay in the 1960s, a result of eutrophication and prevalent dredging activities, apparently resulted in extensive losses of seagrass meadows on the off-shore portions of the shallow estuarine shelf in the upper areas of the bay (Lewis et al. 1985; Johansson and Lewis 1992; Lewis et al. 1998). Historical seagrass records of these areas and current observations of similar areas down-bay, which still have thriving off-shore seagrass meadows, suggest that much of the up-bay off-shore seagrass loss that occurred nearly 50 years ago may have consisted of manatee grass (*Syringodium filiforme*) (Phillips 1962; Lewis et al. 1985).

There also appears to have been substantial erosion of long-shore sandbar formations following the off-shore seagrass meadows losses (Fonseca et al. 2002). The diminished long-shore bars were located at the outer-most edge of the shallow estuarine sand shelf (Lewis et al. 1985). The bar and seagrass losses suggest that the off-shore grasses and the long-shore bars may have coexisted in a functional association. Seagrass meadows are efficient sediment traps (Koch 2001; Seddon 2004; de Boer 2007) and it was hypothesized at the onset of this project that sediment accumulation from the historically well-developed off-shore seagrass meadows may have maintained the long-shore bar systems. When the grasses were lost as a result of the causes named above, the potential sediment/seagrass interaction became nonfunctional and the long-shore bars eventually deteriorated as a result of sediment erosion (Fonseca et al. 2002; Seddon 2004).

Water quality in Tampa Bay has improved substantially over the last 25 years as a result of anthropogenic oligotrophication (Johansson 1991; Johansson and Lewis 1992). In view of the overall bay improvements, and site-specific evaluations of current water quality conditions, the possibility to successfully restore off-shore manatee grass to several areas currently lacking such grass became apparent. Further, by selecting an area for manatee grass restoration which historically had a prominent long-shore bar present and currently is lacking such a feature, it would also be possible to examine and test the theory of the potential interrelationship between the off-shore seagrass meadows and the long-shore bars.

A suitable study area to address the two objectives of this project was identified near the deep edge of the shallow estuarine shelf near the MacDill AFB in northern Middle Tampa Bay (Figure 1). Aerial photographs from approximately 50 years ago suggest that the selected study area had extensive seagrass coverage that extended from near-shore to the deep edge of a shallow sandbar, located approximately 850m from shore. Recent aerial photography by the Southwest Florida Water Management District in 2002 and fixed seagrass transect information (Avery and Johansson 2005) indicate that current seagrass coverage only extends to about 650m from shore. Further, the outer 100m of the estuarine shelf currently lacks a prominent long-shore bar feature; instead, several smaller bars that are parallel to shore occupy the outer area.

A multi-agency work group (see below) implemented a manatee grass restoration project in the selected study area in early summer 2006. Large manatee grass sod units with sediment attached were harvested in southeastern Old Tampa Bay and then transported and planted at the restoration site near the MacDill AFB. Further, extensive measurements of sediment elevations were conducted in the study area during the seagrass transplant project.

The harvesting and planting of the manatee grass sod material was primarily conducted by Tampa Bay Watch staff with help from citizen volunteers and the City of Tampa Bay Study Group (COT-BSG) personnel. Tampa Bay Watch also evaluated the recovery of the seagrass donor sites over the two-year study period with assistance from the COT-BSG. The COT-BSG was responsible for quality control of the field activities, periodic progress monitoring of the planted manatee grass and several bathymetric surveys of the planting site. The Coastal Resources Group, Inc. was responsible for securing necessary project permits, and the preparation and submittal of required project progress reports to the permit granters. The Tampa Bay Estuary Program provided oversight and distribution of project funds. Major funding for the project was provided by the Hillsborough County Environmental Recovery Fund. All project partners provided substantial in-kind contributions.

Additionally, this report is an abbreviated version of the final project report submitted to the Hillsborough County Environmental Recovery Fund, Project 06-02 (Johansson et al. 2009). Further, the final project report includes periodic and detailed monitoring reports prepared by Kruer (2006; 2007a; 2007b; 2008a; 2008b).

METHODS

Selection of the Restoration Area—The project area selected for the seagrass restoration and sediment/seagrass interaction studies is located on the shallow estuarine shelf in northern Middle Tampa Bay near Coon Hammock Creek and the MacDill AFB (Figure 1). The site was selected based on a scientific process which considered water quality conditions, water depth, and estimated photosynthetically active radiation (PAR) availability at the sediment surface. The selection process also included comparisons of those parameters at the restoration site and at other Tampa

Bay areas with abundant manatee grass meadows, including the site selected for donor seagrass material.

Harvesting of the Donor Seagrass—Approximately 1200 large (0.20m by 0.20m) manatee grass sod units with sediment attached were harvested during June and July 2006 within a permitted, near 60ha, donor area located in south-eastern Old Tampa Bay (Figure 1). Large sod units were chosen because prior seagrass restoration efforts in Tampa Bay, and in other areas, have suggested that such material provides a high probability of survival of the planting units (Fonseca 1994; Avery and Johansson 2006; Lewis et al. 2006; Paling et al. 2007).

Harvesting was conducted in a manner designed to minimize impacts to both the sediments and the seagrass of the donor area. Collections of the sod units were spaced over nearly the entire donor area and the removal of individual units were spatially separated by about 3m. Further, harvesting was scheduled during high tides to minimize boat impacts to the area.

The seagrass material was immediately placed in transportation containers following harvesting to ensure that each collected plug was maintained as a near intact unit during handling and transportation (Figure 2). The donor site was located approximately 9km (transportation distance) from the planting site.

Evaluation of the Harvested Manatee Grass—During harvesting of the donor material, 36 manatee grass sod units were randomly selected and examined for: 1) number of apical meristems per unit, 2) number of short shoots per unit, 3) average canopy height, and 4) above and below ground biomass measured as dry weight per m^{-2} (gdwt m^{-2}). All manatee grass blades were carefully scraped of epiphytes; and the seagrass biomass material was dried for twenty-four hours at 102°C and cooled prior to weighing.

Planting Technique—The harvested manatee grass was transported by shallow draft boats, kept shaded and wet, and was planted within a few hours of harvesting at the restoration area near the MacDill AFB. The sod units were carefully removed from the transportation containers and planted level with the surrounding sediment, by hand, on 1m centers at six 10m by 20m sites (Figure 3). Approximately 200 sod units were planted at each site.

The water depth of the six planting plots ranged from about -0.70m LMSL (local mean sea level) to -0.90m LMSL. This depth range was selected because pre-planting estimates of sub-surface light conditions at the planting area indicated that more than 25 percent of incident PAR light would reach the deepest planted grass on an annual-averaged basis. Similar light availability exists in other areas of Tampa Bay that currently have healthy manatee grass meadows (Johansson 2002), including the donor area. Overall light conditions at the transplant area were, therefore, not expected to limit the survival and growth of the transplanted seagrass material.

Monitoring of the Donor Area—The recovery of seagrass and sediment loss caused by removal of the seagrass sod units in the donor area was monitored at 16 selected donor sites during the duration of the study by Tampa Bay Watch. Monitoring was conducted at the time of sod removal and about six months, one year and two years following removal. Please see Johansson et al. (2009) for detailed descriptions of donor site monitoring procedures and discussions of monitoring results.

Monitoring of the Planted Manatee Grass—The restoration sites were monitored for survival and growth of the planted manatee grass about every six months since planting, from June 2006 through September 2008 (Figure 4). In addition to the ground level monitoring events, low level helicopter oblique aerial photography of the planting sites was conducted on a near quarterly schedule. These photos were not used for quantitative determinations, but were used to illustrate the overall progress of the planted manatee grass during the course of the study (Figure 5). Please see Johansson et al. (2009) for more detailed descriptions of monitoring events.

Measurements and Monitoring of Sediment Elevation—In addition to seagrass plantings, the project also included high resolution kinetic DGPS sediment elevation measurements (see Johansson 2002) of the six planting sites and a 100m by 300m area encompassing the planting sites. Measurements were conducted just prior to planting activities in summer 2006 and about every six months following transplanting until summer 2008. To illustrate measured sediment elevations of the planting area at specific time periods, the information from each individual elevation survey was contoured using natural neighbor gridding (Surfer 7, Golden Software, Inc.) and the results were plotted as colored, shaded-relief images.

Measurements of Sediment and Seagrass Interaction—The long-term sediment elevation changes that occurred in the planting area over the two-year study period were evaluated against the two-year final manatee grass monitoring results. The purpose of the long-term examination was to seek support for the hypothesized interrelationship between sediment accumulation and well-developed seagrass meadows.

Further, short-term sediment elevation changes that occurred in the planting area during the six month period between winter 2008 and summer 2008 were also examined and evaluated against manatee grass area expansion or contraction. It was anticipated that a detailed examination of the short-term period would provide instructive insight on potential interactions between sediment gains and losses and the planted manatee grass coverage.

RESULTS

Recovery of the Donor Area—Seagrass and sediment recovery at the 16 designated sod unit collection sites in the donor area were initially evaluated by Tampa Bay Watch in August 2007, about one year following the harvest. Those results indicate that no visible or measurable impacts to manatee grass coverage or sediments were evident at the donor site one year following harvesting. It was concluded that all 16 collection sites had successfully and fully recovered within one year.

It was anticipated, prior to harvesting, that the donor sites would recover relatively quickly following the impacts caused by harvesting because: 1) the seagrass meadows in the donor area appeared stable and healthy at the time of harvesting; and 2) the harvesting methods were specifically designed to minimize impacts to the surrounding seagrass and sediments. Similar relatively quick recovery of manatee grass donor sites in Florida have been reported by Fonseca et al. (1994).

Growth and Horizontal Expansion Rate of the Planted Manatee Grass—The six manatee grass restoration plots were surveyed in July 2008 for two-year progress in area coverage growth following the plantings in summer 2006. This survey determined area coverage and seagrass condition within the perimeters of the planting plots. The amount of manatee grass that was originally planted in each of the six plots, in terms of area coverage, was approximately 8m² for each plot, or a total of about 48m². In July 2008, approximately two years following plantings, the

total estimated cover within the 20m by 10m perimeter of the six planting plots was estimated at 600m², which equals an overall average of 50% area coverage in the six plots. However, manatee grass coverage among the six plots varied greatly, but all plots showed a substantial increase in coverage after two years.

The first evaluation of density and canopy height of the planted manatee grass was conducted in late July 2006. At that time the seagrass at the six restoration plots was relatively dense, on average about 1180 short-shoots/m², and had a canopy height at about 0.18m. In the July 2008 survey, the respective values for these parameters were about 1229 short-shoots/m² and 0.36m. Similar to variations in ground cover between the six plots, there were also large variations between plots in these parameters.

In addition to the random surveys, detailed DGPS ground cover area mappings were conducted in January and September 2008. These measurements included all manatee grass growing within the 10m by 20m plot perimeters and also the grass that had expanded beyond the perimeters. Therefore, the mapping effort provided an accurate evaluation of the total amount of manatee grass restored over the two-year project period. The final DGPS survey in September 2008 showed that the total manatee grass coverage for the six plots was 1340m² (0.33acres), which equals a 28 times increase in manatee grass cover over the approximately two-year study period. There were variations in coverage between the six plots after two years; however, all sites showed a substantial increase (Figure 6).

The September 2008 DGPS mapping survey also included manatee grass short-shoot density and canopy height measurements (Figure 6). The short-shoot measurements showed that all plots had relatively dense grass two years following plantings, but that the values were somewhat lower than that of the harvested material. Canopy height for all six plots, on the other hand, substantially exceeded the height of the grass that was harvested. The two measures of above-ground biomass suggest that the restored manatee grass had relatively high biomass and that its above-ground biomass per unit area was similar to that found at the donor site.

In addition, the detailed perimeter surveys of the planted manatee grass also showed that the area covered by manatee grass in Plots 5 and 6, the two most near-shore plots, had expanded substantially over the eight-month period between January and September 2008 (Figure 7). During the eight-month period, the outer perimeter of the grass in both plots had extended about 1m in most directions. The 1m spreading of the planting plots over this period was equal to a patch edge expansion rate of about 0.004m day⁻¹, or about 1.5m yr⁻¹.

Sediment Elevation Dynamics—The sediment elevation, or water depth at local mean sea level, of the six planting plots ranged from about -0.70m LMSL to -0.90m LMSL at the time of planting. Comparisons of sediment elevation measurements from the start of the study in June 2006 to the end in July 2008 show that substantial sediment perturbations occurred in the outer-most, and generally deeper, half of the planting area (Figure 8 and 9). Seagrass Plots 1, 2 and 3 were planted in this area. Changes in sediment elevation in the off-shore area ranged from some areas becoming 0.20m deeper to others becoming 0.25m shallower. The largest elevation changes were associated with the development or movement of sandbars with amplitudes reaching 0.40m. Several sandbars were present throughout the study period in the outer-most section of the planting area. These bars were located parallel with the deep off-shore edge of the shallow estuarine shelf, but were approximately 50m to 100m inshore of the location of the historical prominent off-shore bar.

The shallower and more near-shore half of the planting area had smaller sediment perturbations (Figure 8 and 9). Seagrass Plots 4, 5, and 6 were planted in this area. Changes in sediment elevation in the near-shore area over the two year study period ranged from some areas becoming 0.15m deeper to others becoming 0.15m shallower. Only relatively small sandbars were present in this area during the study period and, consequently, the relief of the area was less uneven than the off-shore half. The near-shore area was located landward of the series of sandbars in the outer half of the study area and may have been relatively well shielded from wave energy (see Fonseca et al. 2002; Lewis 2002; Dai et al. 2008).

DISCUSSION

Performance of the Manatee Grass Restoration Effort—The final project monitoring results from the donor area and the restoration area attest to a successful completion of the manatee grass restoration effort. First, the donor site monitoring indicated that disturbances caused to the donor site by harvesting activities were fully mitigated within the two-year study period. This was postulated to occur based on results from other harvesting projects conducted in similar stable and healthy seagrass meadows (see Fonseca 1994). Second, the restoration of about 1300m² of manatee grass, two years following plantings, in an area previously devoid of this species clearly indicates that the seagrass transplanting portion of the project was successful. Third, at the end of the study period the per-unit area above-ground biomass of the restored manatee grass in several of the planting plots was similar to, or may have exceeded the biomass of the donor grass at the time of harvest. Finally, several of the restored meadows were actively expanding in area coverage at a rate similar to natural-growing manatee grass meadows.

The positive outcome of the restoration project may have been linked to the application of the following techniques and approaches:

1. The efforts to minimize impacts, to the greatest extent possible, to both the sediments and seagrass of the donor area during harvesting most probably aided in the successful and quick recovery of the donor site within the two-year study period. The quick recovery was evidenced by a lack of visible or measurable impacts to seagrass coverage and sediments one year following the sod removal.
2. The restoration site was selected based on a scientific selection which considered water quality conditions, water depth and estimates of PAR light availability at the sediment surface. This step also included comparisons of water quality and submarine light conditions between the restoration site and other Tampa Bay areas with abundant manatee grass meadows, including the selected donor area.
3. The donor material was carefully evaluated, collected and handled to ensure high survival rate and growth potential. Further, the relatively large planting units, with the natural sediment matrix attached, contained a substantial amount of intact manatee grass biomass with numerous apical meristems.
4. The harvested manatee grass was kept shaded and wet, and were placed in transportation containers to maintain intactness during handling and transportation. The sod units were handled carefully during planting and were planted, by hand, at the same level as the surrounding sediment to minimize effects of sediment scouring. All harvested material was planted within a few hours of harvesting.

5. Planting occurred during early and mid summer and prior to the onset of the annual rainy period. This time window may have allowed the newly planted manatee grass a brief, but potentially important period of relatively calm weather and good underwater light climate to recover from relocation impacts and to acclimatize to the new environment.

6. Finally, no devices were employed at the planting plots to deter bioturbation or encroachment of drift algae. These tools may at times accomplish their intended purposes (see Fonseca 1994); however, it has also been observed that such devices may interfere with the growth and survival of the planted grass (Johansson pers. obs.). For example, they may disrupt the natural flow of water and sediments at the planted sites, and they may attract both animal and vegetation fouling organisms that can shade or in other ways retard growth. Also, the devices could potentially reduce bioturbation activities that may enhance seagrass recruitment to the area (see Fonseca et al. 2008). As discussed below, bioturbation did not appear to have had a substantial impact on the survival and growth of the unprotected planted manatee grass sod units in the current project. Paling et al. (2007) also found that planted seagrass sod units had relatively low susceptibility to bioturbation.

Planting Patch Expansions and Anticipated Future Progress of Growth—The measured expansion rate for the patch edge of the planted manatee grass in Plots 5 and 6 was about 1.5m yr^{-1} . This rate is within the 0.5m yr^{-1} to 1.82m yr^{-1} rate that has been reported for horizontal rhizome elongation of manatee grass in a wide range of environments (Marba and Duarte 1998; Kendall et al. 2004; Cabaço et al. 2008). Further, Fonseca et al. (2004; as discussed in Kendall et al. 2004) calculated that edges of naturally growing manatee grass patches in the Florida Keys advanced at 1.23m yr^{-1} , which is within the range of the reported rhizome elongation rate and very similar to the measured rate for edge of patch expansion in the current study.

The measured expansion rate for the edge of the planted manatee grass in Plots 5 and 6 suggests that these plots will coalesce in early 2010, assuming that the current rate remains constant. Coalescence may also occur between these plots and Plot 4 in that time frame, however, any further coalescence between plots appears unlikely in the near future due to the greater distance between the remaining plots and their apparent slow expansion rate.

The most recent area measurements indicate that all plots, except Plots 2 and 3, have developed substantial area coverage and biomass amounts, and could be expected to withstand, at least temporarily, detrimental conditions such as reduced light availability caused by degraded water quality conditions. Plots 2 and 3 experienced losses of area coverage during the last six months of the project and the losses, specifically in Plot 3, were considerable (Figure 9). The recent losses make the future outlook for these two plots uncertain. Later it will be shown that these plots were established in the most physically dynamic section of the planting area. Forces such as wave turbulence and sediment scouring and accretion, could be expected to be more detrimental to seagrass growth in this dynamic area than in more quiescent areas. Persistent exposure to these forces may cause additional seagrass losses in Plots 2 and 3 and the eventual demise of the planted manatee grass.

Interactions between the Sediment and the Planted Manatee Grass—Seagrasses have been identified as ecosystem engineers because of their ability to change their own environment (Koch 2001). Specifically, their ability to stabilize and accumulate sediments by reducing water currents and turbulence over the meadows is generally recognized to be beneficial to the grass by stimulating growth and decreasing mortality from erosion (see reviews by Koch 2001; Boer 2007). Other important ways the seagrasses may affect their environment are also discussed in those articles.

The manatee grass restoration project addressed the seagrass/sediment interaction process in a two-pronged approach. First, sediment elevation changes that occurred in the manatee grass planting area over the two-year study period were compared to the two-year and final seagrass monitoring results. This step attempted to determine the long-term ability of the planted manatee grass to accumulate sediments within the planting plots or in areas just shoreward of the plots. Second, more short-term seagrass/sediment interactions that occurred over an approximately six- to eight-month period between January 2008 and summer 2008 were also examined. The purpose of the short-term analysis was to provide insight on the ability of the planted manatee grass to withstand sediment burial and erosion.

Long-term Interactions: Long-term sediment accumulation within and near the planted manatee grass plots were not readily apparent at most plots, possibly suggesting that the two-year study period was not sufficiently long to clearly demonstrate a relationship between well-developed manatee grass meadows and sediment accumulation. Four of the six planting plots showed large expansions in ground cover and had dense and tall manatee grass after two years of growth (Figure 6), however, only planting Plot 5 had substantial sediment accretion that may have been associated with the manatee grass. Sediment accumulation ranging from about 0.10m to 0.15m was measured within Plot 5 and just shoreward of the planted area. Sediment accumulation also occurred near several other plots. It is, however, difficult to associate this latter sediment accumulation with the planted manatee grass because areas near these plots, but outside the expected influence of the manatee grass, also had similar or greater sediment accumulation. Shoal grass (*Halodule wrightii*) was abundant outside most plots, specifically towards the end of the two-year study period. However, this species has relatively narrow and short blades and would not be expected to have a large influence on sediment accumulation (see Mellors et al. 2002).

The substantial changes in sediment elevations which occurred in the outer-most, and generally deeper, half of the seagrass restoration area over the two year study period were most likely caused by off-shore generated waves impacting the outer edge of the estuarine shelf. The physical impact of waves in this area may have hindered the planted manatee grass to reach the high densities and area coverage attained by the more near-shore and protected plots (see Fonseca and Bell 1998; Koch 2001; Fonseca et al. 2002; Seddon 2004; Cruz-Palacios et al. 2005). Planting Plots 2 and 3 were located in this dynamic section of the planting area (Figure 9). These plots had the lowest manatee grass area coverage of all plots at the end of the study and also low above-ground biomass (Figure 6). Further, both plots became overall deeper over the two-year study period with sections of the plots losing up to 0.20m of sediments.

The wave-generated turbulence present in the outer-most planting area may have caused numerous impacts to the planted manatee grass, including loss of leaf canopy, sediment accretion that smothered the grass, erosion that dislodged the grass from the sediment and also high-turbidity events that temporarily may have impacted light availability (see Koch 2001; Kendall et al. 2004; Cabaço et al. 2008). It is unlikely that the relatively low values of above-ground biomass and area coverage observed in this dynamic area resulted from bioturbation or man-made local and smaller-scale physical disturbances. In addition, no obvious effects of bioturbation were noted in the very dense and tall manatee grass growing in the more protected plots in the near-shore planting area.

Short-term Interactions: Short-term seagrass/sediment interactions that occurred between January 2008 and summer 2008 were also apparent in planting Plots 2 and 3 in the outer-most and generally deeper portion of the planting area. During this period, both plots had sections with substantial sediment losses, ranging from about 0.10m to 0.20m, in which all manatee grass was lost (Figure 9). In a comprehensive review of short-term impacts on seagrass from experimental

sediment burial and erosion Cabaço et al. (2008) reported that manatee grass suffered 50% mortality at a 0.045m erosion level over 60 days. That level of erosion was greatly exceeded in sections of Plots 2 and 3 where 100% manatee grass loss occurred during the short-term period. Cabaço et al. (2008) also reported that “deep” erosion of 0.10m, which exceeded the anchoring depth of the manatee grass, did not produce a total loss of grass in the laboratory experiments. This unexpected finding may have resulted from the reviewed experiments being of a relatively short duration. Total loss of manatee grass could be expected when the anchoring depth of the grass is exceeded for an extended period; which appeared to have occurred in sections of Plots 2 and 3.

In contrast to the total loss of planted manatee grass in areas of erosion in Plots 2 and 3, a total loss of manatee grass also occurred during the short-term period in a large area of the near-shore portion of Plot 3 where sediment accumulated from about 0.05m to 0.20m. Cabaço et al. (2008) reported that an experimental burial level of 0.10m over 60 days caused 100% mortality of manatee grass. The amount of sediment accretion in Plot 3, where total loss of manatee grass occurred, was similar to or exceeded 0.10m. Therefore, a near 100m² large loss of manatee grass that occurred between January and June/July 2008 in the near-shore section of Plot 3 appears to have been caused by sediment burial. Specifically, the shallowing of the near-shore portion of Plot 3, and the seagrass losses, may have resulted from the apparent movement in the off-shore direction of a large sandbar located between Plots 3 and 4. There are several similar observations of seagrass burial and losses by migrating sand waves referenced in the literature (see de Boer 2007; Cabaço et al. 2008), including observations from Hillsborough Bay by Avery and Johansson (2006).

Considerations for Future Planting Projects—An important finding of the planting project was the identification of an apparent favorable planting location or “sweet spot” for manatee grass restoration in the selected study area. The water depth of Plots 4, 5 and 6, where the highest above-ground biomass developed, ranged from about -0.60m LMSL to -1.00m LMSL. This depth range effectively encompassed the depth range of all planted plots. However, Plots 4, 5 and 6 were located in the near-shore half of the planting area and landward of the series of sandbars in the outer half of the study area. As a result, these plots may have been relatively well-shielded from off-shore generated waves. The observation that relatively minor sediment perturbations occurred in the near-shore area during the project supports that conclusion. The discovery of the apparent favorable planting location for manatee grass restoration in the study area appears to have resulted from the selection of both appropriate water depth and sufficient protection from wave turbulence.

CONCLUSIONS

The manatee grass transplanting project near the MacDill AFB in Middle Tampa Bay, conducted from summer 2006 through summer 2008, was completed successfully. First, the donor site monitoring indicated that disturbances caused to the donor site by harvesting activities were fully mitigated within the two-year study period. Second, the restoration of about 1300m² of manatee grass, two years following plantings, in an area previously devoid of this species clearly indicates that the seagrass transplanting portion of the project was successful. Third, at the end of the study period the per-unit area above-ground biomass of the restored manatee grass in several of the planting plots was similar to, or may have exceeded the above-ground biomass of the donor grass at the time of harvest. Finally, several of the restored meadows were actively expanding in area coverage at a rate similar to natural-growing manatee grass meadows.

Although the two year study did not clearly demonstrate a strong relationship between well developed manatee grass meadows and sediment accumulation, several important observations were made of interrelationships between sediment dynamics and manatee grass growth and

survival. Also, the sediment elevation surveys conducted over the two-year period clearly indicated that the off-shore half of the planting area was very dynamic in terms of sediment perturbations and that the inshore half was less so. These findings assisted in the identification of favorable planting location criteria to be applied in future manatee grass restoration efforts in this area of Tampa Bay.

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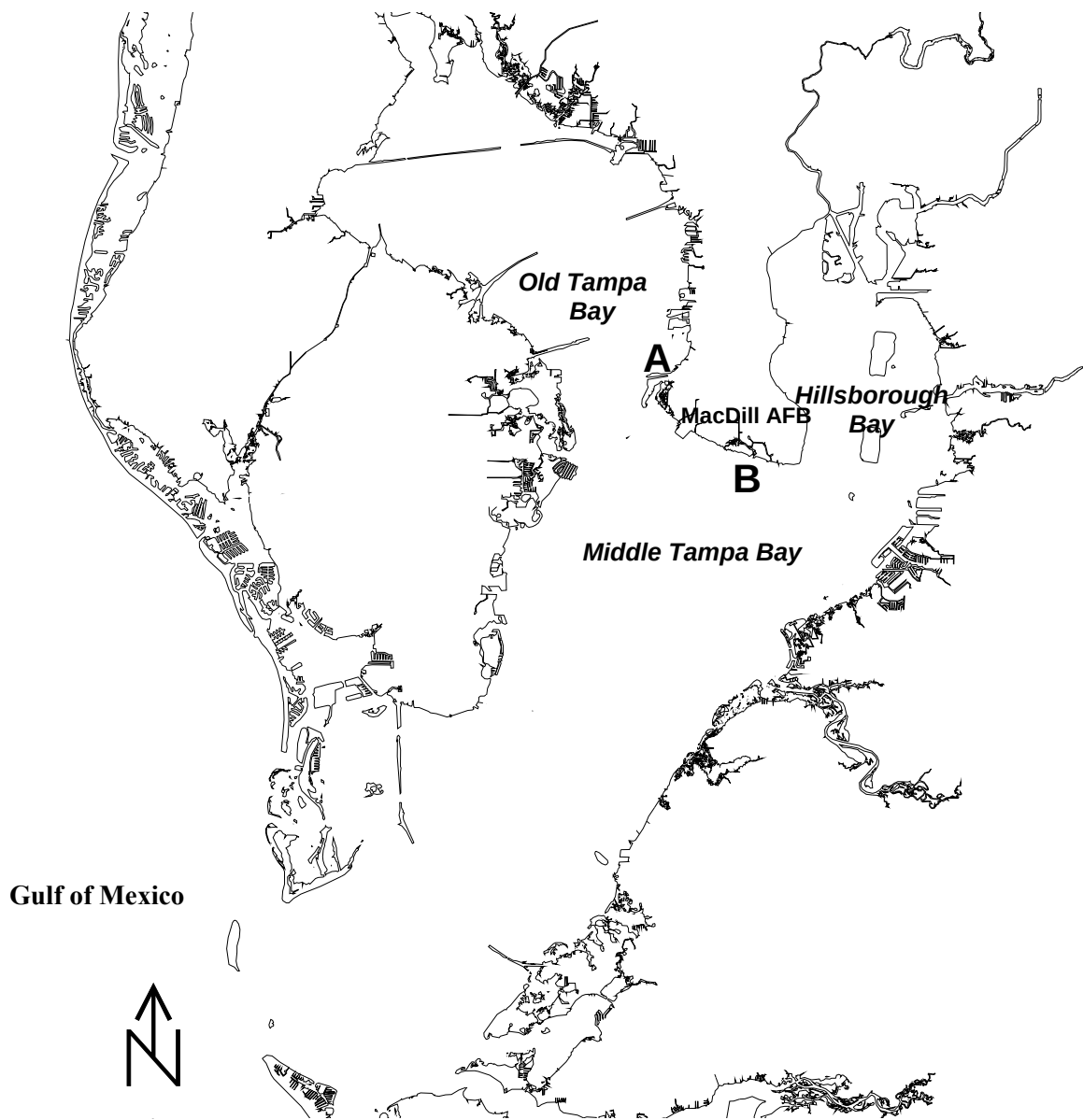


Figure 1. Location of the manatee grass (*Syringodium filiforme*) donor site in eastern Old Tampa Bay (A) and the planting area near MacDill AFB in upper Middle Tampa Bay (B).



Figure 2. Harvesting of manatee grass sod units, showing (A) stainless steel collection device and WAAS corrected GPS recorder, and (B) sod transportation container.



Figure 3. (A) Planting of manatee grass sod units. (B) Underwater photo of a freshly planted unit.



Figure 4. Monitoring of the planted manatee grass sod units. (A) Measurements of percent seagrass coverage and (B) area coverage.

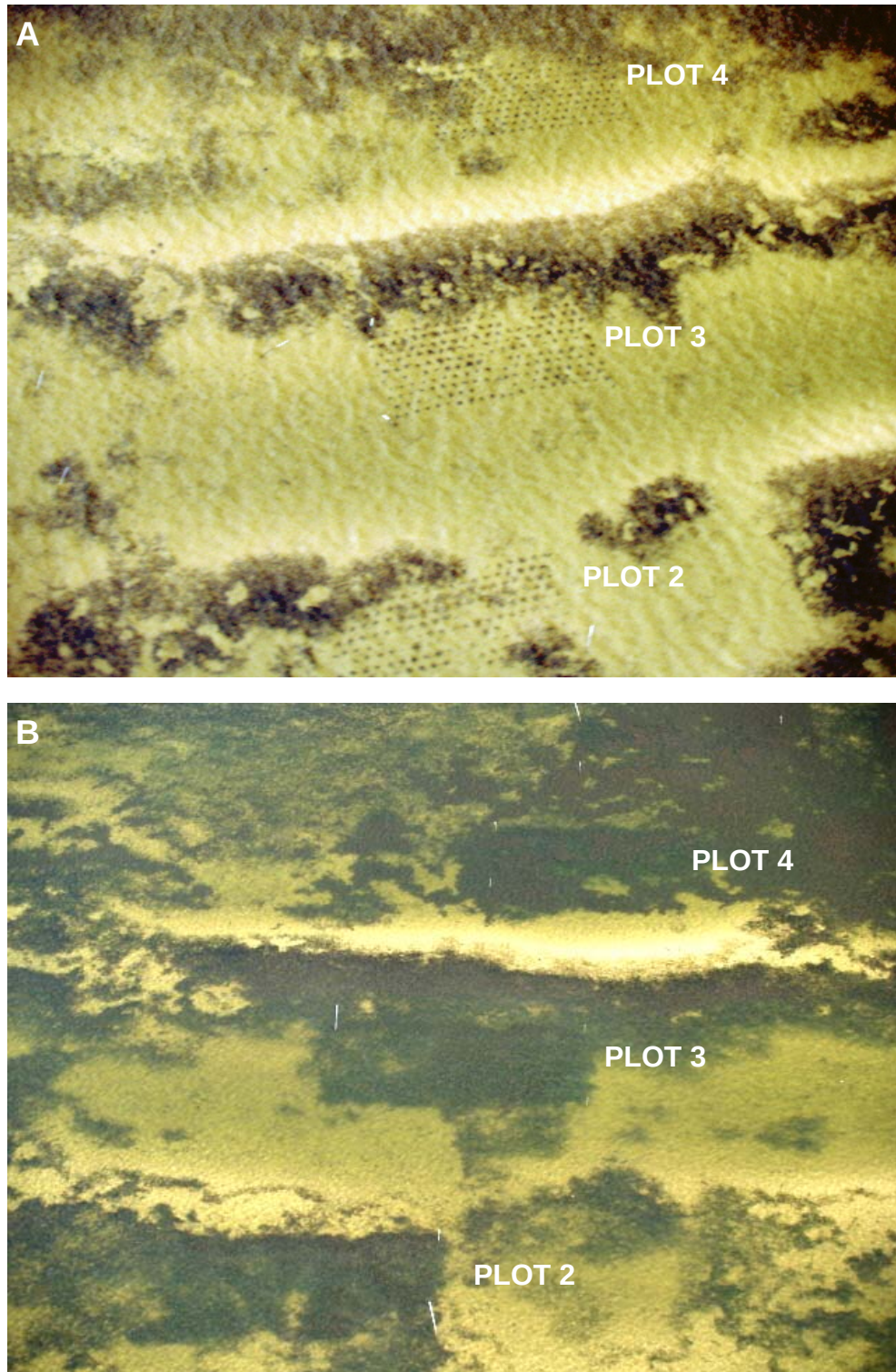


Figure 5. Oblique aerial photographs of planting Plots 2, 3 and 4 taken on (A) November 2006 and (B) November 2007.

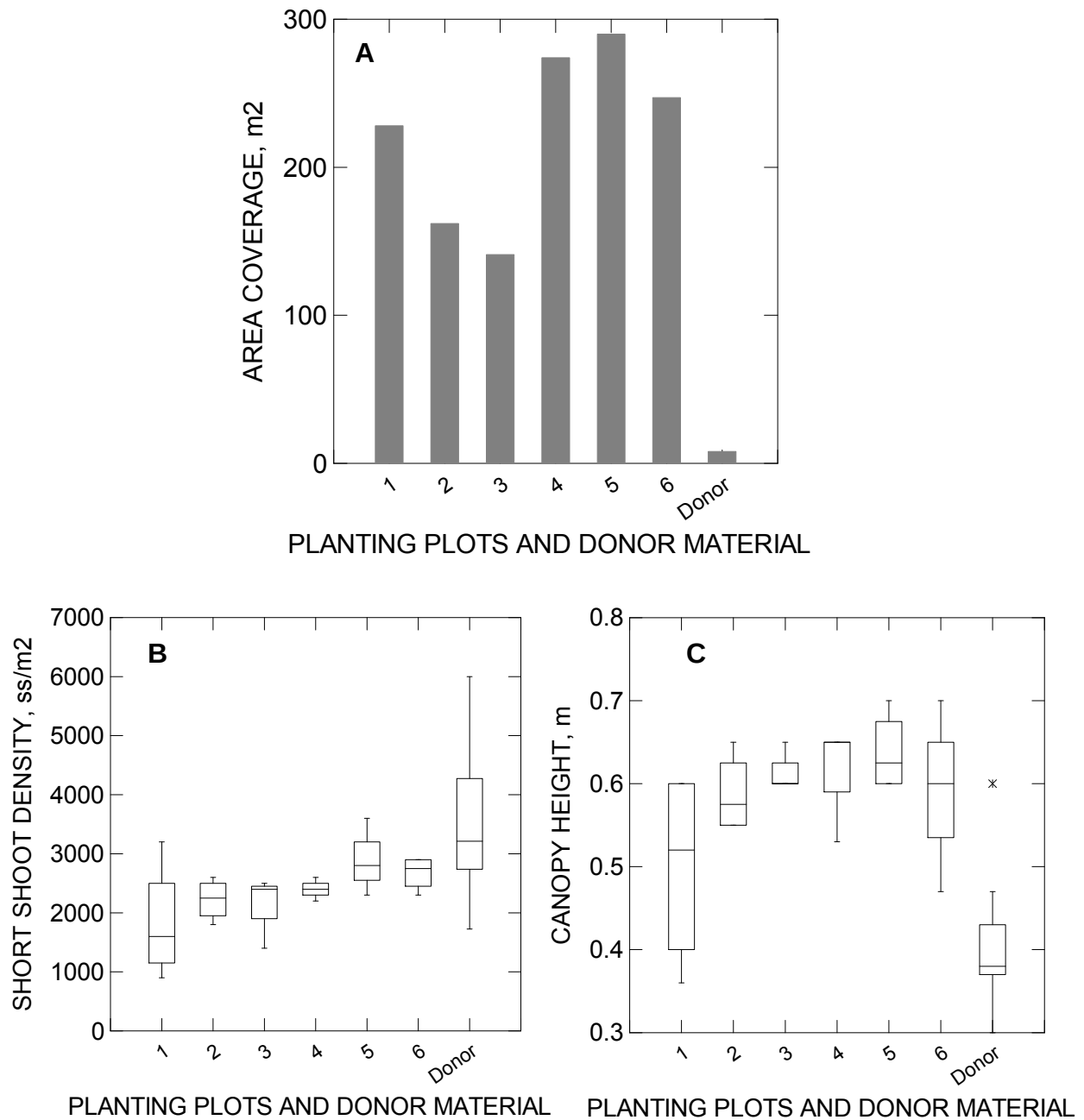


Figure 6. Comparison of results from the September 2008 monitoring of seagrass characteristics for the planted manatee grass at each planting plot and the initial characteristics of the donor material: (A) area coverage (please note that all plots initially had 8m² of total coverage), (B) short shoot density and (C) canopy height.

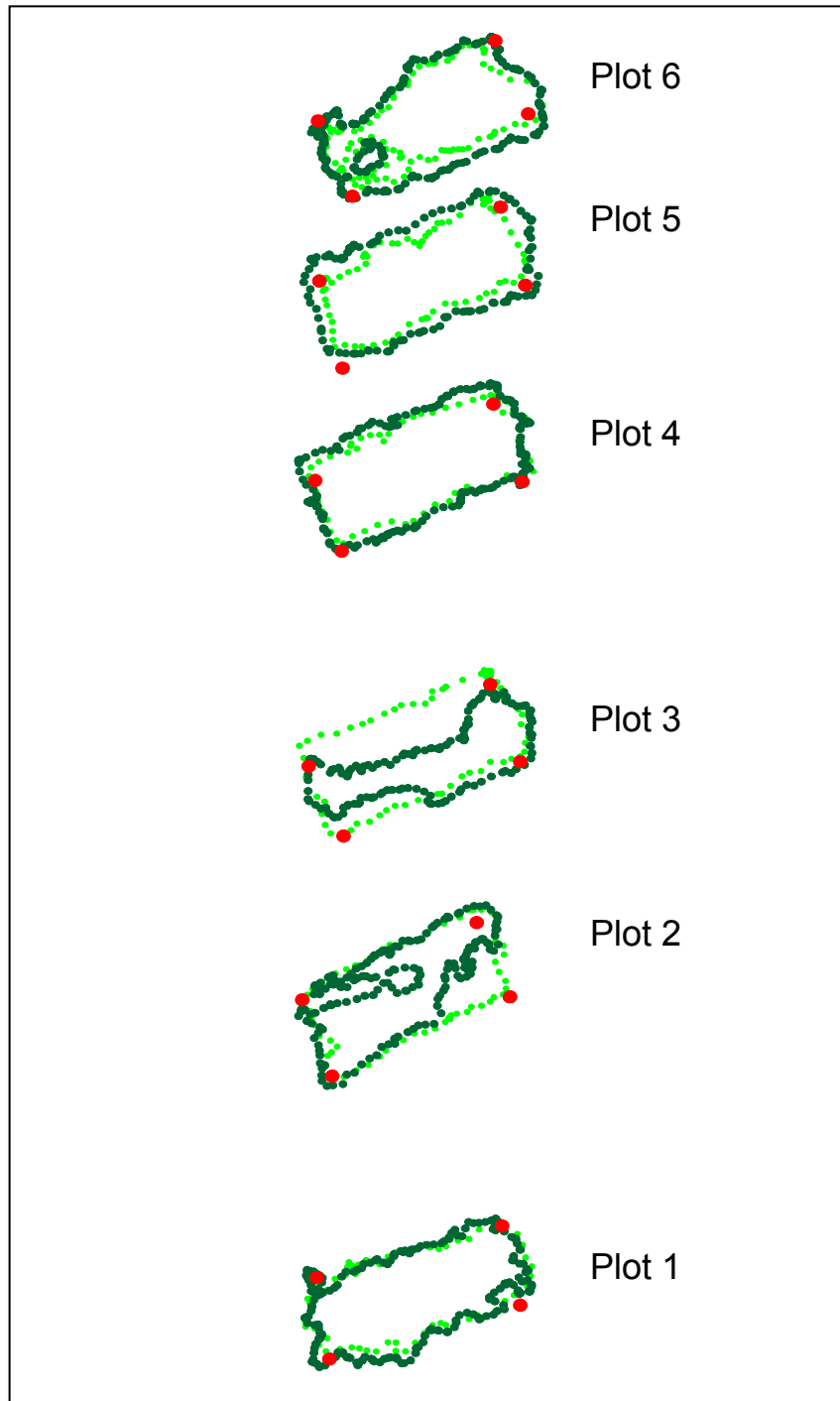


Figure 7. Perimeter surveys of manatee grass area coverage at the six planting plots conducted in January (light green symbols) and September 2008 (dark green symbols). The planting plot corner posts are also shown. The short sides of the plots are 10m.

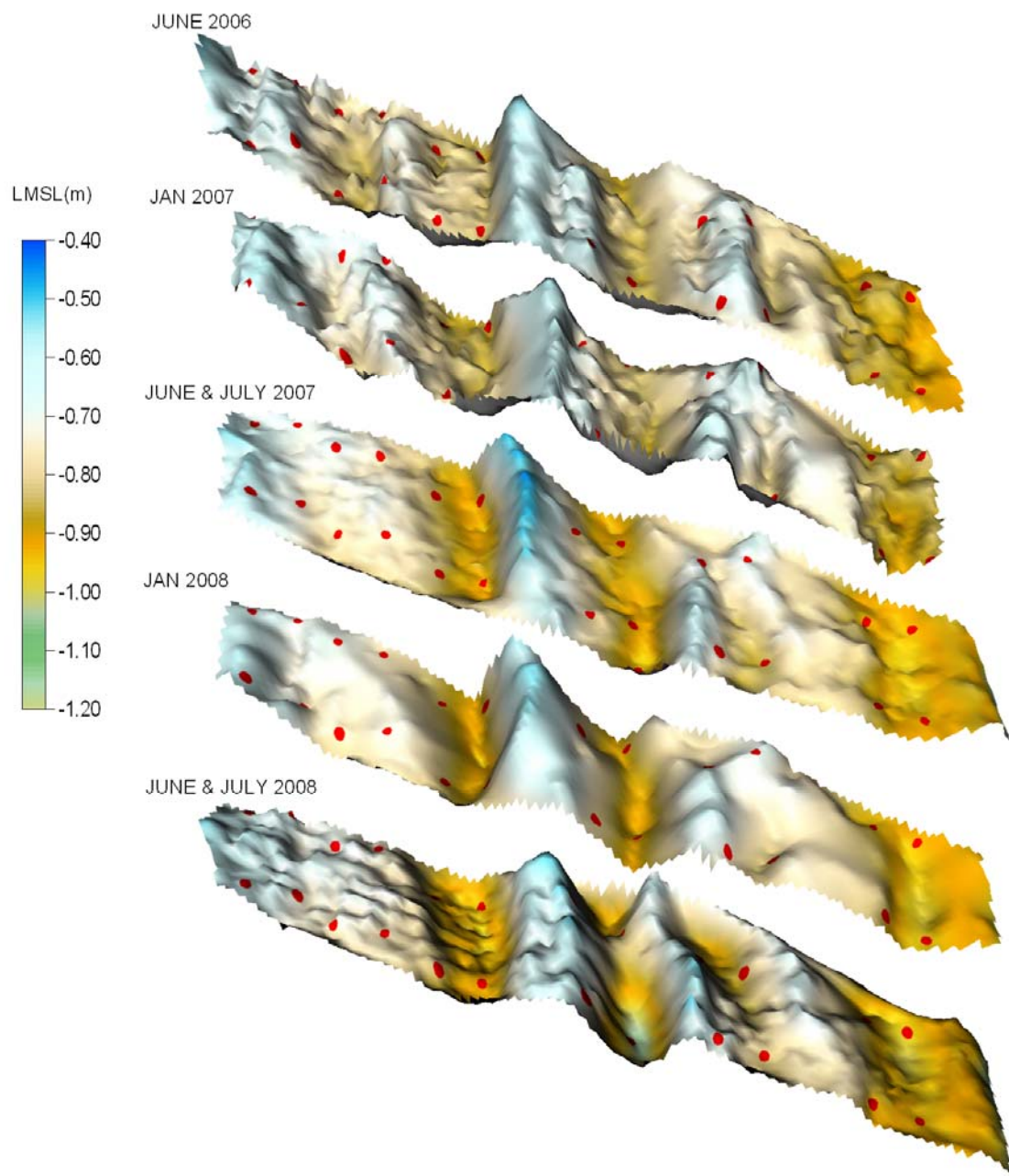


Figure 8. Sediment elevation surface plots showing bathymetry of the planting area at the start of the project and two years later at the end of the project. The red markers show the corners of the six planting plots (Plot 1, the most off-shore plot, is to the right and Plot 6, the most near-shore plot, is to the left).

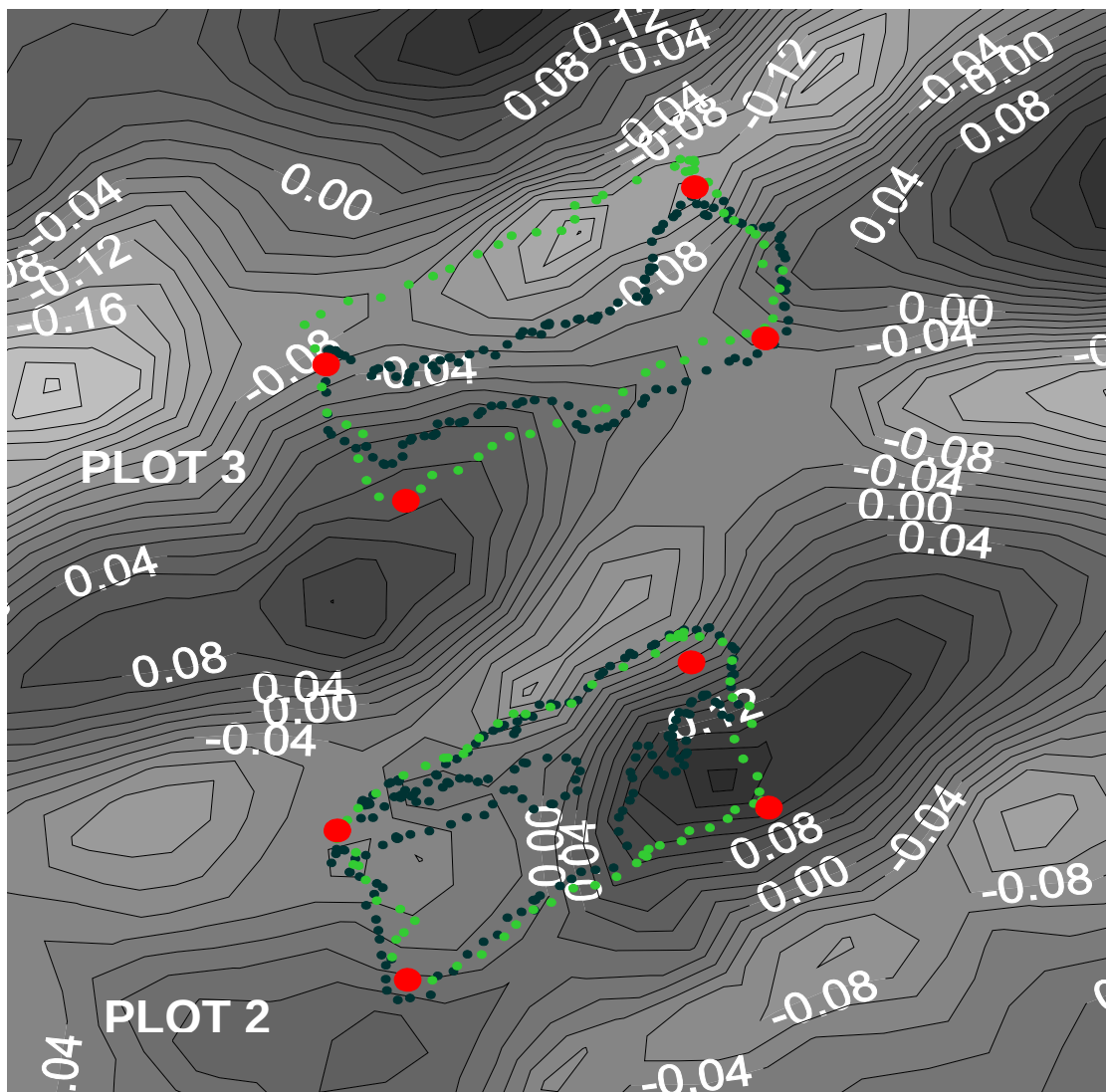


Figure 9. Contours of sediment elevation change that occurred within and near planting Plots 2 and 3 between January 2008 and summer 2008. Positive values and dark shading show areas that became deeper, and negative values and light shading indicate areas that became shallower. Also shown are perimeter surveys of manatee grass area coverage from January 2008 (light green symbols) and September 2008 (dark green symbols). The planting plot corner posts are also shown. The short sides of the plots are 10m.

ABSTRACTS

Session 5: SEAGRASS RESOURCES & MANAGEMENT

PRESENTATIONS

Managing Seagrass in Tampa Bay, Florida: A Multi-Scale Approach. L.M. Cross, H.S. Greening and E.T. Sherwood (Tampa Bay Estuary Program). Managing seagrass resources in Tampa Bay, Florida has evolved from a bay-wide approach, focused mainly on nutrient management, to incorporating localized impacts in smaller management areas. Recovery of seagrass acreage to levels observed in 1950 (15,400 ha) is a long-term goal adopted by local, state, federal and private partners participating in the Tampa Bay Estuary Program. A cooperative effort to reduce nitrogen loading from wastewater treatment facilities, stormwater runoff, fertilizer manufacturers, and power plants was initiated in 1980 and has resulted in a 60% TN load reduction compared to the mid-1970s. As a result, annual regulatory water clarity and chlorophyll-a targets are being met, and seagrass coverage in 2008 was the highest recorded since 1950 (but still 3,400 ha lower than 1950 estimates). However, seagrasses in all areas of the bay are not recovering at the same rate. Increased wave energy and the loss of longshore sandbars may coincide with poor seagrass recovery in some areas. Localized water quality factors, including CDOM (colored dissolved organic matter) and turbidity may also have impacts on seagrass growth. To reach the long-term recovery goal bay managers will have to move beyond merely managing nutrient inputs. A multi-scale adaptive research and application approach is currently underway to ensure continuation of the gains in Tampa Bay seagrass coverage. Specific research areas include defining smaller management areas, examining light quality as a function of depth and seagrass presence, characterizing the underwater light field, identifying persistent and ephemeral seagrass beds, and developing a bio-optical model for Tampa Bay. It is anticipated that seagrass targets will be set for specific management areas, as well as evaluating whether the current bay-wide restoration target is still applicable. [lcross@tbep.org]

Seagrass Persistence Analysis in Tampa Bay, 1988-2008. M. Dema and A.J. Janicki (Janicki Environmental Inc.). It is estimated that seagrass meadows covered approximately 40,400 acres of Tampa Bay in the 1950s. By 1982, that number had declined by nearly half, to 21,653 acres. Seagrass data from 1988 through 2008 indicate seagrass coverage has varied in the years since. In order to better understand the patterns of seagrass coverage in Tampa Bay, a persistence rule was developed and used to determine what the areal extent was by bay segment and which areas were most constant over the period of record. The persistence rule involved converting the seagrass coverage into a grid format. Each grid cell was designated as having seagrass for a given survey if more than 50% of the grid cell contained seagrass.

Persistence maps were created based on these recent surveys and used to characterize trends in seagrass throughout Tampa Bay. The most persistent seagrass areas are located near the shore in the shallowest portions of the estuary, whereas the least persistent areas are in the middle of the channel, where light attenuation is at its highest levels, due to the increased depths in these locations.

The results of this analysis show that some areas have never been, nor will be, well-suited for seagrass growth and that bay-wide persistence is variable by bay segment. Boca Ciega Bay and Lower Tampa Bay have the most widespread persistent patches of seagrasses in the study area. Hillsborough Bay has the least persistent seagrass communities, with most patches present for only one or two of the available surveys. The highly urbanized watershed of Hillsborough Bay receives a vast array of inputs, thereby negatively impacting the ability of seagrasses to grow in this bay segment. [mdema@janickienviroinmental.com]

HISTORY OF HABITAT RESTORATION EFFORTS IN TAMPA BAY: THE TAMPA BAY HABITAT RESTORATION DATABASE

L.M. Cross

ABSTRACT

The Tampa Bay Habitat Restoration Database was developed to compile and track the thousands of acres of established, enhanced, or protected habitat in the Tampa Bay watershed. The database was developed as a component of the 2009 Tampa Bay Habitat Master Plan Update with input from PBS&J, Scheda Ecological Associates, the Tampa Bay Estuary Program and its partners. It was compiled using various data and includes projects involving more than 30 agencies. It currently contains ~250 “complete” entries. While there are more than 50 fields in the database, complete entries must include project name, location (Latitude/Longitude), lead implementer, completion date, target habitat type(s), activity (enhancement, establishment or protection), and acres. Other fields include partners, watershed subbasin, cost, restoration techniques and project benefits. Incomplete projects are maintained in a separate database and will be moved to “complete” status when applicable.

The database was developed in Microsoft Access® and allows agencies to specify the exact acreage (or linear feet) of habitat restored per habitat type and activity. This provides an effective way to catalogue habitat “mosaic” projects without losing any detail. This also enables scientist and citizen users to search for projects using various fields, including location, acreage (or linear feet), habitat type restored or activity.

The interactive database is housed on the Tampa Bay Estuary Atlas www.tampabay.wateratlas.usf.edu/ and was a partnership with the City of Tampa’s Mayor’s Beautification Program. Scientists and citizens can view the projects, utilize map navigation tools, view information and download the complete database in Microsoft Access® or Excel® format. Since 1971, more than 5,000 acres of habitat have been established or enhanced in the Tampa Bay watershed. To submit projects or updates, please contact Lindsay Cross at lcross@tbep.org.

INTRODUCTION AND EVOLUTION OF HABITAT RESTORATION IN TAMPA BAY

Habitat restoration, the agencies performing restoration, and the way it is tracked has evolved over the past several decades in Tampa Bay. The size and complexity of projects and the number of agencies involved in restoration has increased, requiring more complex recording mechanisms.

Some of the first habitat restoration projects in Tampa Bay occurred in the 1970s. They were typically small projects, often less than an acre in size, and were usually limited to only a few wetland plant species. They often involved the planting of mangrove species: red mangrove (*Rhizophora mangle*), black mangrove (*Avicennia germinans*), and white mangrove (*Laguncularia racemosa*); or salt marsh, smooth cordgrass (*Spartina alterniflora*). For example, restoration projects at Terra Ceia, Bird Key and Mullet Key in 1971 all involved planting 0.01 acres of mangroves. Projects today are often more complex and have increased from single species projects to those incorporating a “habitat mosaic approach,” often with numerous habitat types ranging from tidal wetlands to uplands. Many projects include hydrologic modifications and other designs meant to restore historical conditions or provide buffers for climate change and sea level rise impacts. Many projects are also completed in phases and may total several hundred acres. The Cockroach Bay project in western Hillsborough County, for example, is being completed over 10 years and includes ~500 acres of restoration of tidal wetlands, freshwater wetlands and coastal uplands.

The complexity and breadth of agency involvement has also evolved. Projects in the 1970s and 1980s were implemented by agencies such as the Florida Department of Natural Resources/ Florida Department of Environmental Regulation (now Florida Department of Environmental Protection) and National Audubon Society. Currently, there are more than 30 agencies performing habitat restoration, including city and county governments, non-profits and for-profit businesses. Habitat restoration received a boost with the development of the Southwest Florida Water Management District (SWFWMD) Surface Water Improvement and Management (SWIM) Program. The SWIM Program was created as a result of the SWIM Act of 1987 (SWFWMD 1999) and directed the water management districts to “design and implement plans and programs for the improvement and management of surface waters” (Section 373.451, F.S.). Tampa Bay was selected as the first priority waterbody with support from the Agency on Bay Management. Several years later in 1991, the Tampa Bay National Estuary Program was established as one of 28 national estuary programs under the Clean Water Act and administered by the U.S. Environmental Protection Agency (US EPA; TBEP 1996). The current Tampa Bay SWIM Plan (SWFWMD 1999) outlines goals that support the Tampa Bay Estuary Program’s (TBEP) *Comprehensive Conservation and Management Plan* (TBEP 1996; TBEP 2006), such as reducing nitrogen loadings, protecting and restoring seagrass communities, and restoring wetland and upland habitats (SWIM 1999). The first SWIM restoration project in Tampa Bay entailed 0.5 acres of marsh restoration at Hendry Delta in Manatee County. Since then, the SWIM Program has completed more than 80 projects totaling more than 2,400 acres. Partnerships are also increasingly relied upon for implementing large-scale restoration projects.

Similarly, the need for a better tracking system arose as the complexity of projects increased. Record keeping began as hand-written notes kept in file cabinets; sometimes project information was never recorded. As computers became commonplace at agencies, project information was incorporated into electronic files. However, record keeping was typically limited to restoration performed by the agency only and did not include information from other agencies in the Tampa Bay area. The Tampa Bay Habitat Restoration Database represents the first comprehensive, electronic, geo-referenced and searchable database for habitat restoration in Tampa Bay.

Tampa Bay Habitat Master Plan - The development of the Tampa Bay Habitat Restoration Database (Database) was included as a component in the 2009 Tampa Bay Habitat Master Plan Update (Update) (PBS&J 2009). The original Tampa Bay Habitat Master Plan, entitled *Setting Priorities for Tampa Bay Habitat Protection and Restoration: Restoring the Balance* (Lewis and Robison 1995), was developed by Lewis Environmental Services, Inc. and Coastal Environmental, Inc. This report outlined several paradigms for restoration in Tampa Bay, including the concept of “restoring the balance” of emergent tidal wetlands. This recognizes that certain coastal habitats have been lost in greater proportion than others and outlines restoration and protection strategies in order to restore an optimum balance of wetland and associated upland habitats. Figure 1 demonstrates both the change in estimated acreage of tidal wetlands and the relative proportion of three wetland categories (mangroves, salt marsh and salt barren) between ca. 1950 and 2007. Specific restoration and protection targets were set also for low-salinity tidal streams, mangroves, salt marsh and salt barren habitats. Additionally, scientists and resource managers identified 28 specific sites that should be restored or protected when public dollars are available. This document also emphasized the value of providing a mosaic of habitat types to maximize benefits to fish and wildlife and to provide for habitat requirements throughout the life cycle of estuarine species. These elements were included in the original and updated CCMPs for Tampa Bay (TBEP 1996; TBEP 2006) and led to specific management actions and strategies. The Habitat Master Plan Update (PBS&J 2009) builds upon the concepts of restoring the balance and habitat mosaics and included other specific tasks, such as developing a habitat restoration database. The Update was written by TBEP’s technical contractor PBS&J, with Scheda

Ecological Associates, Inc. as a subcontractor, and with input from TBEP's Technical Advisory Committee (TAC) and other members of the Tampa Bay scientific community.

Development of the Database -The development of a database was included in the Update because of the need for a central and comprehensive database of restoration projects. There are numerous entities performing restoration in and around Tampa Bay that employ their own record-keeping strategies, however, there was not a single source that incorporated projects from all of the entities. Many agencies report the progress of restoration initiatives to funding agencies and others to document success. For example, the TBEP is required to report habitat restoration and protection acreages completed by its partners to the US Environmental Protection Agency (EPA) on an annual basis by the Government Performance and Results Act (GPRA) of 1993. Typically TBEP staff or its technical consultant contacts partner agencies, such as city and county governments, non-profits and the SWIM Program, several months prior to the GPRA due date to collate project information. While meeting reporting requirements, the structure and information requested by EPA was not seen as rigorous enough to fully understand restoration activities in Tampa Bay and the specific information on the GPRA reporting forms can vary from year-to-year. The required information included in the Tampa Bay Habitat Restoration Database was designed to satisfy government reporting requirements such as GPRA. Additionally, it is more comprehensive and consistent, giving a more accurate portrayal of habitat activities and is thorough enough to allow detailed search queries by resources managers and the public in Tampa Bay and beyond.

Data Sources - The Database was created using habitat information from numerous TBEP partners. Data sources included, but are not limited to, SWIM Geographic Information Systems (GIS) Shapefiles and other project information, a National Oceanographic and Atmospheric Administration (NOAA) National Estuaries Restoration Inventory (NERI) database, hard copy notes of SWIM and FDNR/FDER projects from 1970-1990, Hillsborough County wetland mitigation projects, Tampa Bay Watch restoration and homeowner oyster reef projects, and TBEP GPRA reports. The inclusive list of ~1,000 entries was provided to members of the TBEP's TAC for review in December 2005. Scheda staff then performed an extensive review of the projects to correct errors and remove duplicate entries as the result of more than one agency recording and submitting information on a project or alternate names being used. The database was then decreased to ~450 entries. TBEP staff then met with partners individually between December 2006 and January 2007 to further refine the database, correct errors and develop appropriate database categories.

Database Structure - There was considerable effort involved in developing the appropriate database structure and fields. TBEP, PBS&J and Scheda staff agreed that it was imperative that the database allowed for detailed queries and be compatible with restoration projects employing a habitat mosaic approach. Therefore, a Microsoft Access® structure was employed as it allows for linked tables and subentries. Often, when reporting results to funding agencies, a resource manager will provide the total project acreage and the dominant habitat types. This information may be an aggregation of multiple restoration techniques and habitat types. For example, an agency may report a project involving 11.25 acres of marsh restoration, while the project details may reveal that there were 5 acres of oligohaline marsh creation, 3.5 acres of high marsh enhancement and 2.75 acres of forested upland protection. This level of detail is important for understanding the true restoration activities occurring in the Tampa Bay watershed. The TB Habitat Database has been developed so that users can enter this level of detail and specificity and, consequently, queries can pull out this information easily. Figure 2 shows an example query from the database which pulls out only projects that established or enhanced oligohaline marsh as the Target Habitat Type 1 or 2.

Restoration Activities - Habitat restoration activities can be described using various terms, such as restoration, creation, enhancement, protection, acquisition and others. In order to ensure consistency in reporting, the project team defined three activities that are used in the Database:

- **Establishment**: The manipulation of the physical, chemical or biological characteristics present to develop a habitat that did not previously exist on an upland or deep-water site. Establishment results in a gain in acres. Includes activities commonly associated with creation.
- **Enhancement**: The manipulation of the physical, chemical, or biological characteristics of an undisturbed or degraded site to heighten, intensify, or improve specific functions or to change the growth stage or composition of the vegetation present. Enhancement results in water quality improvement, flood water retention or wildlife habitat improvement, but does not result in a gain in habitat acres. The term includes activities commonly associated with the terms enhancement, management, manipulation and directed alteration.
- **Protection**: The preservation of habitat by mechanisms such as land acquisition, conservation easements, deed restrictions, etc. and does not result in a gain in habitat acres.

Target Habitat Types - The project team developed Tampa Bay-specific habitat categories for inclusion in the database. The target habitat categories were selected to more accurately portray coastal and upland habitats in the Tampa Bay area and are generally more specific than those used by other agencies. The team reviewed habitat categories used by other agencies such as EPA and NOAA and created a cross-walk of habitat categories so that information collected could satisfy reporting requirements for various agencies. For example, the category “tidal wetland” used by EPA and expressed as “salt marsh” in the NOAA NERI database, was expanded to “high marsh,” “low marsh,” “oligohaline marsh,” and “salt barrens” in the Database. This more fully describes the range of salinities and habitat characteristics of estuarine marsh habitats in the Tampa Bay watershed, however, other agencies and users can roll these back into a more general category for their purposes. The project team also differentiated between natural hardbottom habitats and artificial hardbottom, such as constructed oyster bars, artificial reefs or oyster domes. The complete cross-walk with descriptions is included as a table in the Access® database so that users can easily make comparisons. For reporting purposes, Tampa Bay agencies can use a broader term, such as “tidal wetland” using the crosswalk but will not lose the specificity of the Tampa Bay categories.

Database Fields - Project information includes an array of features ranging from the agency and partner contact information, GIS location of the project site, restoration techniques, acreages and funding. In total, there are more than 50 fields; while not all are required or relevant for each project, there are eight fields that must be completed for the project to be considered “complete” within the database. These fields include the project name and alternate names, if applicable; latitude/longitude; lead implementer; contact information for lead agency (name, address, email, phone); completion date; target habitat type(s); activity (establish, enhance, protect); and acres (or linear feet). Projects for which all required information is not supplied are moved to an “incomplete” status within another database, but can be moved to the complete, active Database when updates are provided.

Partnership with the City of Tampa’s Mayor’s Beautification Program and Tampa Bay Water Atlas - The Database is currently housed on an interactive website that is accessible to scientists, managers and the public. As part of its outreach and scientific efforts, the TBEP supports the Tampa Bay Water Atlas, created by the University of South Florida (USF) (<http://www.tampabay.wateratlas.usf.edu/>). The purpose of the Tampa Bay Water Atlas is “to provide citizens, scientists, professionals, and planners with comprehensive and current water quality, hydrologic, and ecological data, as well as information about recreational opportunities and a library of scientific and educational materials on water resource issues.” It contains

extensive data downloads including real-time water quality data, reference documents on a range of ecological topics, educational links, and waterbody maps. There are currently 11 other Water Atlases in the state of Florida, including a Pinellas County, Manatee County and Hillsborough County and City of Tampa Water Atlas. During the development of the Database, the City of Tampa's Mayor's Beautification Program (MBP) approached USF about posting a database of MBP projects on an Atlas site. The MBP utilized grant funding to implement Tampa Shoreline Restoration Initiative to restore 30 parks within the City of Tampa containing 100,000 linear feet of shoreline. The Program was looking for a location to display the project success and allow volunteers to contribute information and view completed restoration projects.

Working in partnership, the TBEP and MBP directed USF to create a unique geo-referenced page within the Tampa Bay Water Atlas to house the restoration projects and supporting databases (Figure 3). The MBP funded the development of the web page within the Tampa Bay Water Atlas and, in exchange; the TBEP now has a designated site to post geo-referenced restoration projects and the complete database on the site. This beneficial partnership has allowed scientists and the public to view the progress towards meeting habitat restoration and protection goals in Tampa Bay.

Database Mapping Tools - The Water Atlas displays each of the complete projects in a map of the Tampa Bay watershed. Users are able to use a variety of mapping tools, such as zooming in and panning to see a project site spatially. By clicking a "Get Info/Identify" icon, they can also see a subset of project information, such as name, habitat type restored and acreages. If more information is desired, they can download the entire Database in a Microsoft Access® or Excel® format. They can then perform search queries of individual projects within the entire Database. The Atlas also provides additional information that may be relevant to the user, such as downloadable water quality data near the project site or information on wildlife in the area.

Progress in Meeting Habitat Restoration Goals in Tampa Bay - Habitat restoration in Tampa Bay is helping to offset losses due to development and other impacts. Based on updates through 2006, 5,317 acres of habitat have been restored in Tampa Bay (Figure 4). Restoration and acquisition of habitats will be increasingly relied upon to ensure that viable habitats are maintained in the Tampa Bay watershed.

UPDATES AND FUTURE WORK

The Database was designed to be a living document and the TBEP strives to update the Database annually, in concert with GPRA reporting requirements. TBEP also encourages its partner agencies to review "incomplete" projects and supply missing information needed to move projects to a complete status, improving the accuracy of the Database. Additional information that goes beyond the required fields, such as project costs or initial habitat types, will only strengthen the Database and increase its utility. TBEP will also be developing an improved reporting form for partner agencies that will include Database fields and those required by funding agencies. TBEP encourages scientists and citizens to visit the website, view the mapping tools, download the Database, develop search queries and provide feedback to TBEP.

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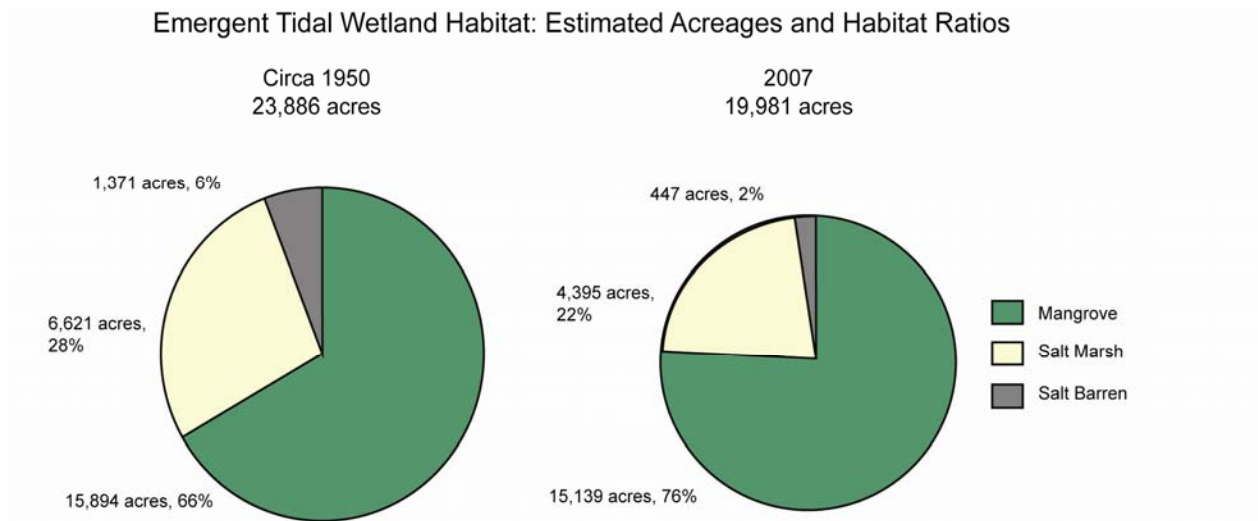


Figure 1. Estimated acreages and ratios of emergent tidal habitats circa 1950 and 2007.

Project Name	Target Habitat Type-1	Target Habitat Type-2	Restoration Activity	Acres of Activity
Allen's Creek at St. Paul's Drive	Oligohaline Marsh		Establishment	1
Bartlett Park	Low Marsh	Oligohaline Marsh	Establishment	4
Braden River: Phase 1 (SR64)	Oligohaline Marsh		Establishment	10
Braden River: Phase 2 (SR70)	Oligohaline Marsh		Establishment	10
Cockroach Bay: Phase 1B	Freshwater Wetland	Oligohaline Marsh	Establishment	24.5
Cockroach Bay: Phase 1B2	Low Marsh	Oligohaline Marsh	Establishment	18.5
Del Oro Park	Oligohaline Marsh		Establishment	1
Harbor Palms Park	Oligohaline Marsh		Establishment	4.4
Little Bayou Habitat Restoration Project: Phase 1	Oligohaline Marsh	Estuarine Water Column	Establishment	3
Long/Cross Bayou	Oligohaline Marsh		Establishment	2
Lowry Park: Spring and Shoreline	Isolated Freshwater Wetland	Oligohaline Marsh	Establishment	0.5
McKay Bay Northeast	Oligohaline Marsh	Coastal Uplands	Establishment	4
North Parcel West A	Coastal Uplands	Oligohaline Marsh	Enhancement	123.9
North Parcel West B	Mangrove Shoreline	Oligohaline Marsh	Enhancement	123.9
Wolf Branch Creek 1	Oligohaline Marsh	Low Marsh	Establishment	181
Total				511.7

Figure 2. Example Query from Database: Projects that Enhanced or Established Oligohaline Marsh as Target Habitat #1or #2.

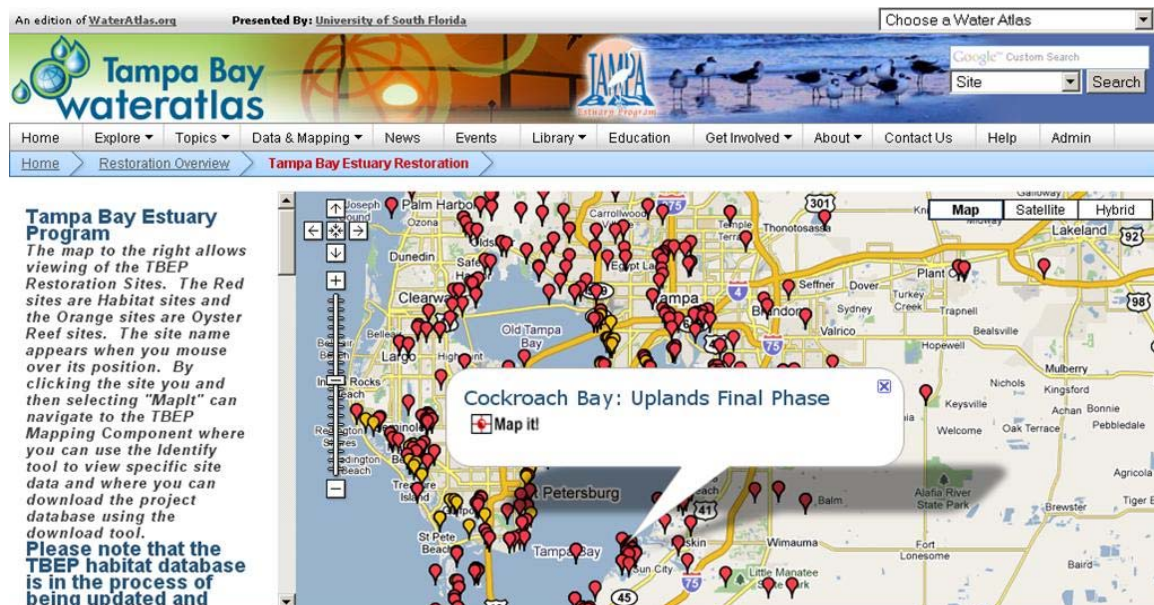


Figure 3. Screen Shot from Tampa Bay Water Atlas



Figure 4. Habitats Restored in Tampa Bay through 2006 (5 habitat types)

GOALS AND STRATEGIES FOR ECOSYSTEM RESTORATION AND PRESERVATION IN THE TAMPA BAY WATERSHED

D.E. Robison

ABSTRACT

This paper summarizes work conducted for the Tampa Bay Estuary Program as part of the *Tampa Bay Habitat Master Plan Update* (PBS&J 2009). Specifically, this paper presents findings and recommendations from the Plan, including: the results of habitat status and trend analyses; numeric target development; and new strategies for attaining more effective habitat restoration and preservation in Tampa Bay and its watershed. Habitats most critical to supporting the life history patterns of estuarine-dependent species include: seagrass meadows, mangrove forests, salt marshes, and salt barrens. Based on habitat trend analyses and the “Restoring the Balance” target setting algorithm, it was determined that deficits currently exist for the areal extent of seagrass (8,905 acres), low salinity salt marsh (1,918 acres), and salt barrens (840 acres) in Tampa Bay, and that restoration of these habitat types are a high priority. In addition, it was determined that the areal extent of mangrove forest is adequate, and that further restoration for this habitat type is not a priority at this time. Strategies for accelerating attainment of habitat restoration goals include: targeted public land acquisition; increased public-private partnerships; developing a linkage between publicly-funded habitat restoration and compensatory mitigation; and the implementation of an integrated watershed approach to large, publicly-funded habitat restoration projects.

INTRODUCTION

The document entitled *Setting Priorities for Tampa Bay Habitat Protection and Restoration: Restoring the Balance* – Tampa Bay Estuary Program Technical Publication No. 09-95 (Lewis and Robison 1996), constituted the first adopted habitat master plan for Tampa Bay. Since its publication, this report has locally been referred to as the “Restoring the Balance” document. This paper summarizes work conducted for the Tampa Bay Estuary Program as part of the *Tampa Bay Habitat Master Plan Update* (PBS&J 2009). Specifically, this paper presents findings and recommendations from this Plan including: the results of habitat status and trend analyses; numeric target development; and new strategies for attaining more effective habitat restoration and preservation in Tampa Bay and its watershed.

The word “habitat” is derived from the Latin word *habitare*, which means to dwell, inhabit, or possess. The term is widely used, not just in ecology, but elsewhere. It is generally understood to mean simply the place where an organism, or a population of organisms, normally lives or occurs (Odum 1971). Habitat can be described in terms of the abiotic environment occupied by an organism, as well as the organisms’ biological dependencies and affinities. In context of the *Tampa Bay Habitat Master Plan Update*, habitat can best be defined as substrates and structures – both physical and biological – that provide critical shelter and food for estuarine-dependent organisms. Habitats considered most critical to supporting the life histories of estuarine-dependent species include:

- Seagrass meadows;
- Emergent tidal wetlands;
 - o Mangrove forests;
 - o Salt marshes;
 - o Salt barrens.

Seagrasses perform many critical ecological functions in the estuarine and marine environment. They help maintain water clarity by trapping fine sediments and particles with their leaves, and

stabilizing bottom sediments with their roots and rhizomes. Seagrass leaf canopies protect young marine animals from larger predators, and some animals, such as manatees and sea turtles, eat seagrass blades. Other animals derive nutrition from eating algae and small animals that live upon seagrass leaves. These colonizing organisms provide an additional link in the marine food web. Seagrass-based detritus formed by the microbial breakdown of leaves and roots is also an important food source.

The three types of emergent tidal wetlands collectively form an important habitat complex in Tampa Bay. They provide critical habitats for much of the bay's wildlife, are an important component of nutrient cycles, stabilize submerged shoreline sediments thereby minimizing shoreline erosion, and assimilate pollutants carried in runoff from upland urban areas. Emergent tidal wetlands provide attachment sites for algal and invertebrate communities, and provide habitat below the water surface for hundreds of recreational and commercially-important species of fish, shrimp, crabs and other shellfish. Such species include pink shrimp, menhaden, blue crabs, mullet, red drum, tarpon, and snook. The marsh grasses and mangrove trees also provide critical feeding, nesting and sheltering habitat for a variety of birds such as pelicans, cormorants, herons, ibises, spoonbills and egrets.

Salinity is a critical factor in determining the distribution and species composition of emergent tidal wetlands. Mangrove forests and associated cordgrass (*Spartina alterniflora*) fringe marshes typically occur within the polyhaline zone (18-30 ppt), whereas mangroves and black needlerush (*Juncus roemerianus*) marshes typically occur within the mesohaline zone (5-18 ppt). The oligohaline zone (0.5-5 ppt) is typically the most diverse, often supporting six or more herbaceous species dominated by black needlerush, leather fern, and occasional black mangroves. Salt barrens occur in hypersaline zones (>40 ppt) caused by seawater ponding and evaporation.

Although these general distributional patterns are well documented, it should be noted that most estuarine plant species have broad salinity range tolerances, and can withstand extremes on both ends of the scale. Consequently, the observed zonation of estuarine plant communities essentially represents an integration of numerous factors, including but not limited to: salinity; wave energy; interspecific competition; sediment grain size and nutrients; and exposure to freezing temperatures.

In addition to seagrass meadows and emergent tidal wetlands, six other habitat types are considered to be important and are addressed in the Tampa Bay Habitat Master Plan Update. These include:

- Oyster bars;
- Hardbottom;
- Tidal creeks and rivers;
- Artificial habitats;
- Coastal uplands;
- Freshwater wetlands.

To a greater or lesser extent all of the critical and important habitat types in the Tampa Bay estuarine system and watershed have been adversely impacted by anthropogenic activities. The remainder of this paper discusses goal setting and strategy development for habitat restoration and protection.

GOAL SETTING PARADIGMS

Two habitat restoration goal setting paradigms were considered early in the TBEP planning process:

- Restore to pre-development conditions;
- Restore to benchmark period conditions.

Restoring to pre-development conditions was considered to be economically infeasible, and eliminated as an alternative early in the planning process. Restoring to benchmark conditions was considered to be economically feasible, and the circa-1950 time period was adopted by the TBEP as the benchmark because: reasonably good data and aerial photography existed for this time period; and the structural and functional integrity of bay ecosystems were considered to be largely intact, even though significant anthropogenic impacts had already occurred.

Once the benchmark period approach was chosen, two alternatives were considered for habitat restoration goal setting:

- Restore the *areal extent* of critical habitats back to benchmark period conditions;
- Restore the *relative proportions* of critical habitats back to benchmark period conditions.

Restoring the areal extent of habitats back to benchmark conditions was considered feasible for seagrass meadows since the current surface area of bay bottom is comparable to the benchmark surface area. However, restoring the areal extent of emergent tidal wetlands was considered infeasible due to the fact that the large areas of intertidal wetlands that existed during the benchmark period have been irretrievably lost to dredge and fill activities. Therefore, restoring the relative proportions was chosen as the most appropriate goal setting paradigm for emergent tidal wetlands. This approach was termed “restoring the balance” (RTB). The underlying assumption of the RTB approach is that estuarine-dependent species evolved to complete their life histories utilizing the relative proportions of emergent tidal wetlands that existed during pre-development conditions, and that the pre-development proportions of these three habitat types were still relatively intact during the benchmark period.

HABITAT STATUS AND TREND ANALYSIS

Goal setting using the RTB approach requires quantification of the areal extents and relative proportions of the three types of emergent tidal wetlands under benchmark and current conditions. The materials and methods for this analysis are described in the Tampa Bay Habitat Master Plan Update (TBEP 2009). Table 1 shows the benchmark (circa-1950) and current (2007) areal extent and relative proportion of mangrove, salt marsh, and salt barren habitats; while Figure 1 graphically depicts the relative proportions of the three types of emergent tidal wetlands for the benchmark and current time periods.

Table 1. Areal extents and relative proportions of emergent tidal wetlands for the benchmark and current time periods.

Habitat Type	ca. 1950 acres	ca. 1950 percent	2007 acres	2007 percent
Mangrove Forest	15,894	66.6	15,139	75.8
Salt Marsh	6,621	27.7	4,395	22.0
Salt Barren	1,371	5.7	447	2.2
Totals	23,886	100	19,981	100

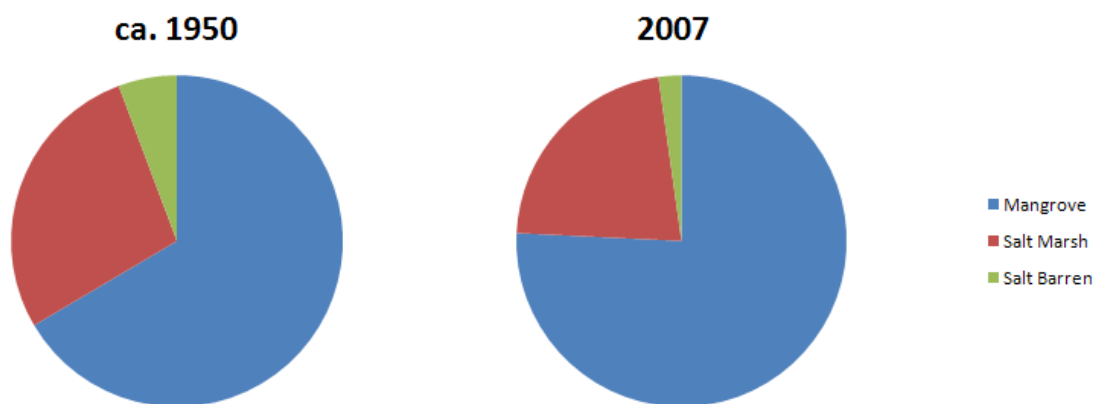


Figure 1. Relative proportions of emergent tidal wetlands for the benchmark and current time periods.

Changes in the areal extent of emergent tidal wetlands were also quantified for two time periods: ca.1950-1990 and 1995-2007. Although the length of these time periods is not equal, they generally represent pre- and post-regulatory controls over wetland impacts, respectively. Table 2 shows the changes in the three types of emergent tidal wetlands during the pre-regulatory and post-regulatory time periods. As shown in Table 2, very significant losses in the areal extent of emergent tidal wetlands occurred between ca. 1950 and 1990, but since 1995 there have actually been gains in all three habitat types.

Table 2. Comparison of emergent tidal wetland changes between the periods 1950-1990 and 1995-2007.

Habitat Type	1950	1990	Change acres	Change percent	1995	2007	Change acres	Change percent
Mangrove Forest	15,894	13,846	-2,048	-12.9%	14,760	15,139	379	+2.7%
Salt Marsh	6,621	4,169	-2,452	-37.0%	4,343	4,395	52	+1.2%
Salt Barren	1,371	887	-484	-35.3%	445	447	2	+0.4%
Totals	23,886	18,902	-4,984	-20.9%	19,548	19,981	433	+2.2%

The methods used to conduct these emergent tidal wetland trend analyses imparted significant errors due to: 1) different GIS standards (1950 data used a different classification system; 1990 SWFWMD data is not directly comparable to data from 1995 forward); 2) photo-interpretation inconsistencies; 3) classification errors; and, 4) difficulty in classifying transitional areas. Nonetheless, the results are valid for large scale changes over long time periods (e.g., ca. 1950 to 1990), and are likely valid for smaller scale changes since 1995.

Changes in the areal extent of emergent tidal wetlands can be classified into three types: 1) losses; 2) gains; and 3) conversions from one type to another type of emergent tidal wetland (e.g., salt marsh transitioning into mangrove forest). The potential causative factors affecting emergent tidal wetland losses include: dredge and fill activities; sea level rise (e.g., compression of the intertidal zone); and erosion. The causative factors affecting gains include: compensatory mitigation; habitat restoration; and accretion. Finally, causative factors affecting habitat conversions include: climate change (e.g., no significant mangrove freezes since the 1990s); sea

level rise (e.g., increased tidal flushing of salt barrens); and consumptive water use (e.g., increased salinity in tidal rivers used for water supply).

Between the circa 1950 benchmark period and 1990, dredge and fill activities were clearly the most dominant stressor affecting change in emergent tidal wetlands in Tampa Bay. However, during the 1995-2007 time period, it is likely that cumulative smaller impacts to emergent tidal wetlands have been caused by a variety of stressors including minor dredge and fill activities, as well as those associated with more subtle perturbations such as climate change, sea level rise, localized hydrologic modifications, and alterations to overall bay hydrology (e.g., surface water withdrawals). Figure 2 shows a summary of emergent tidal wetland conversions for the period 1995-2007.

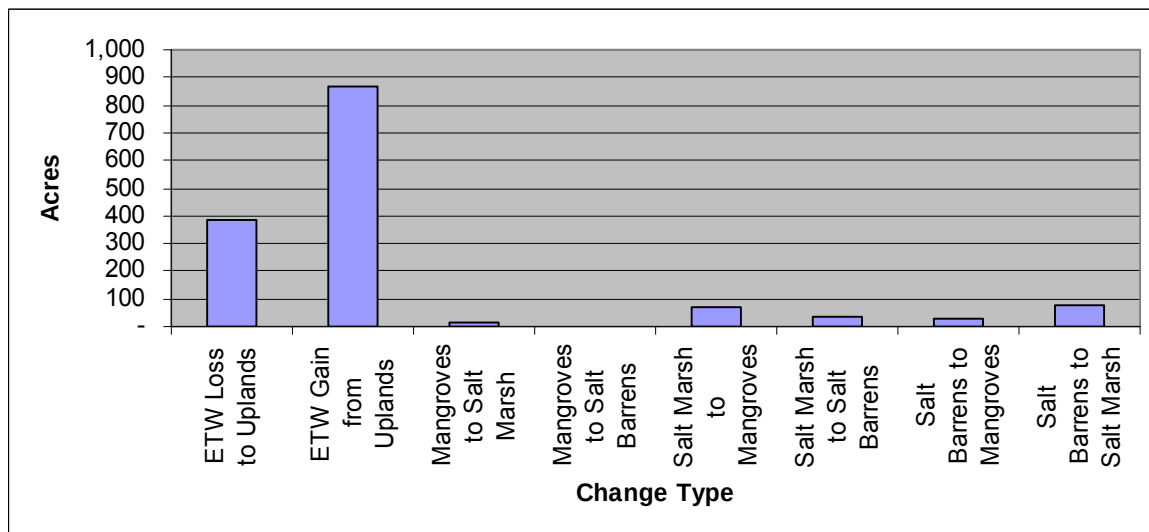


Figure 2. Emergent tidal wetland conversions for the period 1995-2007.

As shown in Figure 2, emergent tidal wetlands losses to uplands are caused by fill activities, while gains from uplands are most likely caused by compensatory mitigation and publicly-funded habitat restoration projects. Other changes between emergent tidal wetland types (e.g., salt marsh conversion to mangrove) may be related sea level rise, climate change and other hydrologic alterations.

TARGET SETTING

In the “Restoring the Balance” document (Lewis and Robison 1996), the RTB algorithm for habitat restoration target development is described as follows: the ratio of benchmark proportion and current areal extent of the *least impacted* habitat type should be equivalent to the ratio of the benchmark proportion and restoration areal extent of the other habitat types. Rather than replicating the benchmark proportions of the various habitat types, the RTB approach uses the least-impacted habitat type as the “common denominator.” The RTB equation is defined below:

If: x = restoration target for habitat of interest
 a = current acreage of least-impacted habitat
 b = benchmark proportion of habitat of interest
 c = benchmark proportion of least-impacted habitat

Then,
 $c/a = b/x$ and $x = (ab)/c$

Emergent Tidal Wetlands—The RTB equation was used to calculate restoration targets for the three types of emergent tidal wetlands using the benchmark (ca. 1950) and current (2007) areal extents and proportions. Table 4 shows the calculated RTB targets, the current areal extents, and the current deficits for the three types of emergent tidal wetlands.

Table 3. RTB restoration targets for emergent tidal wetlands.

Habitat Type	RTB Targets acres	Current Areal Extent* acres	Current Deficit acres
Mangrove Forest	15,139	15,139	0
Salt Marsh	6,297	4,395	-1,902
Salt Barren	1,296	447	-849
Totals	22,722	19,981	-2,751

*2007 emergent tidal wetland data.

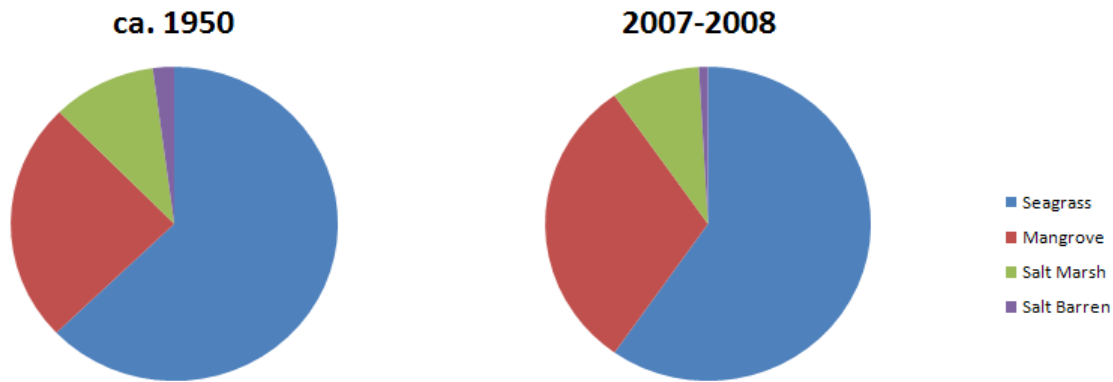
Seagrass/Emergent Tidal Wetland Complex—One criticism of the original RTB target setting approach is that it did not include seagrass meadows as one of the components of the complex of habitats that support the life histories of estuarine-dependent species. There is ample literature supporting the utilization of the full continuum of emergent and submerged plant communities by estuarine dependent fish and invertebrates during their various life history stages. Examples of economically-important species in Tampa Bay that require the full continuum of estuarine habitat to complete their life histories include snook (*Centropomus undecimalis*) (McMichael et al. 1989), and redfish (*Sciaenops ocellatus*) (Peters and McMichael 1987). Therefore, in the *Tampa Bay Habitat Master Plan Update* seagrass was incorporated in the RTB target setting paradigm. Table 4 shows the benchmark (circa-1950) and current areal extent and relative proportion of seagrass (2008 data) and emergent tidal wetlands (2007 data); while Figure 3 graphically depicts the relative proportions of the four critical habitat types for the benchmark and current time periods.

Table 4. Areal extents and relative proportions of seagrass and emergent tidal wetlands for the benchmark and current time periods.

Habitat Type	ca. 1950 acres	ca. 1950 percent	2007-2008* acres	2007-2008* percent
Seagrass	40,420	62.9	29,647	59.7
Mangrove Forest	15,894	24.7	15,139	30.5
Salt Marsh	6,621	10.3	4,395	8.9
Salt Barren	1,371	2.1	447	0.9
Total	64,306	100	49,628	100

*2008 seagrass data; 2007 emergent tidal wetland data.

Figure 3. Relative proportions of seagrass and emergent tidal wetlands for the benchmark and current time periods.



The RTB equation was used to calculate restoration targets for seagrass and the three types of emergent tidal wetlands using the benchmark (ca. 1950) and current (2007) areal extents and proportions, and 2008 seagrass data. Table 5 shows the calculated RTB targets, the current areal extents, and the current deficits for seagrass and emergent tidal wetlands.

Table 5. RTB restoration targets for seagrass and emergent tidal wetlands.

Habitat Type	RTB Targets acres	Current Areal Extent* acres	Current Deficit acres
Seagrass	38,552	29,647	-8,905
Mangrove Forest	15,139	15,139	0
Salt Marsh	6,297	4,395	-1,918
Salt Barren	1,296	447	-840
Totals	61,284	49,628	-11,663

*2008 seagrass data; 2007 emergent tidal wetland data.

Interestingly, the calculated RTB seagrass restoration target of 38,552 acres is remarkably similar to the 38,000-acre seagrass recovery target developed by the TBEP using a water quality-based modeling approach. Likewise, the estimated current deficits, based on the 2008 seagrass estimates, are also remarkably close (8,905 acres vs. 8,353 acres). While the similarity of these results may be serendipitous, both approaches are scientifically-defensible, thus lending additional credibility to the findings. Furthermore, when seagrass is included in the algorithm, the RTB goals for the three types of emergent tidal wetlands are very similar to those calculated when only the three types of emergent tidal wetlands are considered.

TARGET SUMMARY

In addition to the target setting for seagrass and emergent tidal wetlands, the *Habitat Master Plan Update* also addresses quantitative analyses and target setting for the other important habitat types in Tampa Bay and its watershed. Table 6 presents a summary of the preservation and restoration targets for all recognized critical and important habitat types. It was determined that additional artificial habitats in Tampa Bay are not needed, with the exception of hardened urban shorelines (e.g., vertical seawalls) where a quantitative target for additional oyster dome placement was established (1 linear mile per year). For the remaining important habitats

inadequate quantitative information exists to establish numeric preservation and restoration targets. Quantitative assessments are needed for oyster bars, hardbottom, tidal tributaries, coastal uplands and flatwoods marshes. The importance of tidal tributaries to the bay cannot be overemphasized as they provide critical freshwater inputs and deliver nutrients which support the estuarine food web. Furthermore, tidal tributaries maintain the salinity gradients that support the complex of emergent tidal wetlands so critical to the life history patterns of estuarine-dependent species. Without a quantitative restoration target it was concluded that hydrologic restoration of altered tidal tributaries should be a high priority wherever feasible.

Table 6. Summary of preservation and restoration targets (in acres) for critical and important habitats in Tampa Bay and its watershed.

Habitat Type	Preservation Target	Restoration Target	Current Deficit	Comments
Seagrass	29,647 acres	38,552 acres	8,905	Restoration target attained via water quality improvement
Mangrove	15,139 acres	15,139 acres	0 acres	Preservation target only – new restoration is a low priority
Salt Marsh	4,395 acres	6,297 acres	1,918 acres	Restoration target to be attained via public, public-private partnerships and mitigation projects; oligohaline habitats should be identified as a subset of these targets in the future
Salt Barren	447 acres	1,296 acres	840 acres	Restoration target to be attained via public, public-private partnerships and mitigation projects
Oyster Bar	~44 acres	Inadequate Data	Unknown	Quantitative assessment needed to develop numeric targets
Hardbottom	All Existing	Inadequate Data	Unknown	Quantitative assessment needed to develop numeric targets
Tidal Tributaries	~1,400 lm	Inadequate Data	Unknown	Restore natural hydrology wherever feasible
Artificial Habitats	Not Applicable	Not Applicable	0 acres	Encourage the enhancement of urban seawalls
Coastal Uplands	~12,929 acres	Inadequate Data	Unknown	Quantitative assessment needed to develop numeric targets
Flatwoods Marshes	~27,243 acres	Inadequate Data	Unknown	Quantitative assessment needed to develop numeric targets

RESTORATION STRATEGIES

To attain the goals summarized in Table 6, multiple strategies will need to be employed over several decades. The primary mechanism for restoration, and preservation, of critical habitats will continue to be public land acquisition and publicly-funded habitat restoration projects. It is anticipated that the Southwest Florida Water Management District (SWFWMD) and local government environmental lands programs will continue to be the entities bearing most of the

responsibility for these activities. Through this approach, tremendous progress has been made at securing most of the remaining undeveloped parcels under public ownership. Recent budget constraints will likely reduce the capabilities for these agencies to secure additional lands over the next several years; however, reduced land cost may at least partially offset budget limitations. Nonetheless, other mechanisms will be needed to accelerate progress towards attaining habitat restoration targets.

Public private partnerships (P3 projects) offer great potential to accelerate progress without the need for public land acquisition. Private landowners with large strategically located land holdings include:

- Electric utilities;
- Port authorities;
- Mining interests;
- Water supply authorities.

Owners of large private land holdings could be incentivized to cooperatively fund habitat restoration projects, and record conservation easements, on their land holdings through mitigation and/or pollution trading credits, as well as enhanced public relations.

Another strategy for accelerating progress towards attaining habitat restoration targets involves developing a linkage between habitat restoration targets, publicly-funded habitat restoration, and compensatory mitigation in the Tampa Bay watershed. This strategy could include a number of options including:

- Expanding the SWFWMD in-lieu fee program to include wetland impacts incurred by other permittees in addition to the Florida Department of Transportation; and
- Leasing publicly-owned lands to private mitigation bankers.

The former option could significantly increase public funds available for restoration projects within the watershed. The latter option has numerous benefits. First, the public would retain ownership of the lands in perpetuity. Second, it would create strong financial incentives for private investors to facilitate restoration activities because they would avoid land acquisition and ownership costs, and would own the mitigation credits in perpetuity. These options are worthy of further evaluation.

Finally, progress towards attaining habitat restoration targets could be accelerated through the implementation of an integrated watershed approach to publicly-funded habitat restoration projects. Under this approach, public funds would be dedicated to larger, multi-objective projects conducted at the basin or sub-basin level. Such projects would strive to address habitat restoration, hydrologic restoration, water quality improvement, and even flood protection in an integrated manner. Examples of potential projects at this scale in the Tampa Bay watershed include two major basins that have undergone severe anthropogenic alterations: the Tampa Bypass Canal, and the Lake Tarpon Outfall Canal.

The Tampa Bypass Canal, formerly the Palm River, now serves as a major deep-dredged flood conveyance and regional water supply source. Virtually no freshwater discharges to the bay occur under most conditions due to potable water withdrawals. However, during flood conditions, very large volumes of water are discharged through the canal. These operational uses of the Tampa Bypass Canal preclude the establishment of normal estuarine salinity gradients and native plant communities. Restoration of estuarine functions in the Tampa Bypass Canal could be achieved by diverting highly-treated wastewater from the City of Tampa's Howard F. Curren WWTP through an existing tidal wetland system, thus establishing salinity gradients in upper McKay Bay. Figure 4 conceptually illustrates this project.



Figure 4. Conceptual plan for the Tampa Bypass Canal and Upper McKay Bay habitat restoration project.

The Lake Tarpon Outfall Canal was historically a closed basin through which a channel was constructed to reduce flooding around Lake Tarpon, but now delivers very significant quantities of freshwater and sediment to Old Tampa Bay. Enhancement of estuarine functions could be achieved by constructing a cascading series of tidal wetlands adjacent to the canal and diverting low flows from the lake through the wetlands. This project would create the intersection of salinity gradients and native plant communities where none exist currently, and would allow estuarine-dependent species to better utilize the freshwater inputs to this portion of the bay. Figure 5 conceptually illustrates this project.



Figure 5. Conceptual plan for the Lake Tarpon Outfall Canal habitat enhancement project.

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RESTORED HABITATS: LESSONS LEARNED AND THE IMPORTANCE OF MANAGEMENT

T.F. Ries

ABSTRACT

The science and implementation of habitat restoration is comprised of various methods of reestablishing ecological functions to altered or disturbed habitats. Sometimes this includes restoring the landscape to its historic conditions; known as in kind and in place; however, many times it entails converting an altered and severely disturbed habitat into a functional ecosystem to address the historic loss of specific habitats for a region. In either case, the techniques necessary to achieve a functional ecosystem will vary from site to site; however, the mechanical processes are similar. This paper summarizes many years of design and application covering thousands of acres of habitat restoration experience into a catalog of lessons learned. The intent is to pass on the information so others can capitalize on the experiences gained by monitoring various restoration techniques. Many of these 'lessons' were gained by observing the resultant restoration site and examining what worked and why. Most advancements were achieved by examining what did not work as planned and thoroughly scrutinizing what actually transpired onsite. A summary of these observations and hypotheses are presented herein.

INTRODUCTION

The science and implementation of habitat restoration is comprised of various methods of establishing or reestablishing ecological functions to altered or disturbed habitats. Many times this includes restoring the landscape to its historic conditions; however, in other instances it entails converting an altered and severely disturbed habitat into a different, functional ecosystem to address the historic loss of specific habitats for a region. In either case, the techniques necessary to achieve a functional ecosystem will vary from site to site; however, the mechanical processes are similar. Habitat restoration design can be a very complicated undertaking as there are many variables to take into consideration when designing habitat restoration projects. Typically restoration projects are located within disturbed or altered landscapes which are not providing their historic ecological functions. The restoration practitioner must assess a site's historic records and have a thorough understanding of the existing and desired ecosystem to begin the design process. Frequently the goals of the restoration plan are not achieved; even these situations allow the designer to better understand what works and why. These, sometimes hard learned, teachings have resulted in a summary of lessons garnered from field observations. The following list of design considerations was developed over decades of implementing numerous methodologies and practices, covering thousands of acres of restoration experience. Although each restoration site is unique in itself, many of these lessons can be applied to a variety of habitat restoration scenarios. The following list of considerations is not ordered in a specific priority setup, it merely lists items to be aware of when planning a habitat restoration project.

- Mimic natural systems—the primary objective, which can be difficult to achieve, should be to replicate functioning ecosystems; however, by observing unaltered systems, the landscape components can be identified and included within the development of the restoration design.
- Habitat mosaics concept—this concept is based upon the observation of natural systems and noting that many are comprised of a variety of micro ecosystems interacting to complete a functioning landscape. Therefore, when designing a habitat restoration project the design should include a variety of habitats ranging from uplands to aquatic features.

- Wealth of complexity—natural systems are generally quite complex and include micro topography and longitudinal variations. The design of restoration sites should, as much as possible, include similar complexities. Years ago extremely complex designs were not considered as they were thought to be too difficult to construct; however, with experienced site contractors very intricate designs have been successfully constructed in the field. Incorporate as much ecotonal edge as possible and including micro topographical elements to increase the design complexity.
- Understand the site conditions—it is imperative to be intimately knowledgeable of the historic and current site conditions. This can be accomplished by mapping the existing native vegetation, obtaining detailed topographic surveys, perception of archeological resources, and acquiring geotechnical data. With this data the restoration design can be formulated to work with the existing altered and impacted conditions incorporating the existing natural habitat.
- Keep control structures designs simple—there are numerous tactics that can be taken to achieve structural operation within restoration designs. The simple solutions typically are the better due to reduced construction costs and long term maintenance requirements. For example, well-constructed, broad-crested weirs are superior to concrete flumes (if the anticipated flow velocities permit this application). Therefore, whenever possible select simple, non-structural methods to achieve the overall goals.
- Design for and install the pioneer species—Planting the climax community doesn't always work as intended. Allow natural succession to occur to achieve your climax vegetative state. It is far less expensive to install the pioneer species and allow natural plant succession processes to take place to achieve a thriving vegetative community.
- "Restoring the Historic Balance"—this approach recognizes that some habitats have been lost in greater proportion than others, and that these habitats may be critical for specific guilds of habitat-dependent species. It is important to thoroughly understand the regional and historic changes that have occurred in the region so that the restoration design incorporates the environmental components necessary to address historic, disproportional impacts.
- Incorporate stormwater retrofit opportunities into the habitat restoration design—every habitat restoration design should try to integrate or promote stormwater treatment elements. These can be simple passive biological treatment components such as sediment sumps and longer treatment flow-ways, to capture sediments loads and promote the natural assimilation of nutrients within the project.
- Limit the number of hydraulic passes/connections—although complex designs and extensive edge creation are encouraged, the number of hydraulic connections to adjacent water bodies should be limited to less than one per every ten acres of created wetland. Early designs typically included numerous passes to increase edge and provide sufficient hydrology; however, over time many of these passes accumulated sediments and ceased providing the designed hydraulic ties to the wetland community. By limiting the number of hydraulic connections, the resultant connections become self-maintaining systems.
- Limit the number of upland islands within open water landscapes—although the incorporation of island features within open water features is encouraged, it must be recognized that access to these features will typically limit maintenance activities. Island

elements should be included only if a strong and dedicated effort is available to perform the necessary initial maintenance activities to allow these features to become self-sustaining.

- Endorse early public involvement—although a project may have or is capable of obtaining the proper permits and authorizations; without obtaining the public's interest the project may not obtain the public support necessary to implement the project. All projects should encourage public involvement from the onset; by including the public's concerns the project will have greater success obtaining the necessary support and endorsement of the neighborhood, community, and the public in general.
- Regulatory pre-application meetings—regulatory authorization is necessary for all projects and this process can be slow. Thus, it is imperative to solicit early and frequent regulatory coordination to facilitate the review and approval processes. Coordination includes pre-application meetings, design comment sessions, and a post-project field trip with the regulatory staff to demonstrate the objectives. All of these steps have been demonstrated to accelerate and improve the regulatory review and approval processes.
- "Living shoreline" concept—projects that include shorelines, especially areas with wave energy, should be designed to promote living shorelines versus hardened structures or bulkheads. There are many examples of shoreline design alternatives that incorporate native vegetation, rather than vertical wall or similar structures that will ultimately provide the necessary shoreline protection objectives desired.
- Design for sea level rise—all coastal projects should consider sea level rise in the design of restoration areas. This includes designing more gradual slopes and incorporating fewer low marsh components and more high marsh areas to allow the vegetative community to naturally migrate upslope as sea level elevations change over time.
- Transplantation of native vegetation—many plant species can be successfully transplanted, when proper pruning practices are employed, thus providing immediate structure within the restoration area. Transplantation costs are typically higher than planting young nursery-grown specimens; however, by transplanting trees the temporal losses associated with the three-dimensional features is greatly reduced.
- Utilize historic aerials—these resources are invaluable in understanding a site's former attributes and alterations, which will assist in drafting the restoration plan. Historic aerials have made it possible to evaluate former features and incorporate them into the design plan. This has resulted in the appropriate features being placed as they historically existed and functioned, within their original and natural state.
- Utilize GPS technology—today's differentially-corrected global-positioning tools provide designers the opportunity to review the initial design in the field and make refinements to the design plan in real time. These tools also allow contractors to 'load' the construction plans into the cab of the heavy equipment to facilitate the actual construction of the project. GPS can also ensure that the vertical grades are being met as envisioned; these elevations are critical to the survival of the plants and must be constructed as designed for the site to function as intended.
- Construction oversight—the best design plans may result in a functioning ecosystem; however, superior results can be achieved by having the design ecologist onsite to supervise the most critical component of the project - the construction phase. Minor

design alterations can be directed by the designer which will result in a superior restoration project. The restoration design process requires a hands-on approach from the initial design to the installation of the native vegetation. This will result in an award-winning project. But more importantly, it will result in a functioning ecosystem that meets the goals and objectives of the restoration plan.

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LONG-TERM MONITORING AND ASSESSMENT OF NATIVE AND RESTORED HABITATS: THE PAST, PRESENT, AND POSSIBLE FUTURE

B.Weigle

A continuum of habitats and salinity gradients, including sea grass beds and emergent tidal wetlands (ETWs), is needed to support life histories of estuarine dependent species. The Southwest Florida Water Management District maps sea grass distribution and change on a predefined schedule, usually every two years. However, monitoring of the ETWs is not consistent enough or conducted on enough sites to quantify small-scale changes over short periods. To assess these small changes, previous studies need to be reviewed for accuracy and new monitoring sites need to be established around Tampa Bay that are routinely evaluated using the best possible survey and mapping techniques. A combination of data from field sites and additional vegetation signature sites can be utilized to facilitate automated mapping processes that are standardized, non-biased, and repeatable.

Mapping of coastal and benthic features has traditionally been accomplished by viewing overlapping photographic prints with a stereoscope. Imagery collection progressed from black and white photos in the 1930s to color and color infrared photos in the 1960s. Vegetation polygons were originally digitized from the photographs onto mylars and then transferred to another base map for analysis. Beginning in the 1990s, polygons viewed in a stereoscope were digitized into a geographic information system (GIS) using the resource photography on one meter resolution digital orthophotos quarter quads (DOQQs) as a background. Analyses were then conducted in a GIS system using automated classification techniques.

Historic black and white images can be used to map the past. Beginning in the 1930s, the US Department of Agriculture collected black and white stereo photographs for the entire country, revisiting counties on a five- to seven-year cycle. Prints and/or diapositives of these images can still be obtained through the National Archives. After scanning and rectification, the digital images with a resolution of one-half meter have been successfully used to create historic benthic maps for the Charlotte Harbor and Sarasota Bay National Estuary Programs. These data can be used to establish historic baselines for sea grass, oyster beds, and tidal wetlands.

Digital aerial acquisition systems have been developed which are rapidly replacing the collection of film-based imagery. New digital camera systems acquire three natural color bands (red, green, and blue) along with a fourth infrared band. Image resolution on the ground is still determined by the flying height of the plane and many government agencies are acquiring imagery at one-foot or six-inch pixel resolution. Frame-based digital mapping cameras collect images with a 60% overlap along the flight line and 30% sidelap with the adjacent flight line, allowing for the delineation and classification of features in stereo (3D) onscreen.

Automated processing of imagery to extract vegetation features has advanced with the advent of digital imagery. Many projects over large areas still utilize 30-meter spatial resolution Landsat imagery because of the six available bands that can be used for multiple comparisons to map dominant vegetation using measures for wetness, greenness, and normalized difference vegetation index. Analysis techniques consist of software packages that utilize an object-oriented approach of image segmentation along with a variety of classification methodologies to model vegetation dominance type, canopy cover class, and size class. However, the 30-meter resolution of Landsat imagery precludes mapping at the finer scales needed to detect changes in the coastal environment.

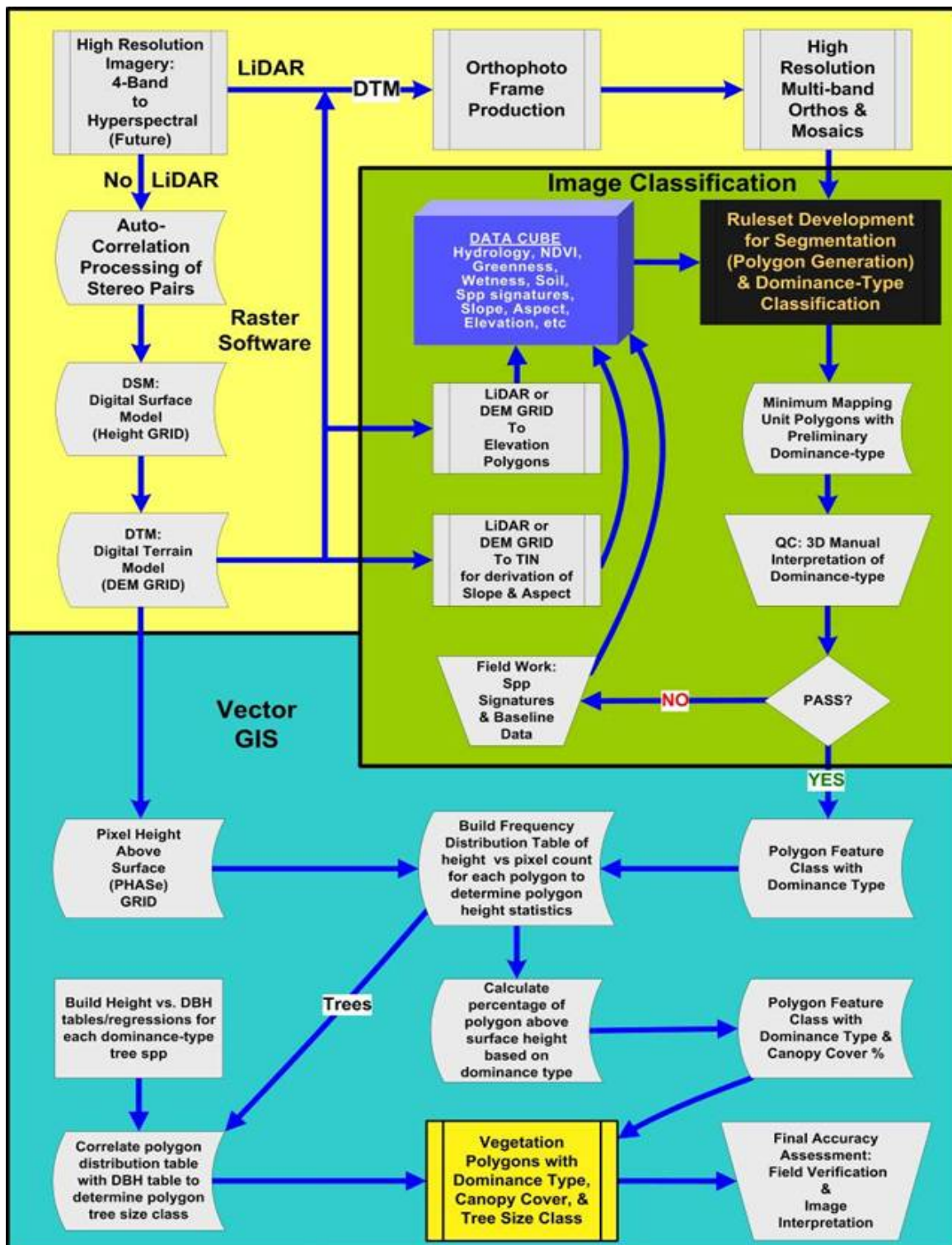
The effective use of automated techniques to derive vegetation layers from 4-band six- inch and one-foot spatial resolution imagery is progressing rapidly and should become a mature analytical tool in the near future. Field collection of definitive vegetation signatures at a variety of site locations is essential for training and fine-tuning any automated classification model. Autocorrelation techniques using stereo imagery can yield vegetation relative height through software such as Socet Automated Terrain Generation and ERDAS Automated Terrain Extraction. When analyzed in conjunction with recent LiDAR data which accurately maps the ground surface, a vegetation height layer could be determined.

Long term monitoring using automated vegetation mapping techniques will require a number of new data sets and technological advances that are also nearing product maturity to enable cost-effective implementation. Examples of these required datasets and technologies include: spectral signature libraries for vegetation map units and developmental phases; high-resolution 8-to 64+-band hyperspectral imagery; widespread LiDAR data acquisition/availability to derive ground surface and canopy layers or improved auto-correlation software to derive these layers from stereo imagery; and desktop supercomputers with petabyte hard drives to support large, multi-year image catalogs. Computer software packages; e.g. Definiens, ERDAS Objective, Feature Analyst, Socet GXP) are already available to automatically process the imagery into homogenous polygons based on spectral characteristics, elevation, and texture but classification to the species level using robust, proven rulesets is still a developing science. Even as these elements develop, the drastic price reduction of stereo computer hardware (from \$50K to \$1-2K) has made on-screen use of imagery in 3D available to most GIS and scientific staff. Government agencies acquiring high resolution imagery need to include specifications for the delivery of all the information required to utilize the stereo imagery for multiple uses in addition to currently-specified orthophotos and mosaics.

As with all studies involving remote sensing, field accuracy assessments need to be conducted to validate classification by automated modeling or by human photo interpretation. Intensive field verification is necessary during initial deployment of automated procedures to ensure that computer models are working effectively. Reduced levels of field verification can be expected after confidence in automated classification has been established. However, ground-level monitoring of permanent sites should be implemented immediately to document community changes in response to climate changes and other stressors that cannot currently be measured using remote sensing techniques. To this end, a minimum of five permanent study areas should be established, with at least one study site in each primary bay segment. This plan would consist of fixed stations with GPS-established monuments and established quadrats that are regularly monitored to determine plant composition and cover along with interstitial salinity.

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Figure 1: A Potential Process for Automated Mapping of Vegetation Using High Resolution Imagery



ABSTRACTS

Session 6: ECOSYSTEM RESTORATION & MANAGEMENT

PRESENTATIONS

Land Acquisition and Management in the Tampa Bay Watershed – Past, Present, and Future. F. Turbiville (Hillsborough County). Land Acquisition programs within the Tampa Bay region have played a major role in the maintenance and recovery efforts of habitat and water quality within Tampa Bay and its contributing watersheds. Florida currently has 34.5 million acres of non-sovereign land. Of this amount, approximately 8.8 million acres have been converted to urban or otherwise altered uses. Given that an estimated 250,000 acres of natural habitat are converted each year to other land uses, it is feasible that within 99 years the only remaining natural habitat will be what has been purchased and/or protected from development.

Within the Tampa Bay watershed, natural habitat has been acquired, managed, and restored through a variety of public and private agencies/programs. Public agencies/programs include: the Southwest Florida Water Management District (SWFWMD), Florida Communities Trust (FCT) Program, Conservation Area and Recreational Lands (CARL) Program, Florida Department of Environmental Protection (FDEP), Hillsborough County Environmental Lands Acquisition and Protection (ELAP) Program, Pinellas County Board of County Commissioners, Manatee County Board of County Commissioners, Florida Department of Transportation (FDOT), Florida Fish and Wildlife Conservation Commission (FFWCC), and United States Fish and Wildlife Service (USFWS). Private agencies/programs include: The Nature Conservancy, Trust for Public Land, and Tampa Bay Conservancy.

Hillsborough County, through its ELAP Program and partnering agencies (e.g. SWFWMD, FCT), has acquired approximately 44,700 acres of environmentally sensitive lands, of which it manages 40,500 acres. Pinellas County currently manages 14,340 acres, while Manatee County manages approximately 29,000 acres. SWFWMD manages roughly 35,000 acres within all three counties combined. The FFWCC and FDEP also manage several thousand acres of uplands and wetlands located along Tampa Bay and the interior sections of all three bay area counties.

Each agency has protected a variety of uplands and wetlands within the Tampa Bay watershed. Uplands consist primarily of pine flatwoods, sandhill, xeric oak/sand pine scrub, and mesic oak hammock. Wetlands include depression marsh, floodplain forest, mangrove/salter and seagrass communities. Each habitat type has its own unique management requirements. Management of protected lands falls into six general categories: habitat restoration, prescribed burning, exotic plant control, feral animal control, floral/faunal monitoring, and public access/recreation. Each agency faces its own unique management challenges, from transforming disturbed lands into viable wildlife habitat to conducting prescribed burns within the urban interface. Exotic plant control, in particular, has emerged as one of the greatest challenges facing land managers. The rapid spread of invasive exotic plants, combined with a lack of financial resources to provide consistent, long-term eradication efforts, threatens nearly all publicly-owned and managed lands within the Tampa Bay watershed.

The continued success of land acquisition efforts within the Tampa Bay watershed depends upon appropriate levels of state, federal and local funding for acquisition along with adequate funding set aside to maintain existing and newly-acquired lands. [Turbiville@hillsboroughcounty.org]

The Watershed Approach to Ecosystem Restoration – Linking Bay and Watershed Habitats for Tampa Bay (Florida). B.F. Henningsen (Surface Water Improvement and Management Program Southwest Florida Water Management District). Destruction and degradation of various coastal habitats of Tampa Bay have been well documented (see this and the previous four BASIS proceedings). As also noted in these proceedings, restoration efforts for Tampa Bay began during 1971, continue through the present, and are proposed for the future. Restoration projects initially were small, simple efforts representing fractions of acres planted with mangroves. By the 1990s, with improved ecological vision and public funding, projects had become larger, increasingly complex interdisciplinary efforts. Restoration projects moved from being single habitat efforts (e.g., salt marsh) to true, large-scale, ecosystem restoration endeavors.

Ecosystem restoration projects, sometimes in the hundreds of acres, have addressed the enhancement/restoration/creation of mosaics of habitats inclusive of various estuarine, freshwater, and upland communities. As feasible, project elements (e.g., stormwater polishing) also have addressed improvements in water quality for the bay and restored/allowed natural processes (e.g., sheetflow, ecological succession, etc.). In addition, restoration projects of the Surface Water Improvement and Management (SWIM) Program of the Southwest Florida Water Management District have attempted to address potential impacts from projected sea level rise for the bay; projects containing liberal amounts of high marsh and transitional zones may allow emergent wetland communities to migrate upslope with sea level rise. Scientific studies and frequent field observations indicate that restoration projects are providing quality habitats for a myriad of plant and animal species for the estuary; wildlife utilization of restored habitats appears widespread (e.g., oysters, blue crab and other crustaceans, sea hares, horseshoe crabs, mullet, snook, redfish, marsh minnows, wading birds, raptors, amphibians, reptiles, mammals inclusive of manatee, bottlenose dolphin, and otter, etc.). Unfortunately, the vast majority of restoration projects for Tampa Bay have only been in the coastal reaches of the estuary, typically not more than one mile inland. Ecological processes of Tampa Bay are dependent upon the estuary's connectivity to its watershed.

With a watershed of over 2200 square miles, ecosystem restoration efforts for Tampa Bay should be performed throughout the watershed, not just in the coastal areas of the estuary. Expansion of restoration efforts up the watershed is especially timely due to decreasing availability of coastal tracts for restoration and the need to plan for the potential effects of sea level rise (i.e., estuarine habitats could "migrate" up the watershed). For all restoration efforts, land availability is a limiting factor, and public land acquisition programs with limited funds have made impressive acquisitions for the bay and portions of the watershed. Almost all existing restoration projects have been performed on public land, but funds for public land acquisition are inadequate to purchase sufficient land to meet restoration goals. To help meet restoration goals for the bay, public-private partnerships should be explored to facilitate the use of private lands throughout the watershed for restoration projects. Several examples of public-private partnerships (i.e., Palmetto Estuary and Newman Branch restoration projects) provide evidence that partnerships can be successfully implemented to the advantage of the private landowner, the public, and the environment.

Another management consideration for the bay is the density of the human population within the watershed. The human population of the watershed must be considered for the long-term proper management of the resource. Although politically and socially volatile, discussions eventually may need to be held discussing the feasibility of establishing a carrying capacity for humans within the Tampa Bay watershed to insure optimal quality of life for humans and the estuary (i.e., a balanced ecosystem approach).

Lastly, to date, the success that has been experienced in the rehabilitation of the Tampa Bay estuarine ecosystem is the end product of decades of groundwork by various public and legislative parties. With the accelerating turnover of the "baby boomer" generation that, in major part, has spearheaded bay restoration and management efforts, it is paramount for emerging restoration ecologists and bay managers to continue the momentum established by their predecessors. [Brandt.Henningsen@swfwmd.state.fl.us]

BENTHIC MACROINVERTEBRATES OF TAMPA BAY REVISITED

D.J. Karlen, T.L. Dix, B.K. Goetting & S.E. Markham

ABSTRACT

Simon and Mahadevan presented a literature review of benthic macroinvertebrate studies in Tampa Bay at the first Bay Area Scientific Information Symposium (BASIS) in 1982. Based on their review they made several general conclusions on the spatial and temporal patterns of the Tampa Bay benthic community and estimated that approximately 1,200 infaunal and epifaunal species were found in the bay. They also made several recommendations for future studies, including the need for “quantitative, comprehensive, and long-term studies of the Bay benthos to better understand (and formulate management plans for) the long-term effects of urban runoff, domestic wastes and industrial discharges to the Bay.” This recommendation has since been met with the initiation of the Tampa Bay Estuary Program’s Bay-wide Benthic Monitoring program in 1993. Since the start of the Bay-wide Benthic Monitoring Program over 1,450 sediment grab samples have been collected throughout Tampa Bay and close to 2,000 additional samples have been collected as part of other monitoring efforts in Tampa Bay and its surrounding tributaries. Analyses of these combined datasets are used here to evaluate the species diversity of benthic infaunal invertebrates in the Tampa Bay system. Over 1,440 macroinvertebrate taxa have been identified from 3,412 samples collected from 1993-2008; eight taxa accounted for >25% of the overall benthic abundance. The brachiopod *Glottidia pyramidata* was the most abundant species overall, which accounted for 5% of the total benthic abundance. The other numerically dominant taxa included the polychaete *Monticellina* cf. *dorsobranchialis*, the cephalochordate *Branchiostoma floridae*, the amphipods *Ampelisca holmesi*, *Apocorophium louisianum*, *Grandidierella bonnieroides*, and *Ampelisca abdita*, and the bivalve *Mysella planulata*. The distribution of species was influenced by salinity regimes and sediment type. Results from the monitoring program have confirmed several of the observations made by Simon and Mahadevan, including the trend of increasing species richness towards the mouth of the Bay and the influence of sediment type in controlling the distribution of benthic species. Long-term trends in the benthic macroinvertebrate community in Tampa Bay since the first BASIS meeting appear to indicate general improvements, although isolated areas of habitat degradation still persist.

INTRODUCTION

Simon and Mahadevan presented a literature review of all published benthic macroinvertebrate studies conducted in Tampa Bay up to that date (approximately 70 published studies) at the first BASIS in 1982. They presented several conclusions on the spatial and temporal patterns of the Tampa Bay benthic community based on their findings:

- 1) an estimate of approximately 1,200 infaunal and epifaunal species in Tampa Bay;
- 2) seasonal fluctuations in diversity and abundance of benthic organisms;
- 3) regular periods of defaunation of benthic communities on an approximately 5-year cycle;
- 4) decreasing faunal diversity due to declines in seagrass beds in the bay;
- 5) an abundance of “pollution indicator” species particularly in Hillsborough Bay;
- 6) faunal distributions were controlled mainly by sediment type;
- 7) a general trend of increasing species richness and decreasing abundance on a north to south gradient (i.e., towards the mouth of the bay).

Simon and Mahadevan also made several recommendations for future studies of the Tampa Bay benthic community and recognized the need for “quantitative, comprehensive, and long-term studies of the Bay benthos to better understand (and formulate management plans for) the long-term effects of urban runoff, domestic wastes and industrial discharges to the Bay.”

This recommendation has since been addressed by the initiation of the Tampa Bay Estuary Program's Bay-wide Benthic Monitoring Program in 1993, which has been conducting annual surveys of the benthic community up to the present time. In addition to the Bay-wide monitoring program, the Environmental Protection Commission of Hillsborough County (EPCHC) established the Hillsborough Independent Monitoring Program (HIMP) in 1999 to monitor the major tributary systems of Tampa Bay for potential impacts due to surface water withdrawals for potable water supply. The HIMP program collected benthic samples annually in the Hillsborough, Palm, Alafia, and Little Manatee rivers and McKay Bay from 1999 until 2006. There have been several other short term projects funded through the Tampa Bay Estuary Program that have included surveys of benthic macroinvertebrates. The Dredged Hole Project (TBEP 2005) surveyed several dredged borrow pits in Tampa Bay to evaluate their use as fish habitats and the feasibility of restoration efforts. A second TBEP-sponsored study focused on the smaller tidal tributaries and their function as fish nursery habitat (TBEP 2008). Both of these studies, along with the HIMP sampling, added much additional information on the benthic macroinvertebrate communities in Tampa Bay, particularly in lower salinity and tidal freshwater habitats which were not well represented in the TBEP Bay-wide monitoring program.

METHODS

Species count and occurrence data were compiled from the EPCHC's benthic database and comprised 3,412 samples collected from the TBEP Bay-wide Benthic Monitoring Program from 1993-2008, the HIMP from 1999-2006, the TBEP's Dredged Hole study (TBEP 2005) and Tidal Stream Initiative (TBEP 2008). The field and laboratory procedures followed the protocols set forth in Courtney et al. (1995). The compiled taxonomic list was evaluated and taxonomic categories which could potentially represent several species were removed from the final species count. However, the abundance values for the removed taxa were included in relative abundance calculations.

RESULTS AND DISCUSSION

The initial database query of 3,412 samples consisted of 1,892 taxa, 450 of which were removed from the analysis. The taxa were removed for two reasons: they either represented higher level taxonomic groupings which encompassed more than one species or there was uncertainty in the taxonomic identification due to the condition of the specimens. Over a quarter of the species (378 or 26.21%) were only present in a single sample. There were 240 species (16.64%) represented by a single specimen.

The 1,442 species represented 15 phyla (Table 1) with over 90% of the species falling within three phyla. The annelids (segmented worms) were the most diverse with 504 species, followed by the arthropods and mollusks (Table 1).

Within the annelids 433 species (86%) were polychaetes. Oligochaetes were represented by 64 species and leeches (Hirudinea) were represented by 7 species (Figure 1). The polychaetes were dominant in higher salinities, while the oligochaetes and leeches were dominant in lower salinities and freshwater. The polychaete *Monticellina* cf. *dorsobranchialis* was the most abundant annelid and also had the highest frequency of occurrence among the annelids (Table 2).

The arthropods were the second most species-rich phylum with 421 species accounting for 29% of the total species found in this study (Table 1). The majority of arthropod species (308 = 73%) were crustaceans, while aquatic insects represented one quarter of the species (Figure 2). Crustacean taxa were abundant across all habitats, but were particularly dominant in estuarine areas while insects were primarily confined to low salinity and freshwater sites. Amphipods had the most species (123 species) within the crustaceans, followed by the decapods with 111 species (Figure 3). The amphipod *Ampelisca holmesi* was the most abundant and most frequently-

occurring arthropod (Table 3). The 10 most abundant arthropod taxa were dominated primarily by amphipods with three species falling within the genus, *Ampelisca* (Table 3). Only one insect taxon, the *Polypedilum halterale* group of chironomids, was among the most abundant arthropods and was predominately found in freshwater habitats.

The mollusks were represented by 385 species, of which 226 were gastropods and 146 were bivalves (Figure 4). The small clam *Mysella planulata* was the most abundant mollusk found in Tampa Bay and the most frequently occurring mollusk species (Table 4). Seven of the ten most abundant mollusks species were bivalves (Table 4).

For the complete dataset, eight species represented over 25 % of the overall macrofaunal abundance in Tampa Bay (Table 5). Half of the eight most abundant species were amphipods while annelids and mollusks were each represented by a single species. The most abundant species was brachiopod *Glottidia pyramidata*, (Table 5), the only representative of the phylum Brachiopoda in Tampa Bay. The cephalochordate *Branchiostoma floridae* ranked third in overall abundance and was one of the few invertebrate chordate species found in Tampa Bay.

A total of 21 species were found in 20% or more of the samples and five of these species occurred in 30% or more of the samples (Table 6). The most frequently found species in Tampa Bay included eight polychaetes, six crustaceans (including four amphipods), four mollusks, *Glottidia pyramidata*, *Branchiostoma floridae*, and the ribbon worm (Nemertea) *Amphiporus bioculatus* (Table 6). The most frequently occurring species in Tampa Bay was the polychaete *Monticellina* cf. *dorsobranchialis*. Several of the most frequently occurring species were not among the most abundant, but they were wide spread across many habitats and salinity zones throughout Tampa Bay.

The spatial distribution of individual species and the overall benthic community composition was largely influenced by the sediment type. The distributions of *Branchiostoma floridae* and *Monticellina* cf. *dorsobranchialis* in Tampa Bay are illustrated by Figure 5. *Branchiostoma floridae* is a filter feeder and inhabits coarser-grained sand and shell sediments, which is reflected in its higher abundances in Middle and Lower Tampa Bay. In contrast, *Monticellina* cf. *dorsobranchialis* is a deposit feeder and its highest abundances are found in areas with silty sediments such as Hillsborough Bay. The overall benthic community composition, expressed as a Multi-Dimensional Scaling (MDS) ordination of species similarity, is shown in Figure 6 with the species richness versus the % silt+clay superimposed on the MDS plot (Figure 6). The two plots show that benthic species richness is generally higher at sites with lower % silt+clay and illustrate the influence of sediment composition on the overall benthic community structure.

Salinity was also an important factor influencing species distributions and population trends. The abundance of the bivalve *Mysella planulata* in Hillsborough Bay, along with the trend in bottom salinity, show temporal changes due to salinity (Figure 7). This trend is also apparent in the overall abundance of benthic macrofauna, particularly in Hillsborough Bay.

Tampa Bay and its tributaries have a rich benthic macroinvertebrate community with over 1,442 species. It should be noted that this species count is based exclusively on sediment grab samples and thus is biased for smaller, infaunal taxa. Larger, more motile species and epifaunal taxa are certainly under-represented in the database and the actual species richness of macroinvertebrates in Tampa Bay is most likely much higher than reported here. Sample collections were also primarily done in the late summer (August – September); therefore seasonal patterns in macrofaunal abundances are also under-represented. Despite the large number of species found, relatively few species dominate the benthic community of Tampa Bay, with only eight taxa accounting for 25% of the total abundance. Over a quarter of the species reported had only a

single occurrence in 3,400 samples collected over a 16-year period. Simon and Mahadevan had estimated that there were around 1,200 benthic macroinvertebrate species in Tampa Bay based on their review of around 70 published papers and technical reports. Most of the studies that they looked at were relatively short-term studies which focused on localized sites or specific habitat types within Tampa Bay. Their dataset however does have a better representation of some of the larger macrofauna species and of seasonal patterns of macrofaunal populations than the current dataset. Our data does concur with several of Simon and Mahadevan's observations, particularly with regard to the influence of sediment type on the macroinvertebrate community and the general increase in species richness towards the lower part of the bay.

The higher number of taxa found in the current study may be due to the higher sampling intensity and longer duration of our monitoring programs. The higher species count may however, also be seen as an indication of improvements in Tampa Bay since the first BASIS. Simon and Mahadevan noted in their study a long-term trend in degradation of the macroinvertebrate community including periodic defaunation due to hypoxia and decreasing faunal diversity due to reductions in seagrasses. Many of these conditions have since improved throughout the bay due to better management practices and restoration efforts. Periodic defaunation of the benthic community has not been seen in Tampa Bay over the course of the current monitoring program and there has been a substantial gain in seagrass coverage over the past 20 years. Despite these improvements there are still many areas which need to be addressed in meeting the goals to restore Tampa Bay, and the benthic macrofauna community will continue to be an important component of Tampa Bay's ecology as well as an important tool in monitoring the bay's health.

ACKNOWLEDGEMENTS

This paper is dedicated in memory of Dr. Joseph L. Simon (1937-2004). Dr. Simon was a professor in the Biology Department at the University of South Florida for over 38 years. He was a pioneer in the study of the benthic ecology of Tampa Bay and also served as one of the organizers of the first BASIS meeting.

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Table 1. Number and percentage of benthic macroinvertebrate species in Tampa Bay by phylum.

Phylum	Number of species	%
Annelida	504	35.0%
Arthropoda	421	29.2%
Mollusca	385	26.7%
Cnidaria	30	2.1%
Echinodermata	29	2.0%
Nemertea	22	1.5%
Bryozoa	15	1.0%
Platyhelminthes	9	0.6%
Porifera	8	0.6%
Chordata	8	0.6%
Sipuncula	6	0.4%
Phoronida	2	0.1%
Echiura	1	0.1%
Brachiopoda	1	0.1%
Hemichordata	1	0.1%

Table 2. Relative abundance and frequency of occurrence for the ten most abundant annelids.

Species (Classification)	Relative Abundance (within Phylum) N = 393,106	Relative Abundance (all infauna) N = 1,046,548	Frequency of Occurrence
<i>Monticellina dorsobranchialis</i> (Polychaeta)	10.48%	3.94%	33.94%
<i>Streblospio cf. gynobranchiata</i> (Polychaeta)	4.79%	1.80%	22.30%
<i>Laonereis culveri</i> (Polychaeta)	4.27%	1.60%	21.98%
<i>Prionospio perkinsi</i> (Polychaeta)	3.76%	1.41%	31.92%
<i>Paraprionospio pinnata</i> (Polychaeta)	3.60%	1.35%	32.12%
<i>Carazziella hobsonae</i> (Polychaeta)	3.48%	1.31%	16.15%
<i>Fabricinuda trilobata</i> (Polychaeta)	3.02%	1.13%	10.32%
<i>Stenoninereis martini</i> (Polychaeta)	3.02%	1.13%	12.66%
<i>Tubificoides brownie</i> (Oligochaeta)	2.39%	0.90%	14.80%
<i>Aricidea philbinae</i> (Polychaeta)	2.37%	0.89%	16.50%

Table 3. Relative abundance and frequency of occurrence for the ten most abundant arthropods.

Species (Classification)	Relative Abundance (within Phylum) N = 285,516	Relative Abundance (all infauna) N = 1,046,548	Frequency of Occurrence
<i>Ampelisca holmesi</i> (Crustacea: Amphipoda)	11.28%	3.08%	31.98%
<i>Apocorophium louisianum</i> (Crustacea: Amphipoda)	10.86%	2.96%	3.90%
<i>Grandidierella bonnieroides</i> (Crustacea: Amphipoda)	9.89%	2.70%	21.07%
<i>Ampelisca abdita</i> (Crustacea: Amphipoda)	8.86%	2.42%	22.42%
<i>Rudilemboides naglei</i> (Crustacea: Amphipoda)	5.74%	1.57%	13.10%
<i>Ampelisca vadorum</i> (Crustacea: Amphipoda)	2.78%	0.76%	8.85%
<i>Cerapus</i> sp. C of LeCroy, 2007 (Crustacea: Amphipoda)	2.74%	0.75%	9.44%
<i>Xenanthura brevitelson</i> (Crustacea: Isopoda)	2.68%	0.73%	12.02%
<i>Polypedilum halterale</i> grp. (Insecta: Chironomidae)	2.52%	0.69%	4.34%
<i>Cyathura polita</i> (Crustacea: Isopoda)	2.30%	0.63%	16.91%

Table 4. Relative abundance and frequency of occurrence for the ten most abundant mollusks.

Species (Classification)	Relative Abundance (within Phylum) N = 230,032	Relative Abundance (all infauna) N = 1,046,548	Frequency of Occurrence
<i>Mysella planulata</i> (Bivalvia)	13.29%	2.92%	33.18%
<i>Amygdalum papyrium</i> (Bivalvia)	8.45%	1.86%	20.22%
<i>Mytilopsis leucophaeata</i> (Bivalvia)	7.82%	1.72%	10.81%
<i>Caecum strigosum</i> (Gastropoda)	7.67%	1.69%	8.88%
<i>Mulinia lateralis</i> (Bivalvia)	5.46%	1.20%	17.32%
<i>Parastarte triquetra</i> (Bivalvia)	4.39%	0.96%	7.39%
<i>Acteocina canaliculata</i> (Gastropoda)	3.25%	0.72%	23.30%
<i>Pyrgophorus platyrachis</i> (Gastropoda)	2.46%	0.54%	3.02%
<i>Nucula proxima</i> (Bivalvia)	2.17%	0.48%	17.17%
<i>Angulus</i> cf. <i>versicolor</i> (Bivalvia)	1.95%	0.43%	21.01%

Table 5. Eight most abundant infaunal macroinvertebrates representing >25% of the total cumulative abundance in Tampa Bay.

Species		Relative Abundance (N = 1,046,548)	Cumulative Abundance
<i>Glottidia pyramidata</i> (Brachiopoda)		5.06%	5.06%
<i>Monticellina dorsobranchialis</i> (Annelida: Polychaeta)		3.94%	8.99%
<i>Branchiostoma floridae</i> (Chordata: Cephalochordata)		3.12%	12.12%
<i>Ampelisca holmesi</i> (Arthropoda: Crustacea: Amphipoda)		3.08%	15.20%
<i>Apocorophium louisianum</i> (Arthropoda: Crustacea: Amphipoda)		2.96%	18.16%
<i>Mysella planulata</i> (Mollusca: Bivalvia)		2.92%	21.08%
<i>Grandidierella bonnieroides</i> (Arthropoda: Crustacea: Amphipoda)		2.70%	23.78%
<i>Ampelisca abdita</i> (Arthropoda: Crustacea: Amphipoda)		2.42%	26.19%

Table 6. Most frequently occurring Tampa Bay macroinvertebrates.

Species	Freq. of Occurrence (n = 3412 samples)
<i>Monticellina dorsobranchialis</i> (Annelida: Polychaeta)	33.94%
<i>Mysella planulata</i> (Mollusca: Bivalvia)	33.18%
<i>Paraprionospio pinnata</i> (Annelida: Polychaeta)	32.12%
<i>Ampelisca holmesi</i> (Arthropoda: Crustacea: Amphipoda)	31.98%
<i>Prionospio perkinsi</i> (Annelida: Polychaeta)	31.92%
<i>Podarkeopsis levifuscina</i> (Annelida: Polychaeta)	28.63%
<i>Amphiporus bioculatus</i> (Nemertea)	28.49%
<i>Glottidia pyramidata</i> (Brachiopoda)	24.24%
<i>Glycinde solitaria</i> (Annelida: Polychaeta)	23.74%
<i>Acteocina canaliculata</i> (Mollusca: Gastropoda)	23.30%
<i>Branchiostoma floridae</i> (Chordata: Cephalochordata)	23.27%
<i>Ampelisca abdita</i> (Arthropoda: Crustacea: Amphipoda)	22.42%
<i>Streblospio cf. gynobranchiata</i> (Annelida: Polychaeta)	22.30%
<i>Amakusanthura magnifica</i> (Arthropoda: Crustacea: Isopoda)	22.27%
<i>Laeonereis culveri</i> (Annelida: Polychaeta)	21.98%
<i>Grandidierella bonnieroides</i> (Arthropoda: Crustacea: Amphipoda)	21.07%
<i>Cyclaspis cf. varians</i> (Arthropoda: Crustacea: Cumacea)	21.07%
<i>Angulus cf. versicolor</i> (Mollusca: Bivalvia)	21.01%
<i>Phyllodoce arenae</i> (Annelida: Polychaeta)	20.63%
<i>Listriella barnardi</i> (Arthropoda: Crustacea: Amphipoda)	20.43%
<i>Amygdalum papyrium</i> (Mollusca: Bivalvia)	20.22%

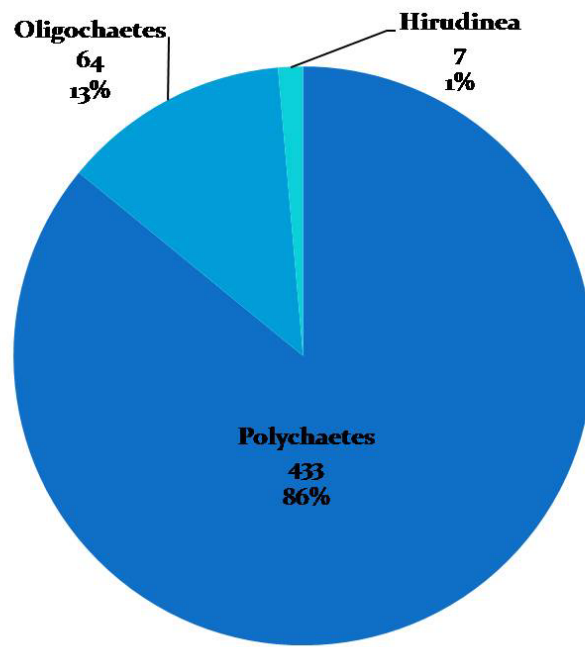


Figure 1. Proportion of annelid species by major taxonomic groups.

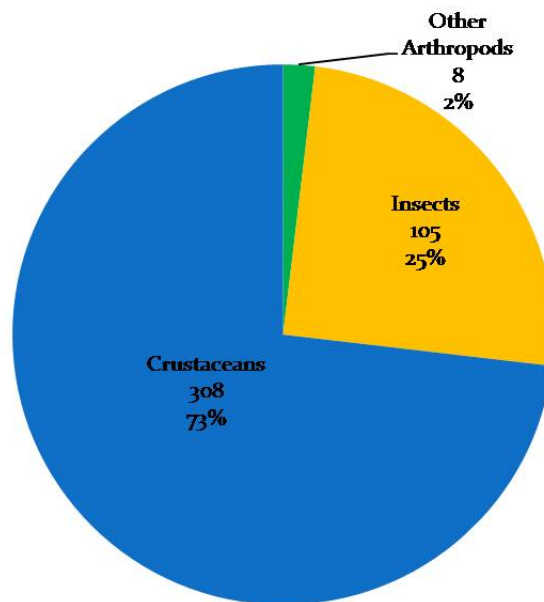


Figure 2. Proportion of arthropod species by major taxonomic groups.

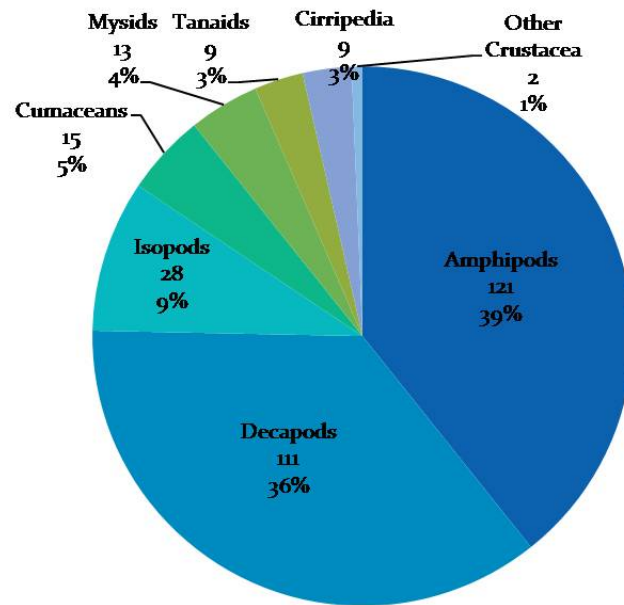


Figure 3. Proportion of crustacean species by major taxonomic groups.

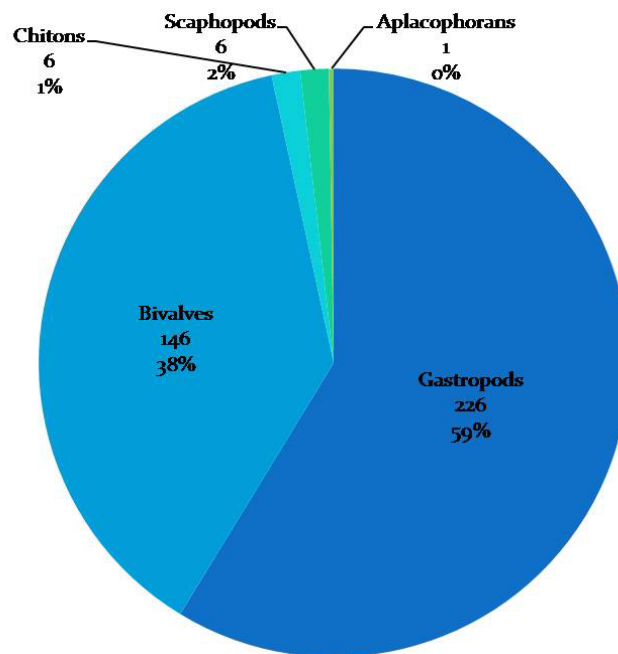


Figure 4. Proportion of molluscan species by major taxonomic groups.

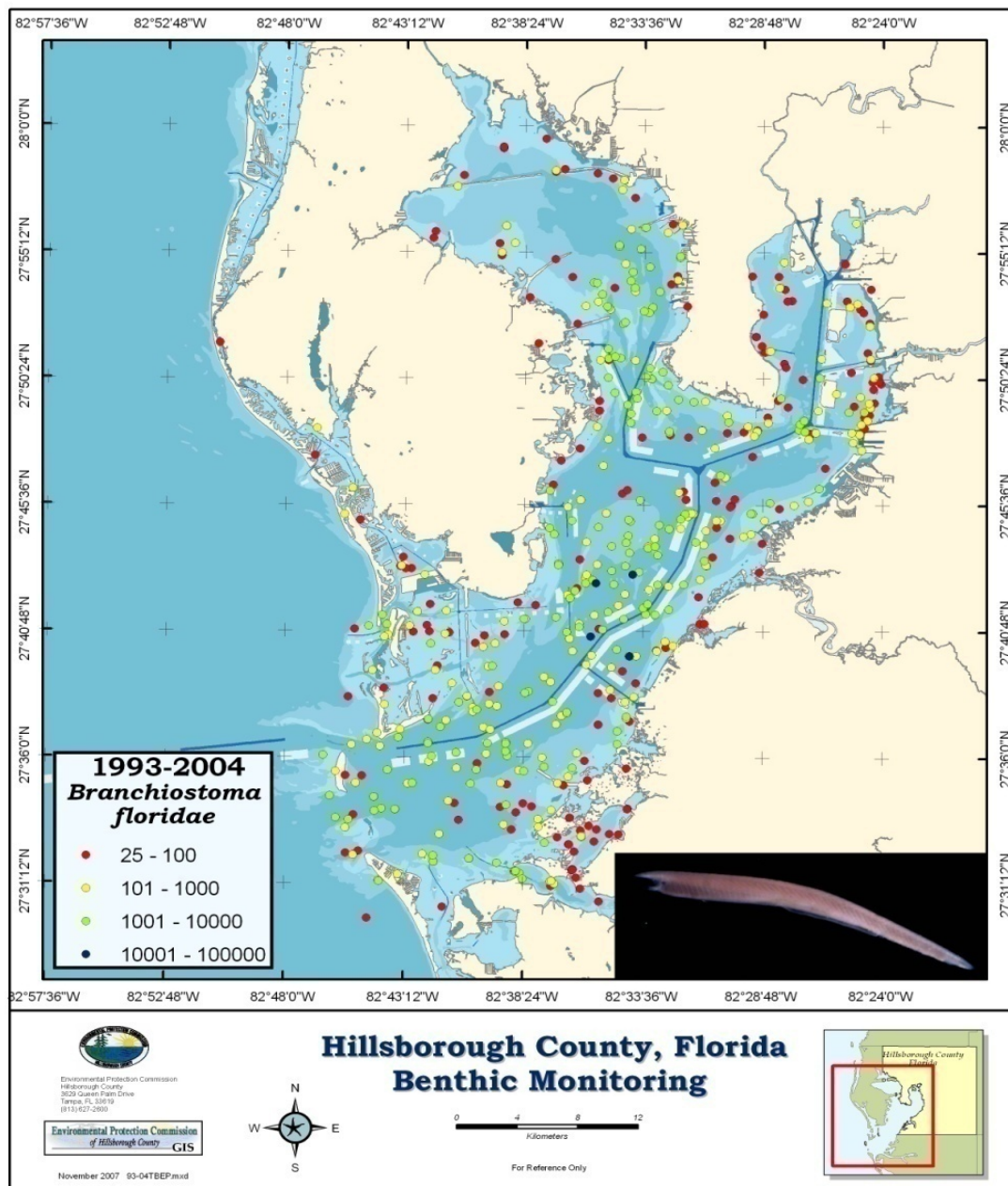


Figure 5 (part 1). Distribution of *Branchiostoma floridae* and *Monticellina dorsobranchialis* in Tampa Bay (from Karlen et al. 2008).

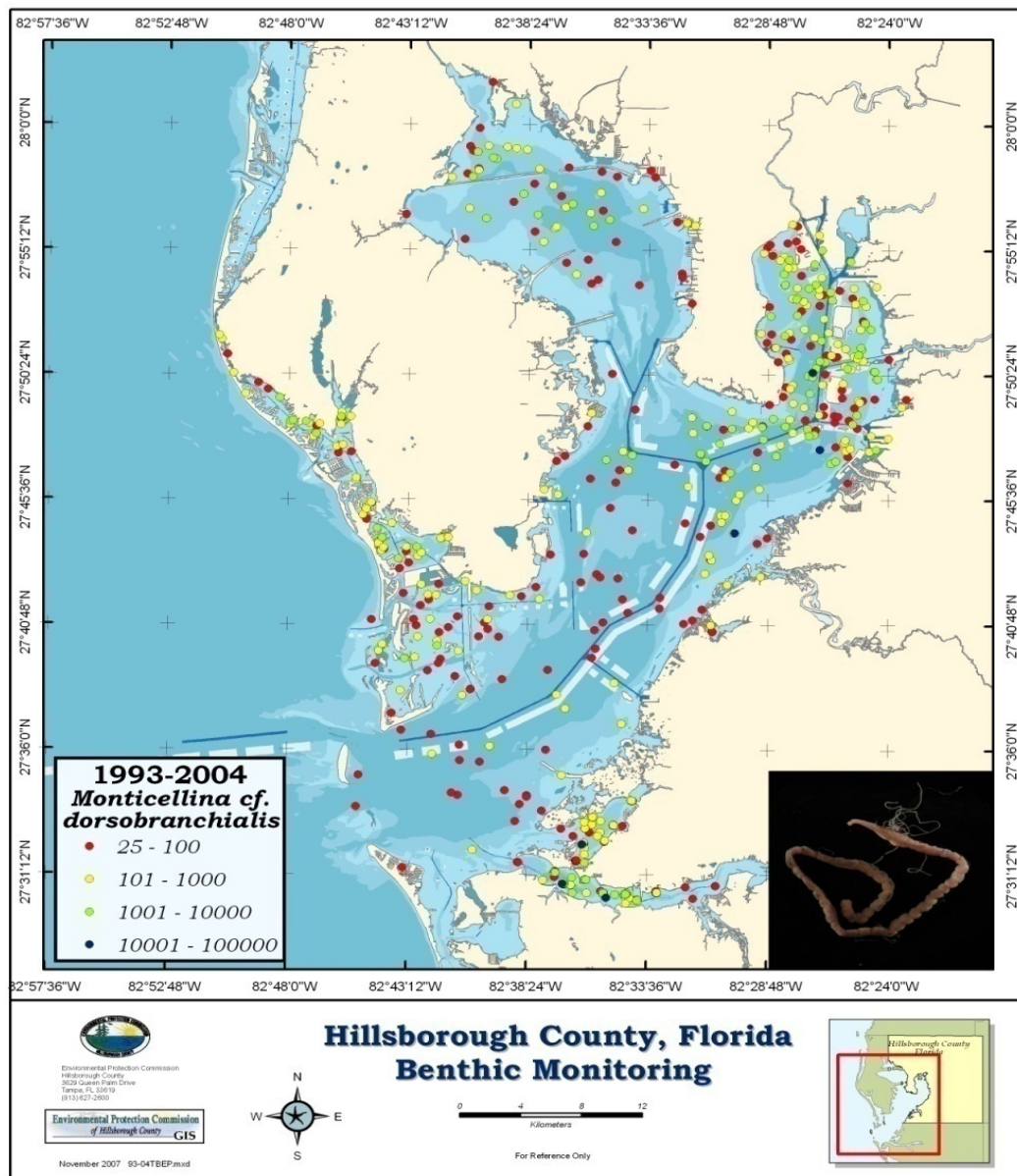


Figure 5 (part 2). Distribution of *Branchiostoma floridae* and *Monticellina dorsobranchialis* in Tampa Bay (from Karlen et al. 2008).

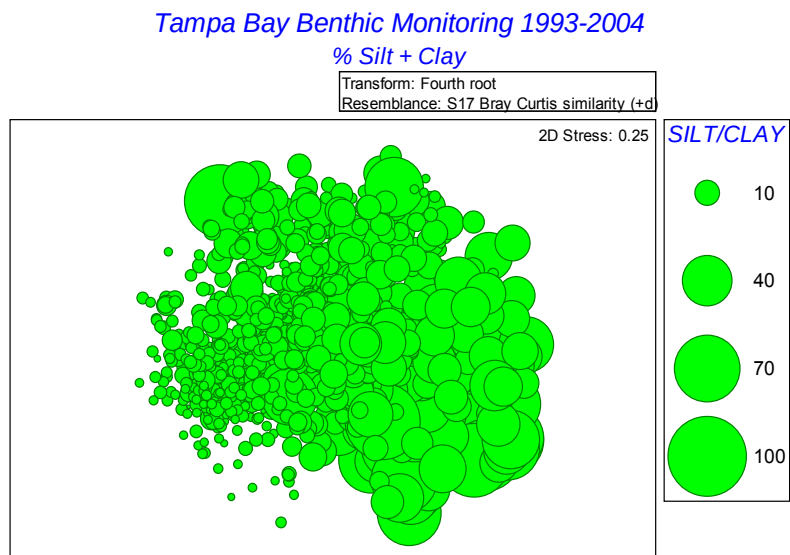
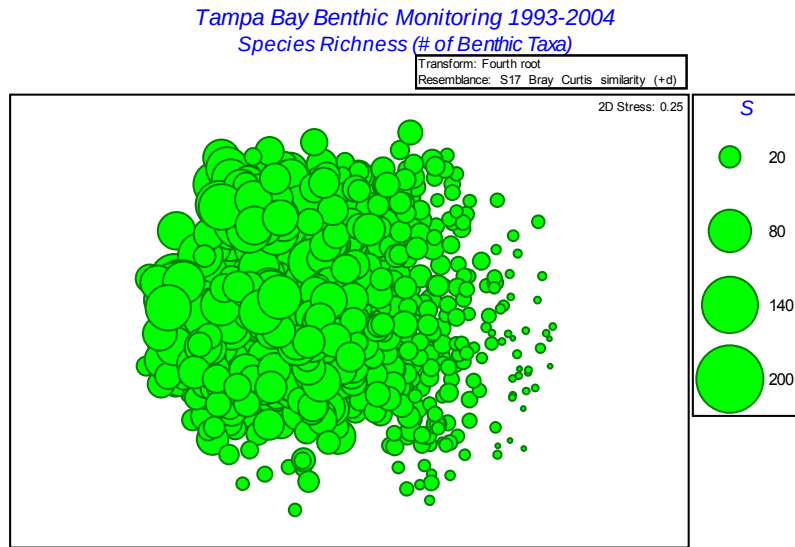


Figure 6. Non-metric Multidimensional Scaling (MDS) plots depicting benthic community similarity (Bray-Curtis similarity index) between Tampa Bay benthic monitoring sites; species richness (left plot) and sediment % silt+clay (right plot) values superimposed as bubble plots (from Karlen et al. 2008).

Hillsborough Bay 1993-2008

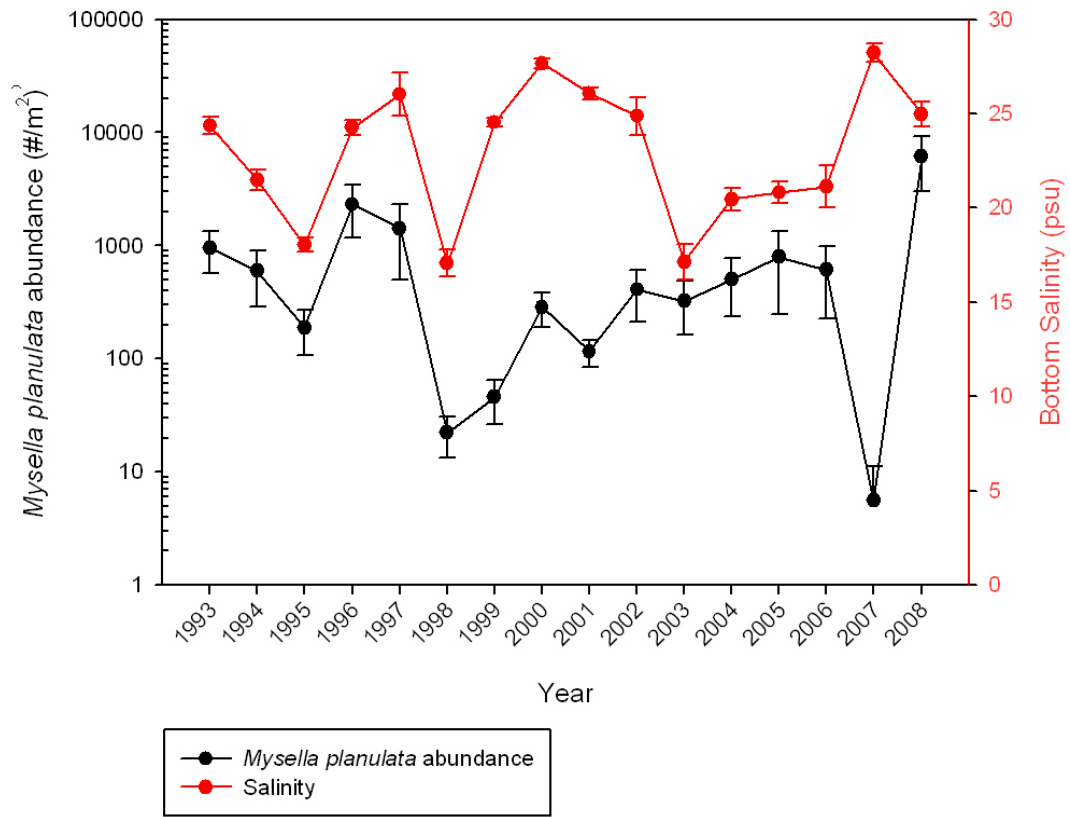


Figure 7. Temporal trends in *Mysella planulata* abundance (black - left axis) and bottom salinity (red – right axis) in Hillsborough Bay from 1993-2008. Mean values $\pm$ 1 standard error. (Note: Abundance plotted on logarithmic scale and salinity on linear scale).

COMMERCIALLY OR RECREATIONALLY IMPORTANT INVERTEBRATES IN TAMPA BAY

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ABSTRACT

Several programs within the Marine Fisheries Research section of the Florida Fish and Wildlife Research Institute (FWRI) collect data on commercially or recreationally-important invertebrates of Tampa Bay. FWRI monitoring programs are either species-specific or collect general data on invertebrates. The Crustacean Fisheries group established a continuous trap-based program monitoring the stone crab (*Menippe mercenaria*) fishery in 1989. The same year, the Fisheries-Independent Monitoring (FIM) program implemented an intensive sampling regime for fish and selected invertebrates (*Callinectes* spp., *Menippe* spp., *Limulus polyphemus*, *Portunus* spp., *Farfantepenaeus* spp., *Rimapenaeus constrictus*, *Sicyonia* spp., and *Argopecten* spp.). The FIM program conducts stratified random sampling monthly throughout Tampa Bay and in the tidal portion of four rivers. The Molluscan Fisheries group implemented a recruitment monitoring program for the bay scallop (*Argopecten irradians*) in 1998. For many species, such as the oyster *Crassostrea virginica*, a baseline has been established from directed studies, but longer-term data sets are lacking, which prevents the development of meaningful conclusions concerning long-term trends. Another important source of data is from commercial harvests; with records dating to the late 1800s in some cases (although sporadic). Other data sources exist, but there is a clear need for a comprehensive compilation of the findings from these short-term studies. Invasive benthic species are increasingly abundant in Tampa Bay, yet their abundance and distribution are not monitored.

INTRODUCTION

The status of the infauna and small epifauna of Tampa Bay has been described in a number of studies published in previous Bay Area Scientific Information Symposium (BASIS) proceedings (Simon and Mahadevan 1985; Culter et al. 1991; Leverone et al. 1991; Karlen et al. 1996). The status of commercial invertebrate species from the late 1950s through 1978, including the blue crab, oysters, and shrimp, were discussed in the first BASIS proceedings (Lombardo and Lewis 1985). Many peer-reviewed, gray-literature, and technical reports describing the invertebrates of Tampa Bay contain relevant information but are limited in spatial or temporal focus. One of the earliest attempts to describe the diversity of invertebrates in Tampa Bay (Dragovich and Kelly 1964) lists 82 species, seven of which could be considered commercially important. Subsequent reports have described hundreds of total species inhabiting the Bay system (Karlen 2010).

The earliest references to Tampa Bay fisheries date to the late 1880s. Collins (1892) reported that 1.67 million lbs. of oysters, 2,475 lbs. of clams, 2,606 lbs. of crabs, and 152,000 lbs. of shrimp were landed from Florida waters statewide in 1888. In 1896 surveys were conducted that provided county-specific data on commercial fisheries landings in Florida (Brice 1898). Brice's study indicated that the primary targets for the 700 or so fishermen who considered Tampa Bay their home port included mullet, red snapper, sea turtles, alligators, otters, oysters and sponges. The primary oyster beds were located in Hillsborough Bay and at the mouth of the Manatee River (442,400 lbs. landed from July 1895 through June 1896). Sponges were landed primarily in Tarpon Springs, on the Anclote River. Harvests of sea turtles (9,375 lbs.) and alligators (16,750 skins) were already declining, although more otters were being taken (2,750 pelts). Many species, such as red snapper, were probably caught as far away as Key West and Pensacola but taken to port in

Tampa Bay. Federal fisheries reports were published at irregular intervals between the turn of the century and the 1930s, after which they were published more consistently.

At the first Bay Area Scientific Information Symposium (BASIS), total shellfish landings (~4 million pounds) were described as consisting primarily of bait shrimp, food shrimp, and blue crabs (Lombardo and Lewis 1985). Food shrimp landed in the Tampa Bay area were harvested throughout the Gulf of Mexico and were not thought to be representative of Tampa Bay stocks. The level of oyster harvests common in the 19th century (> 1 million lbs. per year) were not achieved in the 20th century, but large harvests were made in the 1940s (~100,000 lbs.) and 1960s (~150,000 lbs.). Oyster harvest activity since 1970 has been negligible. Blue crab landings reached a maximum in 1964–67 (~2.2 million lbs.) and again in 1975 (~900,000 lbs.).

Since the mid-1990s, several monitoring programs within the Marine Fisheries Research section of the Fish and Wildlife Research Institute (FWRI) have collected data on commercially or recreationally important invertebrates in Tampa Bay. Two work groups, Molluscan Fisheries and Crustacean Fisheries, target specific invertebrate species, while two others, Fisheries-Independent Monitoring (FIM) and Fisheries-Dependent Monitoring (FDM), take a broader approach to monitoring the bay's biota. The FIM group samples at stations selected at random throughout the bay and records data on all marine fish species and selected invertebrate species collected. The FDM group adapted sampling methods from federal fisheries programs and monitors the landings of commercial and recreational fishers in Florida and neighboring federal waters. The FDM data on recreational fisheries focus primarily on finfish.

METHODS

Bay Scallop—The monitoring of the bay scallop (*Argopecten irradians*) in Tampa Bay involves two approaches: adult surveys and the monitoring of juvenile recruitment. Staff members of Tampa Bay Watch, a nonprofit association, have conducted annual surveys of adult bay scallops since 1995 (except in 2002 and 2003, when water quality was poor). Its Great Bay Scallop Search (GBSS) is a one-day event that involves 75–100 volunteers per year who swim 50-m transects and count all bay scallops detected within 1 m of the transect line. This event focuses on the seagrass beds of lower Boca Ciega Bay and Pinellas Point. During periods of very low scallop abundance, 1999–2001, for example, FWRI staff partnered with commercial shrimp roller-frame trawlers to survey 30 stations per year in the lower Boca Ciega Bay area, with paired tows of ~ 15-min duration. No attempt was made to determine density from roller-frame trawls, but spatial distribution was noted. Quantitative transect surveys (20 stations) have been conducted annually throughout the bay by FWRI Molluscan Fisheries since 2004.

FWRI monitors juvenile bay scallop (spat) recruitment by using methods similar to those described by Geiger et al. (2006). A spat collector consists of a one half-bushel citrus bag stuffed with a 30 x 48 cm piece of polypropylene mesh. The collectors are deployed by attaching them to a line suspended under a buoy and anchored to a cinder block. From 1996 through 2006, collectors were deployed for a 6-week period but since 2007 have been deployed for a 2-month period. Complementary collectors were deployed either three weeks (1996–2006) or one month after the initial deployment (beginning in 2007) at the same sites, creating overlapping deployment periods. This allowed detection of recruitment peaks that could have been missed if a collector had been removed during a peak when spat were still too small to be detected (< 1.0 mm). When a collector was retrieved, all spat on or within it were counted. Initially, during fall 1996, 15 spat collectors (five collectors at each of three stations) were deployed in lower Tampa Bay and the configuration remained the same for one year. Collectors were deployed similarly in fall/winter 1997–98 and 1998–99. Monthly recruit collector deployments have occurred continuously since fall 2000. Deployment stations near Indian and Tarpon keys (in lower Tampa Bay) focused on areas known

to contain populations of the bay scallop. The number and locations of recruitment stations varied slightly during the study period. Recruitment rate is calculated by dividing the number of spat on a collector by the number of days the collector was in the water.

Bay-scallop restoration activities were initiated in Tampa Bay in 1997 and continued through 2009. In early restoration attempts, wild-harvested or hatchery-spawned scallops were placed in wire cages, which were then attached to the bottom within seagrass beds. In 2006 the method of restoration changed from use of cages to the release of larval and juvenile scallops. Restoration, survey, and recruit-monitoring studies were all ongoing at the beginning of 2010.

Eastern Oyster—In 2005, the distribution of the eastern oyster (*Crassostrea virginica*) was estimated from photointerpretation of digital orthophoto quadrangles (DOQQs; O’Keefe et al. 2006) and verified using ground surveys. In the surveys, 10% of the sites identified as oyster reefs were visited and the perimeter of each reef walked with a handheld GPS unit to classify a number of points on the reef and in adjacent habitats. The accuracy of photointerpretation was estimated for both fringing (reef that is located along the edges of a shoreline habitat) and freestanding (reef that isolated from other shoreline habitats) oyster habitat. The total length of manmade structures in Tampa Bay and Boca Ciega Bay was calculated from the Environmental Sensitivity Index (ESI) shoreline layer in FWRI’s GIS database. Segments of similar shoreline habitat were ranked numerically according to susceptibility to damage by oiling, with 10 being most susceptible and 1 being least susceptible. The vertical portions of seawalls are the most common man-made structure listed in this ESI that are covered by oysters. The bay-wide area of this habitat was estimated to be 1 m² for each linear meter of seawall, assuming that oysters occupy a 1-m wide band from roughly the mean high-tide line to the mean low-tide line. From 2005 through 2007 the abundance of the oyster at three reefs in Lower Tampa Bay was estimated by counting of all live oysters in 10 1-m² quadrats on each reef; this was done six times over the three years (Arnold et al. 2008).

Crustaceans—The Crustacean Fisheries group’s stone-crab (*Menippe* spp.) monitoring program deployed five stone-crab traps biweekly at each of four stations (Rattlesnake Key, Bean Point, bay side of Passage Key, and ~3 miles west of Anna Maria Island) and studied the catch from 1989 through 2009. This monitoring program samples year-round, unlike the commercial fishery, which harvests crabs from October 15 to May 15. Each slatted, plastic commercial stone-crab trap has a central regulation-sized opening (3.5" by 5.5") with a vertically-oriented throat extending into the trap, a single degradable wooden slat on one side (required by law to reduce mortality in abandoned or lost traps), and a surface buoy attached by a polypropylene line. Each time the traps are retrieved, they are baited with 1–1.5 kg of mullet or other bait.

A suite of biological and morphological data is recorded for each captured stone crab. Biological data include gender, reproductive condition, condition of the exoskeleton, molting phase, injuries, claw type (i.e., crusher or pincer), claw size determined by propodus length (PL: distance from the tip of the claw to the elbow), and crab size determined by carapace width (CW: distance between the tips of the lateral spines). To estimate the species identity of each crab, color and patterning of the carapace, claws, and legs of crabs with CW ≥ 30 mm are recorded. Previous studies indicated that the transition from juvenile to adult coloration is complete at this size (Geiger and Bert 2006). Stone crabs in the Gulf of Mexico are a complex of two species and their hybrids but they are managed as a single species. On alternating sampling trips, the traps are scraped clean of fouling organisms and all newly-settled juvenile stone crabs (those of CW < 20 mm) are collected, counted, and their CW is measured. The catch per unit effort for stone crab claws (cl-CPUE) is calculated as the average number of legal-sized claws (≥ 70 mm PL) per trap.

Blue Crab—The Crustacean Fisheries group's trapping studies of the blue crab (*Callinectes sapidus*) were conducted during two 4-year periods: October 1980–September 1983 (1980s) and October 2002–September 2005 (2000s). The data presented were collected from four stations with different salinity regimes during both studies. Five vinyl-coated, wire blue crab traps (61 cm X 61 cm X 33 cm) were deployed 100 m apart along a 500m transect at an average water depth of 2 m. On a weekly basis, each trap was retrieved and all crabs were removed. The trap was re-baited with mullet, and was then redeployed. For each trap, staff recorded the number of blue crabs and, for each, gender and carapace width (distance between the tips of the lateral spines).

Fisheries-Independent Monitoring—The FWRI initiated its FIM program in 1985, and sampling in Tampa Bay began in 1989. The FIM program is funded primarily through a Federal Sport Fish Restoration grant and by revenues from the sale of Florida saltwater fishing licenses. Additional FIM collections in the Hillsborough, Alafia, and Palm rivers and McKay Bay began in 2000 as part of Tampa Bay Water's Hydrobiological Monitoring Program (HBMP). The program uses different types of gear so that a range of life-history data from a variety of habitats can be collected about fishes and selected invertebrates. A 21.3-m seine was the primarily gear used to sample juvenile fish and small fauna in shallow, near-shore habitats. A 183-m seine was used in open-water areas less than 2m depth, and a 6.1-m otter trawl was used in deeper, open-water habitat; both were used to target adult fishes and other larger, more mobile fauna. The sampling protocol followed a fixed-station design from 1989 to 1995 and was changed to a stratified-random design in 1996. Tampa Bay is divided into five zones overlaid with a sampling grid with sectors of 1 nm². The index of abundance is calculated for each species in samples taken with 21.3-m seines and for the otter trawls as the number of individuals per 100 m² of bottom sampled. Catch per unit effort (CPUE) is calculated for the 183-m haul seines as the number of animals/set. Abundance of juvenile blue crabs is estimated mostly from 21.3-m seine catches, while that of adults is estimated from 183-m haul seines. Abundance of pink shrimp (*Farfantepenaeus duorarum*) is estimated from catches made with all three types of gear.

Fisheries Dependent Monitoring—The FDM group at FWRI has monitored landings from roughly 11,000 commercial and 6.6 million recreational fishers in Florida since it was created in 1985. Similar programs run by the National Marine Fisheries Service operated from 1950 to 1985 and sporadically from the late 1880s until 1950. Since 1950, landings have been reported by county, three of which are relevant to Tampa Bay: Pinellas, Hillsborough, and Manatee. Invertebrates fished commercially in Tampa Bay include blue crabs, stone crabs, bait shrimp, and food shrimp. Commercial fishers are required to report all landings to the FDM group. Data was available through 2008.

Historical landings of mollusks included oysters, clams, and scallops. Of those, only oysters contribute minor quantities today. The commercial harvest of bay scallops has been prohibited since 1994 and wild clams have not been a major commercial species in the bay area for at least 30 years, so their landings are not reported here. Landings of shrimp and blue crabs are recorded in pounds of whole animals and in trips fished per year. Landings of stone crabs are reported in pounds of claws and trips fished per year. Landings of oysters are recorded in bushels but are reported in pounds of meat landed by estimating that 6.5625 lbs. of meat are harvested per bushel of whole oysters landed. For each, a harvest rate is estimated as the total number of pounds harvested divided by the number of trips, yielding pounds per trip (CPUE).

RESULTS

Bay Scallop—Bay scallops were present in very modest numbers in the mid-1990s and were nearly impossible to find from 1998 through 2006. Two distinct efforts found similar results. In the GBSSs in 1995–1997, about 70 scallops were encountered each year (data are presented here as the

total number of scallops observed per event), but in the subsequent nine years, very few scallops were detected (Figure 1a). From 1999 through 2001, roller-frame surveys counted only 52 scallops at 91 stations. In 2007, GBSS volunteers counted more than 500 scallops, and baywide surveys conducted by FWRI staff found high densities near Pinellas Point and Fort DeSoto. In 2008 and 2009, both groups likewise detected scallops in abundance. FWRI's baywide surveys detected scallops at all but one station. Anecdotal observations of scallops were reported from seagrass beds along the Courtney Campbell Causeway, in Hillsborough Bay, near Weedon Island, in Rookery Bay, and along the southern shores of Tampa Bay.

The period with modest number of scallops being detected during GBSSs from 1995 through 1997 was followed by a large peak in scallop recruitment in the fall of 1997. During the following decade, few scallops were detected during GBSSs and spat recruitment peaks remained small. A large recruitment peak was observed in the winter of 2006–07 but summer 2007 scallop surveys indicated only modest numbers of scallops. Large recruitment events also occurred in the winters of 2007–08 and 2008–09. Each of these strong recruitment events translated into a high abundance of adult scallops the following summer. The increase in recruitment in 1997 preceded the largest-ever single planting of juvenile scallops in Tampa Bay; 12,800 planted in 1998. Juvenile plantings in 2000, 2001, and 2004 were not followed by increases in adult abundance or recruitment. The combined releases of 13,000 juveniles and 14 million scallop larvae in 2004–2008 are contemporaneous with a period of abundant adults and increased recruitment (Figure 1).

Eastern Oyster—The spatial extent of natural oyster reefs in 2005 was estimated to be 136,686 m² in Tampa Bay and 26,142 m² in Boca Ciega Bay (O'Keefe et al. 2006). From ground surveys, the authors concluded that photointerpretation was 78% correct overall, 85% correct for free-standing reef, and 72% correct for fringing reef. Much of the error in classification came from the inability to detect oysters beneath the ~44.2 million m² of mangrove canopies from aerial photographs. The length of seawall in Tampa Bay was calculated to be 313,120 m and in Boca Ciega Bay, 236,045 m. Using an estimate of 1 m² for each m of seawall, the total area of oysters on manmade structures (549,165 m²) probably surpasses that of natural oyster reef. Oysters are estimated to cover 711,993 m² (176 acres). Oyster abundance on natural reefs ranged from 26.0 ($\pm$ 25.2) m⁻² to 110.4 ($\pm$ 78.7) m⁻² (Arnold et al. 2008), and studies are underway to estimate densities on a variety of manmade structures.

Stone Crab—The sampling of stone crabs by the Crustacean Fisheries group indicates that catch rates fluctuate by as much as 20% annually, with peaks occurring in 1993 and 2006 (Figure 2A). A period of overall smaller and more variable catches occurred from 1996 through 2003. Data from FDM show that commercial landings vary greatly among counties. The CPUE has declined for catches landed in Hillsborough County (Figure 2B). The CPUE for catches landed in Manatee County has been highly variable with no apparent trends. Landings in Pinellas County show a strong and steady positive trend in CPUE. Landings for Hillsborough County have essentially ceased, and landings for Manatee County have plateaued (Figure 2C). Pinellas County is the only bay-area county that demonstrated an upward trend in landings of stone crabs over the period. Stone crabs are only occasionally captured in FIM gear and are not reported.

Blue Crab—The abundance of blue crabs can be estimated from data collected in three FWRI programs: Crustacean Fisheries, FIM, and FDM. In monitoring conducted by the Crustacean Fisheries group, crab catch was much less for 2002–05 than for 1980–83. Catch rates of crabs of CW > 100 mm decreased the most, and adult females were seldom caught in the habitat studied (Figure 3). The number of juveniles collected in the 21.3-m seine by FIM fluctuates annually and appear to undergo a cycle of four to seven years. The mean density since inception was 0.4 crabs per 100 m² (Figure 4A) but the 1989–2000 landings were greater than those in 2001–2008. The

number of adults caught in FIM 183-m seine sets fluctuates on cycle of six to seven years, with minima in 2002 and 2008. Commercial landings reported to FDM have fluctuated widely, ranging since 1986 from < 200,000 lbs. to > 1.1 million lbs. for the three-county region (Figure 4C). CPUE increased through 2006, but since then total landings have plummeted and CPUE has diminished in Hillsborough and Pinellas counties.

Shrimp—Catches of pink shrimp by FIM in Tampa Bay have varied greatly by year and gear type. All three types of gear collected very few shrimp in 2005 but returned to relatively normal, albeit variable, levels in 2006–09, >2 shrimp per set (Figure 5A). Landings of commercial bait shrimp (pounds/trip) were similar in all counties in most years—just less than 100 lbs. per trip, with two notable exceptions. In 1988–89, CPUE for shrimp landed in Hillsborough County was notably more than was typical (possibly an artifact of the small number of trips), and in 2007, CPUE for shrimp landed in Pinellas County was notably greater than typical. Total landings in the three counties varied greatly (Figure 5B). Hillsborough County has had virtually no bait shrimp landings since 1986. Manatee County had greater landings from 1987 through 1993 (57,302 - 101,173 lbs.), after which landings decreased and remained between roughly 12,700 and 25,830 lbs. Landings appear to be declining in Pinellas County, although they are highly variable. Total landings of bait shrimp decreased from almost 300,000 lbs. in 1999 to < 50,000 in 2008. The number of trips landing bait shrimp in Tampa Bay counties decreased from around 3000 per year in the mid-1990s to <1000 per year in 2008. Total landings of food shrimp appear to be decreasing, but they are more variable than those of bait shrimp. Landings in Pinellas County decreased from almost 3.4 million lbs. in 1998 to < 200,000 lbs. in 2008. In Hillsborough County, landings have ranged from 1 to 3 million lbs. per year, decreasing from 2.4 million lbs. in 2004 to 1.1 million lbs. in 2007, then increasing slightly in 2008. Few food shrimp are landed in Manatee County (Figure 5C). The annual number of trips landing food shrimp in Tampa Bay counties decreased from around 1000 in the mid-1990s to <200/year in 2008. As the number of trips has decreased, the number of shrimp landed per trip has increased. Landings per trip in Hillsborough and Pinellas counties increased from around 200 lbs. in the early 1990s to around 8000 lbs in 2006; Hillsborough landings continued to increase through 2008, while those for Pinellas decreased. The pattern in Manatee County was similar to that in Pinellas, although smaller in scale (Figure 5D).

DISCUSSION

Broad Patterns—Data collected by established research programs have already provided some insight to long-term trends in abundance and harvest of commercially and recreationally important invertebrates of Tampa Bay. Each species studied seems to exhibit unique patterns in variability. Scallops exhibit large shifts, with distinct periods of great abundance separated by long periods of lesser abundance. This pattern has been observed in oysters in the Chesapeake Bay and has been said to describe a multiple steady-state system (Powell et al. 2008). Both crab species studied exhibit a fairly large annual fluctuation in abundance, as well as an abundance cycle that repeats every 7–15 years. Finally, long-term trends are species-specific, but the data often suggest that socioeconomic changes drive changes in harvest, rather than changes in abundance. The main evidence for this comes from the observation that in some cases CPUE is steady or climbing but effort, the total number of trips per year, is falling.

A major change since the first BASIS has been the recognition that invasive and introduced species are becoming an increasingly-large component of the fauna in the bay area. Some of those species, such as the Asian green mussel (*Perna viridis*), can represent as much as 75% of the biomass in certain locations (Dix et al. 2005). Nonnative species can harm native species in many ways, including predation, disease, and competition for resources. Estimates of the number of introduced species to the bay area range from 65 (Baker et al. 2004) to 85 (<http://nas.er.usgs.gov/queries/>; accessed February 2010), and these are probably underestimates.

Bay Scallop—The population of the bay scallop is its healthiest since commercial harvest was prohibited in 1996. The increase in abundance is most likely the result of a combination of factors: conservative management by the Florida Fish and Wildlife Conservation Commission, improved water quality, increased seagrass habitat, and multi-organizational restoration efforts. Other factors, such as resurgent populations of the scallop in adjacent waters and patterns in climatic conditions (such as El Nino-Southern Oscillations and North Atlantic Oscillations) may be contributors to the return of bay scallops to Tampa Bay.

Oyster—A common means of estimating the value of a filter-feeding organism like an oyster is to estimate the percentage of the volume of the body of water it inhabits that the filter-feeding species can filter in a day. This rate can also be expressed in terms of the number of days it would take the species population to filter the entire body of water it inhabits. The volume of Tampa Bay has been estimated at about 4×10^{12} L (Sheng et al. 1997; Comp and Seaman 1985). If each square meter of the 711,993 m² of oyster habitat in Tampa Bay contained 50 oysters, the population of oysters in Tampa Bay would be approximately 36 million. If each oyster filtered 5 L/hr (Newell and Langdon 1996), the total volume of water filtered could reach 4×10^9 L/day, so oysters might filter the bay once every 1000 days. Many factors could affect this estimate, including variation in size and abundance of oysters and biological changes related to temperature. Additionally, localized areas with long and convoluted shorelines and relatively smaller areas of open water would receive greater benefits from oysters than the bay as a whole, because of the relatively higher surface to volume ratio. There is no reliable estimate for the area of mangrove forests that shelter oyster beds. If even 10% of the 44.2 million m² of mangrove canopy in Tampa Bay overlaid oyster reefs, the estimated abundance and subsequent filtration capacity would increase six-fold because we did not include any estimate of the oyster population in mangrove habitat. Similarly, if oyster densities were greater or smaller in the upper reaches of the bay, the estimated capacity of the bay's oyster population to filter the bay water would change. To date, quantitative photointerpretation of oyster abundance based on historic aerial photographs has not been possible in Tampa Bay. Many other species contribute to filtering phytoplankton, and the volume of the bay filtered each day would undoubtedly be affected by estimates of their abundance and filtration rate.

Stone Crab—Many valuable observations concerning stone crabs were made during the 20-year study. Tampa Bay has some of the largest claw CPUEs in the state, with the largest claw CPUEs occurring in the fall. Males were generally more common in traps in later summer through fall, and females were more common in winter through early summer. The sizes at which most stone crabs attain a legal-sized claw (2 3/4" claw) and thus become available to the fishery is at an approximate CW of 76 mm for males and 88 mm for females (Gerhart and Bert 2008). There is no tolerance for undersized claws. According to recent analyses of size data collected during the long-term stone-crab monitoring program in Tampa Bay, about half of male stone crabs entering the fishery have not reached sexual maturity and thus have not contributed reproductively to the population. Female stone crabs mature at a smaller size than males and it is likely females landed have reproduced at least once prior to entering the fishery. Simonson and Hochberg (1992) found that 70% of the stone crab claws in the marketplace had "good" breaks, i.e., had been taken in such a way as to be unlikely to kill the crabs they were harvested from. They also found that 7.6% of the claws had been regenerated at some point in their life, indicating the crab survived the break and regenerated a legal-size claw. The CPUE for stone crabs has one of the strongest short-term fluctuations among Tampa Bay invertebrates but nonetheless appears to be increasing overall in Pinellas and Manatee counties.

Blue Crab—The methods for estimating the status of the blue crab can lead to different conclusions. Both FIM and FDM data on abundance suggest no long-term trend, although juvenile abundance may have decreased since the 1990s. The FDM data suggest that the variability in

landings is greater than the variability in CPUE. Observations by the Crustacean Fisheries group suggest that not only has abundance decreased since the 1990s but also that there has been a dramatic shift in the size distribution of crabs caught in commercial traps. In particular, very few adult females are now present in Tampa Bay, relative to estimates from previous studies. All the data sets indicate that recently blue crab abundance has been low.

Shrimp—The methods for estimating the status of shrimp in Tampa Bay can lead to different conclusions. Although FIM data on abundance of pink shrimp show greater annual variability than FDM monitoring data of commercial landings do, these do not suggest any long-term trends in abundance. The FDM data suggest that the variability in total landings is greater than the variability in CPUE. For both bait and food shrimp, total landings appear to be declining. For bait shrimp, the catch per trip is relatively stable, but for food shrimp, landings per trip were clearly increasing. Socioeconomic factors are probably contributing to the decrease in shrimp-fishing effort, including reduced availability of commercial waterfront property (which reduces the availability of access points for launching vessels, harvesting, and processing commercial seafood), increased fuel costs, stable or decreasing dockside prices, and changes in harvest regulations. The ban on commercial shrimp trawling in Tampa Bay in the 1990s also probably had an effect.

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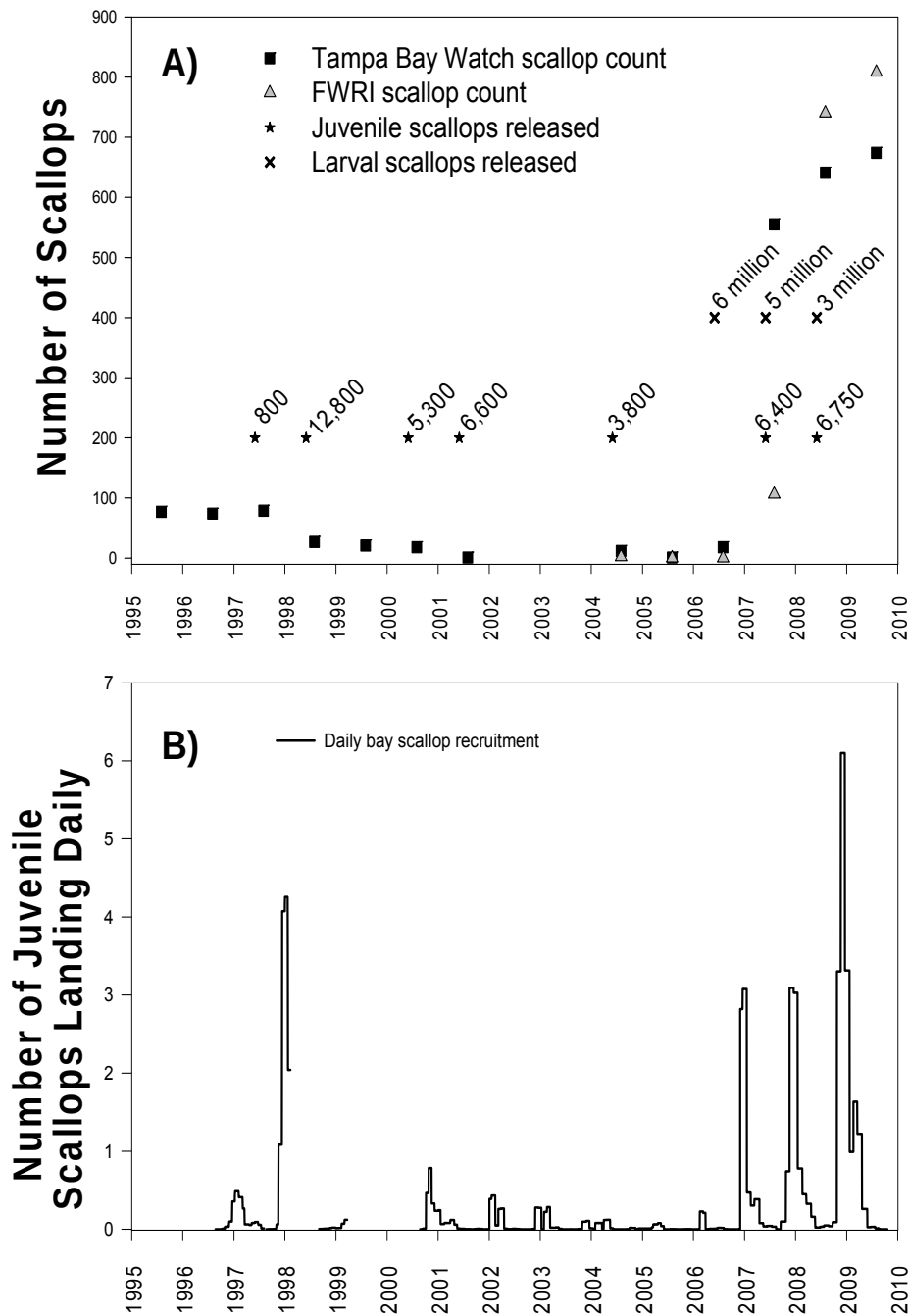


Figure 1. A) Annual counts of adult bay scallops, number of juveniles and larval scallops released and daily scallop recruitment in Tampa Bay, 1995–2009. Counts of adults and releases of juveniles (★) or larvae (×) are shown as the total number observed during each count. B) Recruitment to collectors is expressed as the number of spat on a collector divided by the number of days the collector was deployed.

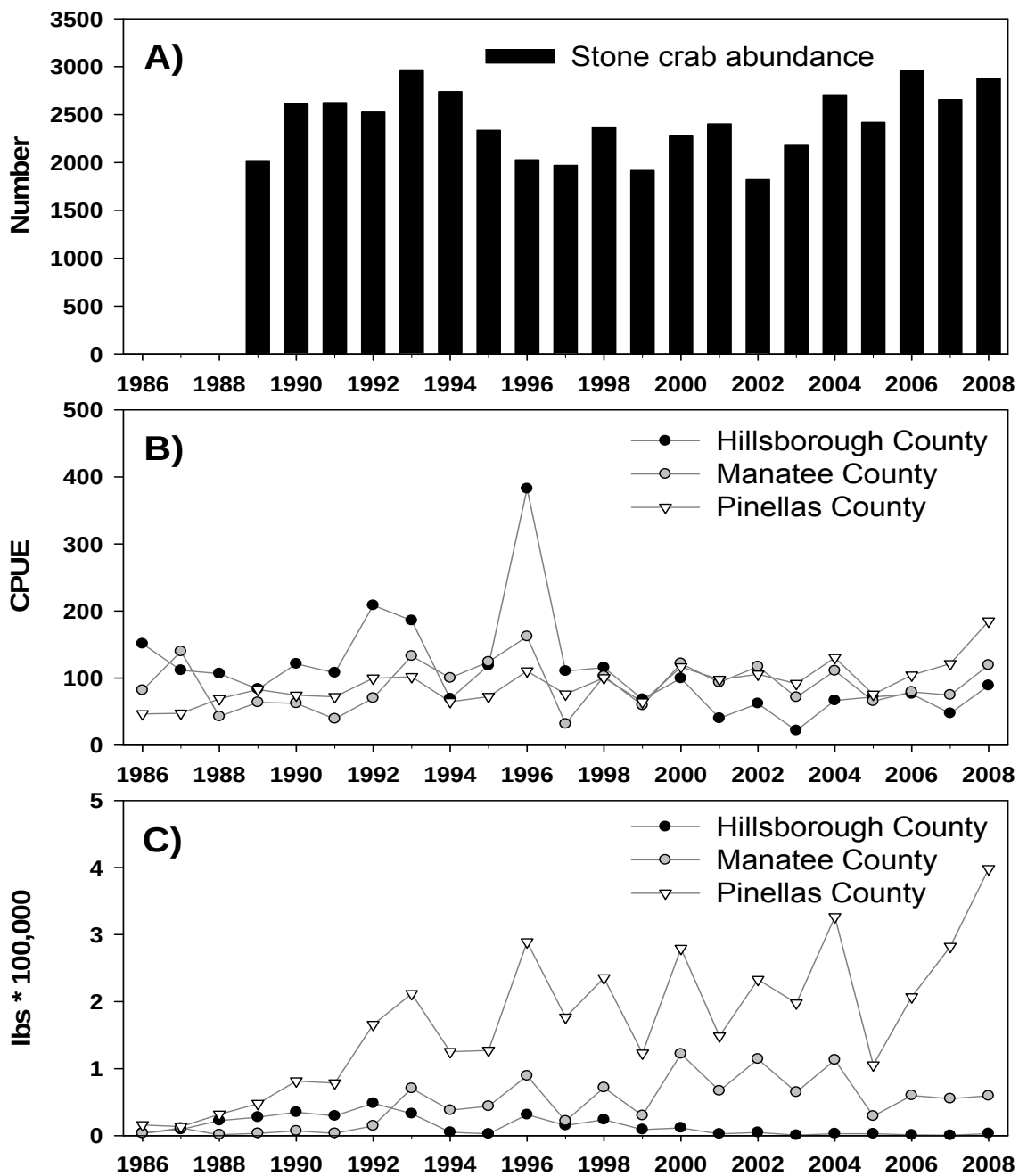


Figure 2. Catch of stone crabs, 1986–2008: a) total annual catch (20 traps; live crabs) in Tampa Bay by FWRI's Crustacean Fisheries group; b) average annual harvest rate (pounds of claws per trip, CPUE) reported to FDM for bay area counties; c) total annual landings (pounds of claws) reported to FDM.

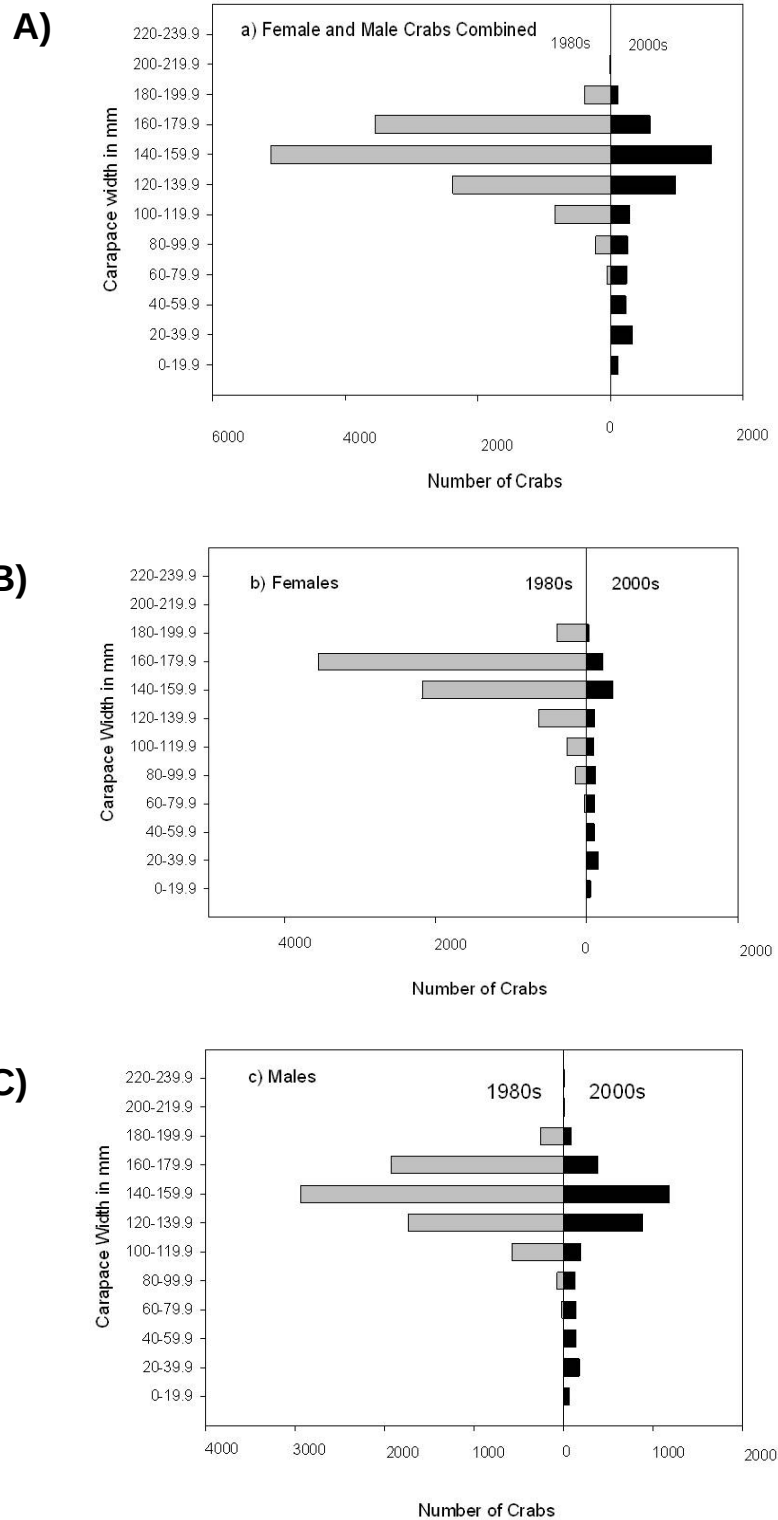


Figure 3. Numbers of blue crabs from FWRI's 1980–83 study and an identical study in 2002–05 study: a) all crabs; b) females; c) males. In each graph each bar represents the total number of crabs collected during the three-year study after being categorized into 20mm CW increments.

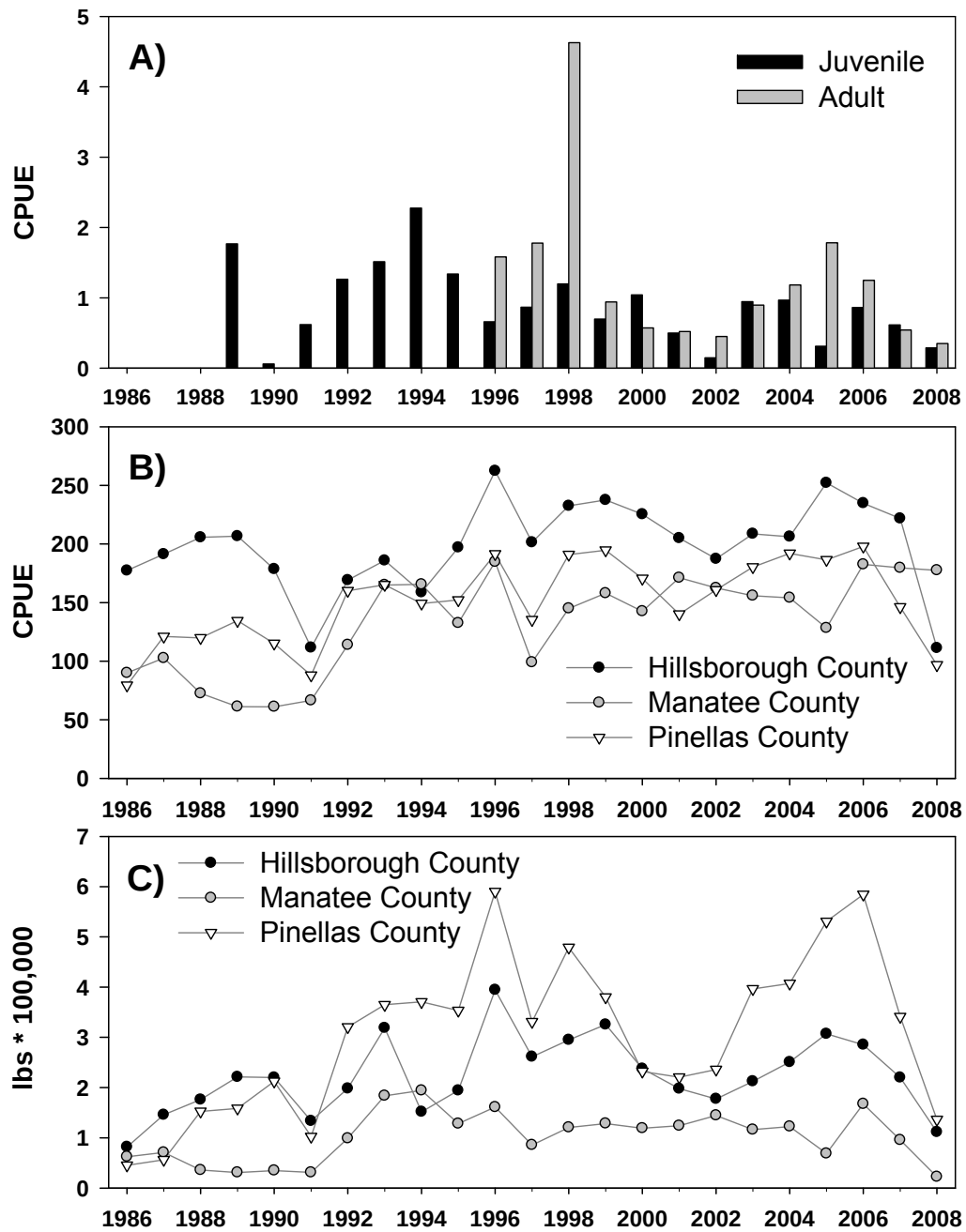


Figure 4. Catch of blue crabs in Tampa Bay, 1986–2008: a) baywide juvenile density (crabs per m^2 for 21.3-m seines) and adult CPUE (number per set for 183-m seines) in FIM samples; b) CPUE (lbs. of whole crabs per trip) reported to FDM, by county; c) total annual landings (lbs. of crabs) reported to FDM.

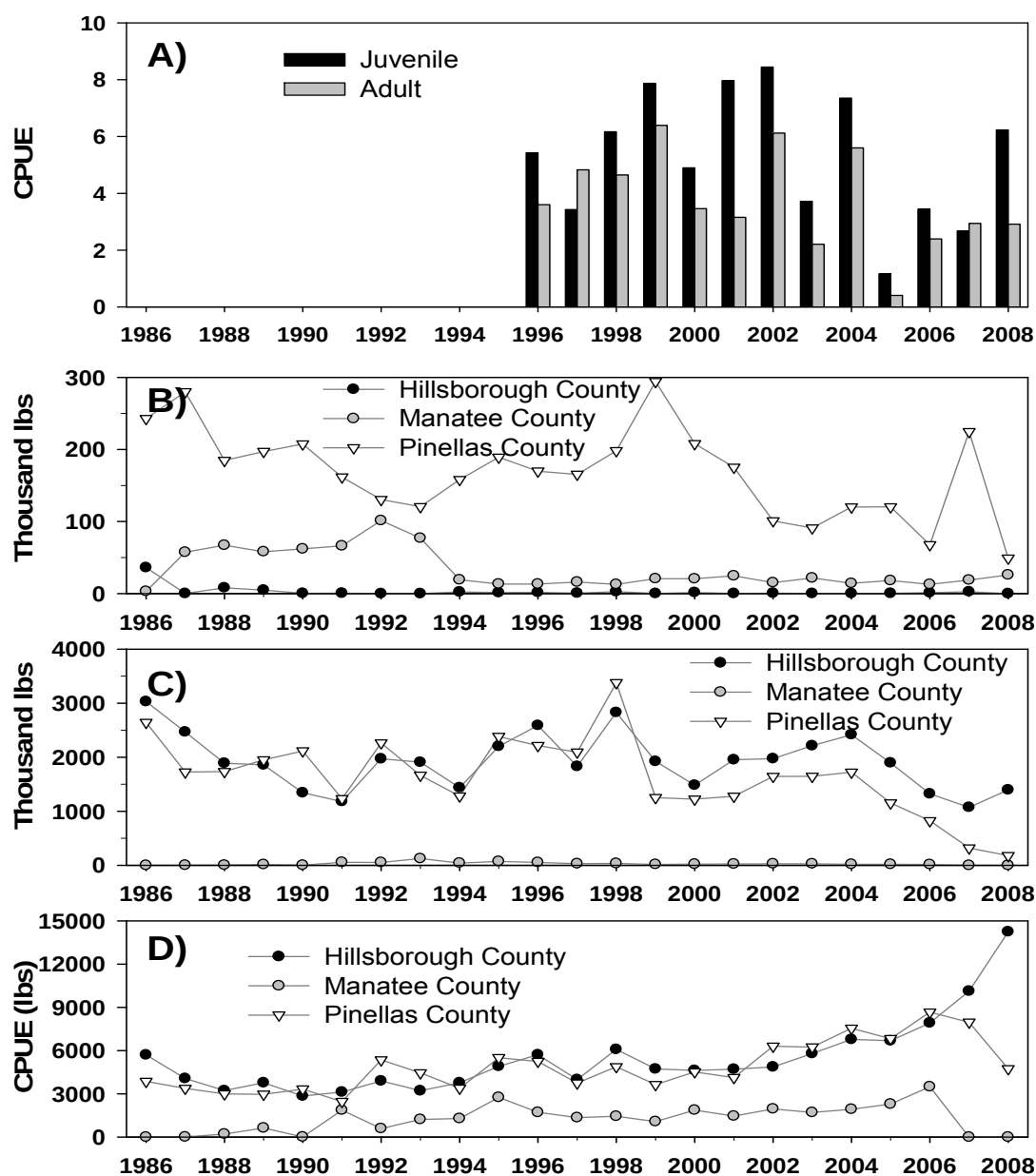


Figure 5. Catch of food and bait shrimp in Tampa Bay, 1986–2008: a) pink shrimp density (# per m² in 21.3-m seines) and adults (number per set in 183-m seines) in FIM samples; b) total annual landings of bait shrimp (pounds) reported to FDM; c) total annual landings of food shrimp (pounds) reported to FDM; d) CPUE (lbs. per trip) for food shrimp landings reported to FDM.

TWENTY-FIVE YEARS AFTER BASIS: AN UPDATE ON THE CURRENT STATUS AND RECENT TRENDS IN COLONIAL WATERBIRD POPULATIONS OF TAMPA BAY

A.B. Hodgson & A.F. Paul

ABSTRACT

Representatives of four orders dominate the avifauna of Tampa Bay: pelecaniformes (pelicans, cormorants, anhingas); ciconiiformes (herons, ibis, spoonbills, storks); anseriformes (waterfowl); and charadriiformes (shorebirds, gulls, and terns). The first bay-wide assessment of colonial waterbird populations was presented at BASIS by Paul and Woolfenden (1985). Twelve of the 22 colonies they reported have been abandoned since due to various causes of habitat loss or disturbance, and c. 59,000 pairs (mostly Laughing Gulls) nested on five colonies that no longer support very large populations. After 1985, 50 new colonies became active, including 15 inland colonies, of which 16 were abandoned later. Using annual breeding bird surveys, we provide trends in the populations of 30 bird species breeding in Tampa Bay, 13 of which deserve enhanced conservation protection through their listing by federal or state agencies. The Tampa Bay breeding population totals 30,000-58,000 nesting pairs, averaging 39,000 annually. The 2009 nesting population (all species) was 58,500 at 44 colonies. Up to 50% of the total colonial waterbird nesting occurs in Hillsborough Bay; the remainder is distributed at colony sites around Tampa Bay. The Cockroach Bay-Terra Ceia Bay, Hillsborough Bay, Johns Pass, and Lower Tampa Bay Important Bird Areas are listed by Audubon of Florida among its 100 Important Bird Areas in Florida. Lower Tampa Bay and Hillsborough Bay were designated as "Important Bird Area of Global Significance" by Birdlife International in 2003 and 2009, respectively. Human disturbance has become the most significant cause of nesting failure annually, accompanied by anthropogenically-induced predator population increases and urban development affecting the number and ecological integrity of estuarine and palustrine wetland foraging sites. We provide a suite of habitat and population management recommendations that should be implemented to conserve the bay's avifauna.

INTRODUCTION

The species richness of colonial waterbirds that nest in the Tampa Bay estuarine system is unique, as many birds of temperate North America breed here, as well as some typically "tropical" birds (Reddish Egrets, Roseate Spoonbills) that do not nest further north, and some species that nest only in low numbers anywhere in Florida (Caspian, Royal, Sandwich, and Gull-billed terns) (Howell 1932; Paul and Woolfenden 1985; Paul and Schnapf 1997; Paul and Paul 2005; Hodgson et al. 2006).

Within Tampa Bay, colonial waterbirds [pelecaniformes (pelicans, cormorants, anhingas); ciconiiformes (herons, ibis, spoonbills, storks); and charadriiformes (shorebirds, gulls, and terns)] nest preferably on small islands that are off-shore, separated by open water and deep channels with tidal currents that discourage predatory mammals from swimming to them, and have no resident mammalian predators. Large numbers of birds of many species may breed at a single site. Generally, sites occupied by larids are sparsely vegetated sand or shell beaches or dredged spoil material, while pelecaniform and ciconiiform birds nest where shrubs or trees are available. (Schreiber and Schreiber 1978). Thirteen species are currently listed by the state and federal wildlife management agencies to receive elevated regulatory protection. Several other species that nest in the watershed, although not formally listed, are very rare (Willet, Wilson's Plover, Gull-billed, Caspian, Royal, and Sandwich terns) and warrant comparable protection.

The importance of Tampa Bay's bird community has been widely recognized by national and international authorities. The Cockroach Bay-Terra Ceia Bay, Hillsborough Bay, Johns Pass, and Lower Tampa Bay Important Bird Areas (IBAs) are listed by Audubon of Florida among its 100 Important Bird Areas in Florida, and BirdLife International and the National Audubon Society recognized Lower Tampa Bay and Hillsborough Bay as globally-significant IBAs in 2003 and 2009, respectively.

In this paper, we briefly summarize the current status and population trends of 30 species of birds nesting in the Tampa Bay system, mostly colonial but also some territorial nesters that often select sites within a mixed species colony, review current management programs to protect them, and provide conservation recommendations to maintain stable populations in the future.

METHODS

We [Florida Coastal Islands Sanctuaries (FCIS)] surveyed colonial waterbird colonies and territorial shorebirds in Tampa Bay, using direct nest counts or flight line counts, and counting nesting pairs and productivity (chicks/nest) when possible (Buckley and Buckley 1976; King 1978; Portnoy 1980; Erwin and Ogden 1980; Erwin 1981; Paul et al. 2004). Laughing Gulls were censused using a circular plot technique and extrapolating nesting density among areas of similar nesting density (Patton and Hanners 1984). We added colony locations to the survey schedule as they were discovered. We also included 15 bird colonies that occur on the bay's periphery at inland locations within the Tampa Bay Estuary Program's watershed boundaries in Hillsborough, Pasco, and Polk counties, but not colonies outside the watershed in Clearwater Harbor and St. Josephs Sound, although they contribute to the regional population (Agency on Bay Management 1995). Numbers of colonies surveyed varied inter-annually contingent on colony activity, personnel, weather, and other constraints. English and scientific names follow the Check-list of North American Birds 7th edition (American Ornithologists' Union 1998) and 50th Supplement (Chesser et al. 2009).

RESULTS

In Tampa Bay, 58,424 nesting pairs of colonial birds (all species), 42.7% of which were Laughing Gulls, bred at 44 colonies in 2009 (Table 1). The 10 year (2000-2009) mean number of nesting pairs (all species) was 44,141 (SD 10,946.57), and the mean number of active colonies was 32 (SD 6.88) (Table 2).

Of the 71 colonies mapped in the Tampa Bay watershed, 22 were discussed in BASIS, of which 12 (54.5%) were abandoned ("winked out") later for various reasons (altered habitats [e.g., urban development, plant succession], predators, human disturbance) since 1985, including 5 colonies that supported most of the gull population (Figures 1, 2, 3). In the past 25 years we located and surveyed 50 new sites undescribed in 1985; however, 16 colonies (32.0%) subsequently collapsed and were abandoned. Cumulatively, the inland colonies supported 10.0% of the regional population. Of the initial 22 colonies, all but six were islands (Paul and Woolfenden 1985). Five were small colonies of Yellow-crowned Night-Herons or Great Blue Herons nesting high in tall oak trees or slash pines near the bay, and the last site was the shore of the Howard Frankland Causeway, where the Florida Department of Transportation planted the roadside in the early 1990s to discourage Black Skimmers from nesting and causing traffic hazards. All recently-active colonies were islands, except the Mobbly powerlines, scattered oystercatcher territories in Apollo Beach, and the Cockroach Bay borrow pit.

In 1985, the Alafia Bank Bird Sanctuary, Washburn Sanctuary, and Tarpon Key National Wildlife Refuge were the three largest mixed colonies of pelecaniforms, herons and ibis in the region. In 2009, pelicans nested at only four sites, Washburn Sanctuary had very few pairs since 2004, and

Tarpon Key was abandoned in 2005, so that the three largest colonies with similar species composition were Egmont Key National Wildlife Refuge and State Park (33,700 pairs, of which 300 were pelicans and >25,000 were larids), the Richard T. Paul Alafia Bank Bird Sanctuary (10,500 pairs, only 150 pairs of pelicans), and Alligator Lake (745 pairs), which had no pelicans.

DISCUSSION

Species richness (30 species) of the regional colonial waterbird population did not change in Tampa Bay from 1985 to 2009, with every endemic species and introduced Cattle Egrets represented. This community remains the largest and most significant colonial waterbird population in Florida outside of the Everglades. The Laughing Gull population has diminished by around 50% since the 1980s and is now concentrated in Hillsborough Bay and Egmont Key. These populations have persisted despite significant and continuing alteration of shoreline habitats, bay bottom, and freshwater wetlands, although recent population declines in Brown Pelicans, Laughing Gulls, Least Terns, and Snowy Plovers suggest that, as elsewhere in Florida, progressive urbanization threatens to further reduce the ecological integrity of the Tampa Bay ecosystem. Roseate Spoonbills and Reddish Egrets, extirpated as nesting species from Tampa Bay until the mid-1970s, have increased significantly, while widely expanding their distribution among suitable habitats in the bay, and Wood Stork, and Royal and Sandwich tern populations have increased slightly. The other pelecaniformes, ciconiiformes, charadriiformes and larids have remained relatively stable. The inland colonies are particularly important for small herons and Wood Storks.

Five additional species are found uniquely in coastal habitats: Clapper Rails, Mangrove Cuckoos, Gray Kingbirds, Black-whiskered Vireos, and Prairie Warblers. Clapper Rails occur in low and high marsh and require expansive areas of continuous cover, areas which are diminishing as the shoreline has been developed. Black-whiskered Vireos have virtually disappeared from Tampa Bay since c. 1991. Mangrove Cuckoos were found annually in mangroves in Boca Ciega Bay, Weedon Island, and Terra Ceia Bay in some years, but are infrequent now. Prairie Warblers are more widely distributed along Tampa Bay mangrove shorelines. Although Gray Kingbirds may also nest in uplands beyond the mangroves, all five species are primarily coastal birds whose populations have decreased in recent years. The four estuarine passerines are susceptible to nest parasitism by increasing populations of Brown-headed Cowbirds.

Paul and Woolfenden (1985) identified a number of biotic and abiotic stressors that influence bird abundance in Tampa Bay. In the decades leading up to the 1980s, coastal habitat loss dominated. In the 1990s, with the large increase in registered watercraft, the most significant issues to have emerged are anthropogenic disturbances from the increasing numbers of recreational boaters and beachgoers that: "...present a vast potential for annual disturbance of breeding birds", as predicted by Paul and Schnapf (1997), continued dredge and fill activities that have had both beneficial and negative effects for colonial waterbirds and beach-nesting species, continued loss of palustrine wetlands (particularly short hydroperiod and ephemeral "prairie ponds"), the trend toward reducing the spatial distribution of palustrine wetlands by condensing them into stormwater ponds and mitigation banks from the natural patterns that birds cue to throughout the landscape, and extremely high populations of meso-carnivores (raccoons, to a lesser extent opossums and, potentially, coyotes and invasive exotic herptiles).

Management Initiatives—Through site-specific management initiatives by FCIS at Audubon-owned and leased sanctuaries, Audubon's Project ColonyWatch, which engages volunteers to observe and protect colonies in cooperation with site managers, and a continuous effort to expand colony management partnerships among agencies and landowners, most of the now-active colonies have been posted, are managed during the year to control predators and remove entangling fishing line during the Tampa Bay Watch and Audubon Monofilament Cleanup, are regularly surveyed to

establish colony species composition and productivity, and are intermittently patrolled. However, with the dramatic increase in public recreation on the water, this program is insufficient to fully protect most colonies. In the past five years we have also implemented a series of inter-agency workshops for law enforcement marine units about the habitat requirements, and laws protecting colonial.

Management Recommendations — Environmental education - In collaboration with land managers and management partners, continue to produce and distribute to the public boaters guides describing the bay's natural areas, and present informational talks about the bay's avifauna.

Colony management - Continue current management activities, and establish and enforce spatial buffers around colonies to prevent site disturbance. Increase enforcement of wildlife protection laws.

Habitat management - Manage existing sites to provide required habitats; the spoil islands in the Hillsborough Bay Important Bird Area support some of the largest colonies of pelicans, herons, ibis, gulls, and oystercatchers in the state. Many nesting colony sites have been abandoned and fewer new sites will be available in the future given the development density. Currently functioning sites must be carefully protected.

Habitat restoration - Continue to acquire land and restore coastal ecosystems to replace the large areas of coastal mangroves, salterns, intertidal mudflats, and freshwater wetlands that have been lost; restore tidal creeks and re-establish altered coastal drainage patterns.

Wetland protection - The loss of both coastal estuarine and inland palustrine wetlands by drainage or alteration has been a dominant cause of population declines of colonial birds regionally and statewide. Locally, habitat fragmentation, seasonal wetland draw downs, and consolidation of freshwater wetlands decreases wetland functioning in the landscape, and reduces forage availability, which particularly affects successful nesting of White Ibis, small herons, and Wood Storks.

Sea level rise - Participate in the dialogue about climate change and potential effects of sea level rise; include in future conservation planning initiatives acquisition of lands and sites that will not be affected by increasing water levels.

Maintaining the vibrant, diverse colonial waterbird population in Tampa Bay in the future will be more challenging than during the past three decades since BASIS, and much more difficult than in the decades preceding widespread coastal development. Despite 25 years of intensive public outreach and environmental education activities by Audubon and others, sedulous volunteers in Audubon's Project ColonyWatch and in the Florida Shorebird Alliance providing colony guardianship, and expanded coordination between non-governmental, local, county, state, and federal wildlife protection programs, human disturbance is an incessant threat to the persistence of local bird colonies. More protective regulations, more enforcement, and heightened public cooperation will all be needed to protect the spectacular, charismatic bird populations of Tampa Bay.

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Table 1. Colony characteristics and management status of colonial waterbird colonies in Tampa Bay, Florida, USA, in 2009.

Colony Number	NAME	Bay Segment	Taxa	Species	Pairs	Winked out after 1984	New since 1984	Ownership / Management	Protected Status	Regional Population (%)	Active within last 5 yrs?	LATITUDE	LONGITUDE
25	Dogleg Key	BCB	P, Ci	12	296		X	FDEP-AP / FCIS	Y	0.51	Y	27.8021	-82.7618
26	Johns Pass, Little Bird Key	BCB	Ci	1	2			Suncoast Seabird Sanctuary	Y	0.00	Y	27.7932	-82.7777
27	Johns Pass, Middle Bird Island	BCB	Ci	2	5			FDEP-AP	Y	0.01	Y	27.7913	-82.7739
28	Johns Pass, Eleanor Island	BCB	Ci			X		City of Treasure Island	Y	0.00	Y	27.7878	-82.7738
29	South Pasadena Marker 34	BCB	L			X	X	City of Pasadena		0.00	N	27.7431	-82.7299
30	Sunset Beach	BCB	L			X	X	City of Treasure Island	N	0.00	N	27.7391	-82.7565
31	Don CeSar Colony	BCB	P, Ci	6	50		X	Private	N	0.09	Y	27.7059	-82.7352
32	Bayway Spoil	BCB	L			X		Developed	N	0.00	N	27.7094	-82.6995
33	Indian Key NWR	BCB	Ci			X	X	USFWS NWR	Y	0.00	Y	27.7011	-82.6909
34	Little Bird Key NWR	BCB	Ci	5	16		X	USFWS NWR	Y	0.03	Y	27.6852	-82.7169
35	Cow and Calf Islands	BCB	P, Ci	2	9		X	FDEP-AP		0.02	Y	27.6856	-82.6916
36	Darling Key	BCB	P, Ci	3	17		X	FDEP-AP		0.03	Y	27.6765	-82.6813
37	Jackass Key NWR	BCB	P, Ci	4	30		X	USFWS NWR	Y	0.05	Y	27.6693	-82.7177
38	Tarpon Key NWR	BCB	P, Ci			X		USFWS NWR	Y	0.00	N	27.6666	-82.6932
39	Whale Island NWR	BCB	P, Ci			X	X	USFWS NWR	Y	0.00	N	27.6626	-82.6930
40	Shell Key County Preserve	BCB	Ch					Pinellas Co. / Florida	Y	0.00	Y	27.6645	-82.7445
41	Mule Key NWR	BCB	P, Ci			X	X	USFWS NWR	Y	0.00	Y	27.6619	-82.7178
42	Listen Key NWR	BCB	P, Ci			X	X	USFWS NWR	Y	0.00	N	27.6596	-82.7179
43	Sister Key	BCB	P, Ci			X	X	Pinellas County		0.00	N	27.6503	-82.7312
44	Ft. DeSoto Park	LTB	L, Ch			X	X	Pinellas County	Y	0.00	N	27.6488	-82.7433

Colony Number	NAME	Bay Segment	Taxa	Species	Pairs	Winked out after 1984	New since 1984	Ownership / Management	Protected Status	Regional Population (%)	Active within last 5 yrs?	LATITUDE	LONGITUDE
45	Egmont Key NWR/State Park	LTB	P, Ci, Ch	10	36, 521		X	USFWS NWR / Florida State Parks	Y	62.51	Y	27.5894	-82.7614
46	Little Bayou Bird Island	MTB	P, Ci	10	140		X	FDEP-AP / FCIS	Y	0.24	Y	27.7196	-82.6312
47	Coffeepot Bayou Bird Island	MTB	P, Ci	14	612		X	Private	Y	1.05	Y	27.7916	-82.6241
48	Gandy Radio Tower	OTB				X	X	Unknown	N	0.00	N	27.8772	-82.5902
49	Howard Frankland	OTB	L			X		FDOT	N	0.00	N	27.9046	-82.6335
50	Cooper's Point	OTB				X		Pinellas Co. / City of Clearwater	N	0.00	N	27.9730	-82.6891
51	Alligator Lake	OTB	P, Ci	12	745			City of Safety Harbor / Pinellas Co.	Y	1.27	Y	27.9813	-82.6990
52	Philippe Park	OTB	Ci			X		Pinellas Co.	N	0.00	N	28.0053	-82.6778
53	Mobbly Bay Powerlines	OTB	P	1	19		X	Progress Energy	N	0.03	Y	28.0038	-82.6677
54	Courtney Campbell Causeway	OTB	L			X	X	FDOT	N	0.00	N	27.9736	-82.5958
55	Wilson Property/Grand Hyatt	OTB	Ci			X		Private	N	0.00	N	27.9654	-82.5514
56	Sunset Park	OTB				X		City of Tampa	N	0.00	N	27.9374	-82.5201
57	Westshore	OTB				X		City of Tampa	N	0.00	N	27.9002	-82.5361
58	McKay Bay	HB				X	X	City of Tampa / TPA	Y	0.00	N	27.9371	-82.4143
59	Hooker's Point	HB				X	X	TPA	Y	0.00	N	27.9076	-82.4338
60	Tampa Port Authority Spoil Island 2D	HB	Ch	9	2,152			TPA / FCIS	Y	3.68	Y	27.8805	-82.4313
61	Fantasy Island	HB	Ch	1	1			TPA / FCIS	Y	0.00	Y	27.8683	-82.4253
62	Spoil Area C	HB	L, Ch			X	X	Mosaic	Y	0.00	N	27.8571	-82.4003
63	Richard T. Paul Alafia Bank Bird Sanctuary	HB	P, Ci, Ch	16	6,234			Mosaic / FCIS	Y	10.67	Y	27.8483	-82.4106
64	Tampa Port Authority Spoil Island 3D	HB	Ch	2	23			TPA / FCIS	Y	0.04	Y	27.8331	-82.4352

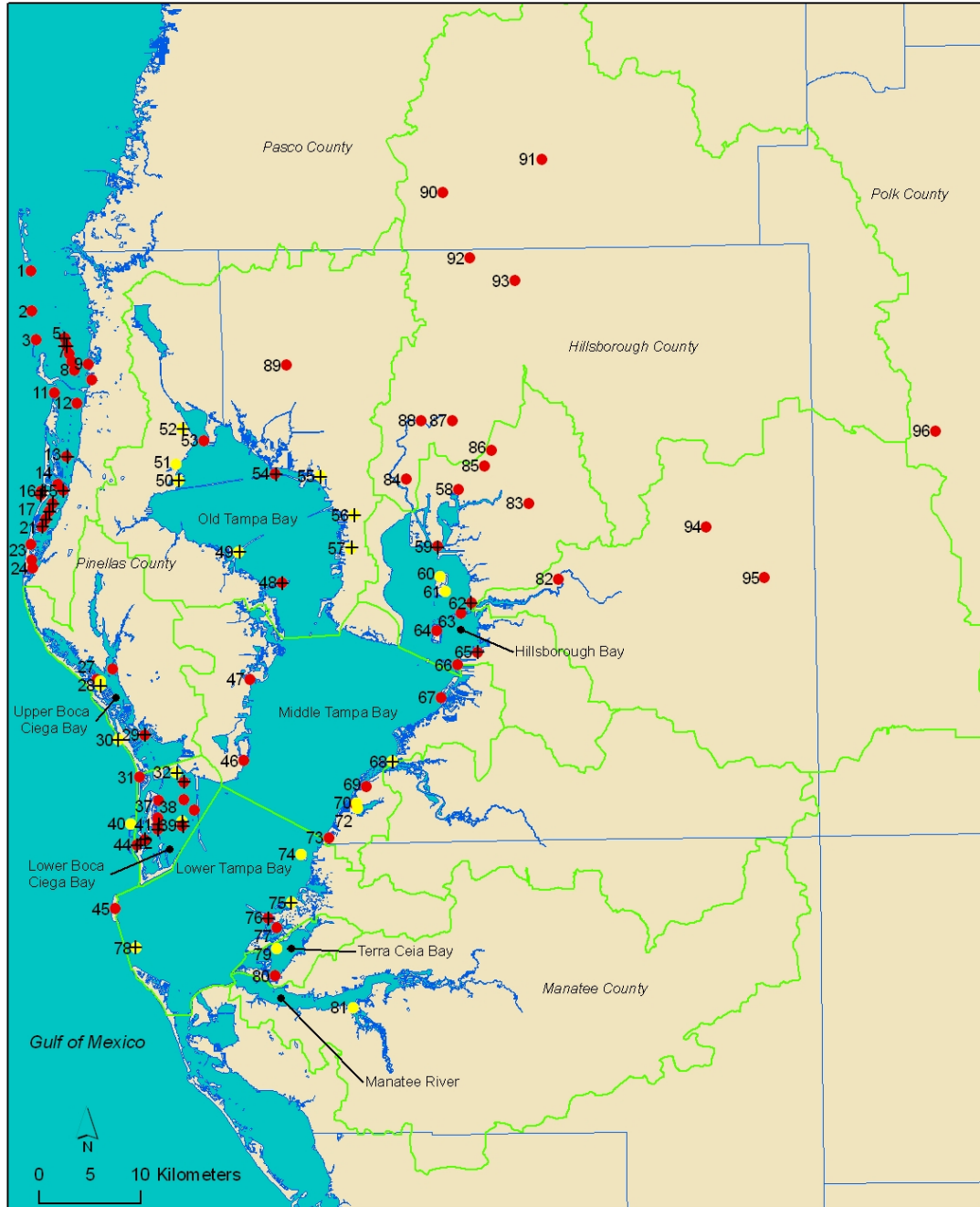
Colony Number	NAME	Bay Segment	Taxa	Species	Pairs	Winked out after 1984	New since 1984	Ownership / Management	Protected Status	Regional Population (%)	Active within last 5 yrs?	LATITUDE	LONGITUDE
65	Port Redwing	HB	L, Ch			X	X	TPA	Y	0.00	N	27.8132	-82.3951
66	Fishhook Spoil Island	HB	Ch	2	13			TPA / TECO	Y	0.02	Y	27.8024	-82.4152
67	Apollo Beach Oystercatchers	HB	Ch	2	15		X	Private	N	0.03	Y	27.7733	-82.4318
68	Mouth of Little Manatee River	MR	P, Ci			X		FDEP Cockroach Bay Aquatic Preserve	N	0.00	N	27.7160	-82.4823
69	Cockroach Bay Preserve	MTB	Ch	1	30		X	ELAPP	Y	0.05	Y	27.6955	-82.5079
70	Hole in the Wall, Cockroach Bay Preserve 1	MTB	Ci				X	ELAPP	Y	0.02	Y	27.6811	-82.5183
71	Hole in the Wall, Cockroach Bay Preserve 2	MTB	Ci	1	20		X	ELAPP	Y	0.02	Y	27.6799	-82.5198
72	Hole in the Wall, Cockroach Bay Preserve 3	MTB	Ci				X	ELAPP	Y	0.02	Y	27.6764	-82.5169
73	Piney Point	MTB	P, Ci	14	2,795		X	SWFWMD	Y	4.78	Y	27.6505	-82.5462
74	Manbirtee Key	MTB	Ci, Ch	4	24			MCPA / FCIS	Y	0.04	Y	27.6359	-82.5740
75	Two Brothers Island	LTB	Ci			X		Private	N	0.00	N	27.5935	-82.5847
76	Skyway Bridge Least Tern colony	LTB	L			X	X	FDOT	N	0.00	N	27.5808	-82.6090
77	Miguel Bay Colony	LTB	P, Ci				X	FDEP-AP / FCIS	Y	0.00	Y	27.5708	-82.5995
78	Passage Key	LTB	P, Ci, L, Ch			X		USFWS NWR	Y	0.00	Y	27.5545	-82.7404
79	Nina Washburn Sanctuary	TCB	P, Ci	7	52			FCIS	Y	0.09	Y	27.5527	-82.5999
80	Washburn Junior/Terra Ceia Bay Little Bird Key	TCB	P, Ci	14	407		X	FDEP Terra Ceia Aquatic Preserve / FCIS	Y	0.70	Y	27.5285	-82.6015
81	Dot Dash Dit Colony	MR	P, Ci	13	2,360			Private / Florida / FCIS	Y	4.04	Y	27.4993	-82.5243

Colony Number	NAME	Bay Segment	Taxa	Species	Pairs	Winked out after 1984	New since 1984	Ownership / Management	Protected Status	Regional Population (%)	Active within last 5 yrs?	LATITUDE	LONGITUDE
82	Heath Yellow-crowned Night-Heron Colony	HC	Ci	1	5		X	Private	N	0.01	Y	27.8772	-82.3129
83	Office/Ferman Bird Colony	HC	P, Ci	8	74		X	Private	Y	0.13	Y	27.9448	-82.3417
84	Robles Park	HC	Ci	4	31		X	City of Tampa	Y	0.05	Y	27.9740	-82.4550
85	Corporex Colony	HC	P, Ci	7	94		X	Private	N	0.16	Y	27.9786	-82.3857
86	East Lake Island	HC	P, Ci	5	14		X	Florida Audubon Society	Y	0.02	Y	27.9922	-82.3784
87	Temple Crest/Orange Lake/Wargo Bird Colony	HC	P, Ci	8	51		X	City of Tampa / TPA	N	0.09	Y	28.0193	-82.4174
88	River Cove Yellow-crowned Night-Heron colony	HC	Ci				X	City of Tampa	N	0.02	Y	28.0192	-82.4486
89	Citrus Park Bird Colony	HC	P, Ci	9	486		X	Private	N	0.83	Y	28.0699	-82.5834
90	Heron Point	PaC	P, Ci	7	57		X	Private	N	0.10	Y	28.2157	-82.4349
91	Saddlebrook	PaC	P, Ci	3	48		X	Private	Y	0.08	Y	28.2277	-82.3297
92	Cypress Creek Preserve	HC	P, Ci	11	3,294		X	ELAPP	Y	5.64	Y	28.1629	-82.3975
93	Cross Creek Colony	HC	P, Ci	2	8		X	Private	N	0.01	Y	28.1424	82.3520
94	Medard County Park	HC	P, Ci	10	477		X	Hillsborough Co.	Y	0.82	Y	27.9218	-82.1630
95	Alafia River Corridor Preserve	HC	P, Ci	5	46		X	ELAPP	Y	0.08	Y	27.8756	-82.1053
96	Wood Lake/Somerset Lake	PoC	P, Ci	14	1,151		X	City of Lakeland	Y	1.97	Y	28.0036	-81.9311
	Totals				58,424	27	48			100.00%			

Taxa: P-pelecaniformes, Ci-ciconiiformes, Ch-charadriiformes, L-larids.

Values are number of species, nesting pairs, and % of 2009 regional nesting population.

Abbreviations: ELAPP – Hillsborough County Environmental Lands Acquisition & Protection Program, FDEP-AP - Florida Department of Environmental Protection Aquatic Preserves, FDOT – Florida Department of Transportation, MCPA – Manatee County Port Authority, TPA – Tampa Port Authority, USFWS NWR - U. S. Fish & Wildlife Service National Wildlife Refuge.



 **Audubon** OF FLORIDA

Legend

- Basis I Colonies
- Post Basis I Colonies
- + Abandoned Colonies
- Tampa Bay Watershed Basins

Figure 1. Bird colonies in the Tampa Bay, Florida, USA, ecosystem from 1984-2009 (colonies 1-24 are excluded because they are not in the Tampa Bay watershed).



Figure 2. Bird colonies in Boca Ciega Bay from 1984-2009.



 Audubon OF FLORIDA

Legend

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Figure 3. Bird colonies in Terra Ceia Bay from 1984-2009.

Table 2. Nesting pairs (no./species) of colonial waterbirds and shorebirds and assessment of recent population trends in Tampa Bay, Florida, USA, from 2000 – 2009.

SPECIES	MEAN	STD. DEV.	POPULATION ASSESSMENT
Brown Pelican	1,024	326.15	45 is the major nesting site since 2004 when 79 and 38 collapsed; widespread also at several smaller colonies, declining
Double-crested Cormorant	455	68.48	Widely distributed at 7 sites; shifted from 79 and 38 when they collapsed; stable
Anhinga	334	93.11	Widely distributed at 7 sites; stable
Least Bittern	2	1.69	Uncommon – nesting at 4 or more freshwater sites with large cattail stands; under-surveyed
Great Blue Heron	217	61.80	Widely distributed at 10 heronries, and various misc. sites; stable
Great Egret	740	148.15	Nesting at 18 sites, >100 pairs at 63, 81, 25, 47, and I-25 (Clearwater Harbor) in that order; stable
Snowy Egret	923	193.63	c. 75% decline since 1970s (Ogden 1978); stable last 10 yrs; 73 increased to 300 pairs
Little Blue Heron	315	88.92	Nesting at 73, 63, and 94, and other sites; declined since 1950s with freshwater wetland loss; stable last 10 yrs
Tricolored Heron	788	178.87	Widespread at all mixed heronries; c. 60% of the population at 3 colonies: 73, 63 and 51; stable
Reddish Egret	57	21.19	Nesting at 6 sites: 63 largest group; 51 – only known freshwater site; c. 16% of state population in Tampa Bay
Cattle Egret	4,146	2,836.85	Abundant at 63, 73, 51, 92, and 81; increasing since 1980s
Green Heron	29	12.01	Nesting at 11 sites, notably 73, and other solitary locations; stable
Black-crowned Night-Heron	112	52.27	Nesting at the major heronries, notably 73, and inland sites; stable
Yellow-crowned Night-Heron	73	39.58	Nesting in mixed heronries; other small groups in tall coastal trees in residential areas; declining since 1980s; recent decline more rapid
White Ibis	9,180	3,464.63	Most common endemic wading bird; dependent on El Niño cycles and prey concentrated as freshwater wetlands draw down; most nesting at 63 and 73
Glossy Ibis	285	102.58	Nesting only at 63, 73, and 92; formerly approx. 50% were at 79; require shallow freshwater wetlands; stable to declining
Roseate Spoonbill	329	111.26	Exponential increase at 63 since 1975; radiated to 11 sites in the past 5 yrs; population not stabilized
Wood Stork	212	116.93	Nesting only at 81, plus inland colonies , 92, 93, 86, 95, and 89
Snowy Plover	0.4	1.26	Rarely nesting at 44, 40, 45; and usually unsuccessful due to disturbance

SPECIES	MEAN	STD. DEV.	POPULATION ASSESSMENT
Wilson's Plover	25	20.68	Spottily distributed in salterns and suitable bare habitat; 74 recently important; stable; prob. under-surveyed
American Oystercatcher	91	13.58	c. 72 prs in Hillsborough Bay on spoil island shorelines (60, 63, 64, 66); the rest at widespread sites; stable, approx. 21% of state population nests in Tampa Bay
Black-necked Stilt	32	31.35	Nesting sporadically at 60, 64, 69 around drying algae mats; rare
Willet	34	14.43	Rare and inconspicuously distributed in salt marshes and dunes; under-surveyed
Laughing Gull	19,698	8,741.13	Nesting only at 60, 64 and 45; approx. 50% decline since early 1980s; Tampa Bay hosts c. 20% of entire southeast U. S. population
Gull-billed Tern	8	5.69	A few pairs annually, often with Black Skimmers, nearly annually at 60, 64
Caspian Tern	83	10.57	Most nesting at 60, 64; formerly 63; Hillsborough Bay colony is the state's largest
Royal Tern	3,618	1,857.76	Nesting formerly at 63 and 78; now at 45 and Hillsborough Bay 60 or 64; increasing since 1990s
Sandwich Tern	811	341.14	All at 45 in 2009; formerly Hillsborough Bay (60, 64, or 63); poss. increasing
Least Tern	116	91.38	Most natural habitat lost; recently c. 80% are rooftop nesters; declining; most nesting on beaches unsuccessful due to human disturbance
Black Skimmer	406	192.24	In the last five years, nested at 60, 64, 45, 78, 40, and 29; stable, but in some years, zero nesting success

Values are mean and standard deviation of nesting pairs; See Table 1 for colony identification numbers.

A BRIEF HISTORY OF THE FISH FAUNA OF TAMPA BAY

R.E. Matheson, Jr., R.H. McMichael, Jr., T.C. MacDonald & G.E. McLaughlin

ABSTRACT

Although there have been large changes in abundance for some species, the composition of the ichthyofaunal community in Tampa Bay has remained relatively stable since it was last comprehensively reviewed in the early 1960s. Since the 1960s, Tampa Bay has been subjected to many stressors, including multiple red tides, industrial spills, changes in nutrient loading, changes in seagrass coverage, changes in freshwater inflow, changes in current patterns due to activities such as bridge and causeway construction, and losses of shallow- water habitat due to activities such as dredging and filling. Nevertheless, the primary components of the Tampa Bay fish community remain intact. A review of palaeontological data indicates that 46 species currently found in the Tampa Bay region have been in the area since the Pleistocene. A review of historical literature, museum databases, and unpublished field collection data indicates that of the more than 300 species currently known from Tampa Bay, approximately 40% had been recorded from the system by 1900, another 40% had been documented by the early 1960s, and approximately 20% of the fauna has been documented since the early 1960s. Most of the species in the latter group are represented by one or a few specimens and are species long known from nearby freshwater or marine habitats. A few of the new species are established exotics, and a few more are apparently recent invaders that have established populations in the bay. Eliminating rare species and questionable records reduces the core of the ichthyofauna of Tampa Bay to 195 species, only 12% of which were first recorded there after the 1960s.

INTRODUCTION

The aquatic ecosystem of the Tampa Bay region has been affected by numerous changes on both geological and historical time scales. Over geological time, changes in climate and sea level have radically altered the shape and size of the Florida peninsula (Randazzo and Jones 1997). Historical changes have been both anthropogenic and non-anthropogenic and include large fish kills due to toxic algal blooms or low temperatures, seagrass die-off and recolonization, mangrove loss, wetland loss, channel dredging, modification of freshwater-inflow patterns, species introductions, and fisheries harvests (Romans 1776; Heilprin 1887; Brice 1897; Henshall 1898; Taylor and Saloman 1968b; Barnett 1972; Courtenay et al. 1974; Lewis 1976; Lewis et al. 1985; Bruger and Haddad 1986; Erftemeijer and Lewis 2006).

Fish species, or their fossilized body parts, are reported from the Tampa Bay area from approximately the Miocene to the present. In this study, we document the known tenure of various species in the Tampa Bay area in order to estimate the longevity of the co-evolved regional fish assemblage and to determine any recent species losses or gains experienced by this assemblage. We examined museum collections (both palaeontological and recent material), historical literature, and recent field-collection data for this work, but we must introduce one caveat: this work must be considered preliminary because none of these data sources have been exhausted, although we believe that we have examined 80-90% of the relevant information. This information is of obvious historical interest, but it also provides important background data for restoration efforts. In particular, we want to determine if the major species comprising the co-evolved fish assemblage in Tampa Bay area are still present and available for recolonizing newly restored habitats. Especially important in this regard would be losses of estuarine residents with limited inter-estuary dispersal capability (e.g., killifishes of the families Fundulidae and Cyprinodontidae).

METHODS

The geographical and ecological boundaries of the study area differed somewhat between palaeontological and historical data. Fossil occurrences were documented for the three counties surrounding the present Tampa Bay: Hillsborough, Pinellas, and Manatee. We realize that some of these fossil beds may lie closer to other present-day systems, such as Sarasota Bay, but given the vast changes in shoreline configuration over the past 18 million years, we concluded that the best approximation of the ancient Tampa Bay ichthyofaunal composition would be obtained by examining regional occurrences. Historical occurrences were included only for the area from the 10-m bathymetric contour (generally <10 miles west of the barrier islands and beaches of Pinellas County) to salinities ≥ 0.5 in the tidal tributaries of Tampa Bay. Obviously, the latter boundary is not stationary, but it does represent an ecological distinction between fishes of marine–brackish waters and those restricted to freshwater. The northern and southern boundaries of the Tampa Bay area were set at approximately the southern end of Honeymoon Island in St. Joseph Sound and the northern end of Anna Maria Sound, respectively.

Palaeontological data were obtained from the Vertebrate Paleontology Master Database of the Florida Museum of Natural History (all URLs of this database and others used in this study are listed at the end of this document). We did not consider the many fossils not identified to the species level, although in some cases these fossils do indicate ancient occurrences of genera that are currently monotypic in the Tampa Bay area (e.g., *Centropomus*). The fossil record is at best incomplete, and parts of certain fishes (e.g., teeth of elasmobranchs and larger otoliths of teleosts) make those fishes more likely to be detected in the fossil record. Therefore, we consider the fossil record as primarily an indicator of the presence of a given species and not necessarily the absence of any species.

Electronic museum databases examined for recent occurrences include those of the Academy of Natural Sciences of Philadelphia (ANSP), the American Museum of Natural History (AMNH), the California Academy of Sciences (CAS), the Florida Fish and Wildlife Research Institute (FSBC), the Field Museum of Natural History (FMNH), the Florida Museum of Natural History (FLMNH), the Museum of Comparative Zoology (MCZ), the University of Michigan Museum of Zoology (UMMZ), and the United States National Museum (USNM).

More than 200 publications were reviewed for fish records from Tampa Bay, but other publications that may contain relevant records have yet to be reviewed. Due to the space limitations for contributions to this symposium, we do not list all of these publications in the References section, but a complete list is available from the first author.

Unpublished data sets considered in this study included that (CZM data) from a survey of the fish fauna of the Little Manatee River conducted during 1988–91 with the help of Coastal Zone Management funds and another (FIM data) from a monitoring program conducted in Tampa Bay during 1989–2009 by the Fisheries-Independent Monitoring Program of the Florida Fish and Wildlife Research Institute.

We employed our best professional judgment in determining the validity of species records. Criteria used in this process included the existence of a description or illustration of a species in a published work, personal knowledge of the fish-identification expertise of authors of publications or identifiers of museum specimens, personal examination of specimens, and the existence of similar species in the system that may be easily misidentified. Extreme examples of the last criterion might include the whale shark, *Rhincodon typus*, which is similar to no other species, and silversides of the genus *Menidia*, which are very similar and can easily be confused with one another. Many museum records contained no indication of who made the identification, and we

did not examine most such specimens and consider their identification tentative. Another obvious factor that entered into our acceptance or rejection of a record was taxonomic change. A good example of the effects of taxonomic change is found in the halfbeaks of the genus *Hyporhamphus*. One can find many records of *H. unifasciatus* from the Tampa Bay area prior to 1993, the year in which the new species *H. meeki* was originally described (Banford and Collette 1993). *Hyporhamphus meeki* is by far the more abundant of the two species in Tampa Bay, but *H. unifasciatus* does occur here. Therefore, records of *H. unifasciatus* prior to 1993 cannot be assigned with complete confidence to either species unless the authors provide a detailed description of their specimens.

We report estimated abundance of species on a log scale based on available records. A species was deemed *rare* if it was known from <10 specimens, *occasional* if known from 11 to 100 specimens, *common* if known from 100 to 1,000 specimens, and *abundant* if known from >1,000 specimens. This scale underrepresents large species such as sharks and overrepresents small, schooling species such as anchovies. Also underrepresented are ecologically cryptic species that are difficult to collect with standard gear (e.g., seines and trawls) because they burrow or dwell in crevices within structured habitats (e.g., reefs, rock jetties, oyster reefs).

RESULTS

Prehistoric records—Approximately 80% of the fish species recorded in the fossil record from the Tampa Bay region are still present there. Records in the FLMNH indicate the occurrence of 59 species in the Tampa Bay region from the Miocene to the Pleistocene (approximately 24 million to 10,000 years before present) (Table 1). Of these species, 46 still occur in the region, but of those, we have found no records for three species within the study area as defined for historical data above (i.e., they only occur only in nearby freshwater or marine habitats). Another four species are not found in the region but are extant; for each of those species, related species do occur in the region. Eight other species once found in the region are extinct, including some rather impressive animals such as the giant shark *Carcharocles megalodon*, which probably reached a size of 17–18 m and was present in the region as recently as the Pleistocene.

Historical records—The first historical records that we can assign to a known species are from a period in the late 18th century when the British controlled Florida. The first two species clearly documented from Tampa Bay were the red drum, *Sciaenops ocellatus*, and the striped mullet, *Mugil cephalus*. These records are both from Stork (1769), who indicated that “mullet” and “bass” were the “chief” food fishes in Tampa Bay. Other literature from this period and the abundance of red drum in Tampa Bay clearly identify red drum as Stork’s “bass.” Striped mullet are by far the most likely candidate for a “chief” food fish among the three common species of mullet in Tampa Bay.

After the work of Stork (1769), new records of species from Tampa Bay did not appear for a century, and the rate of accumulation of new species after 1869 resembles a step function (Figure 1). Large inter-decadal increases in the rate of documentation of new species are attributable to particular publications or collections. Prominent among the publications were the first synopsis of the fishes of Florida by Evermann and Kendall (1900) and the first synopsis of the fishes of Tampa Bay by Springer and Woodburn (1960). Prominent collections included a large number of fishes in UMMZ collected during the late 1930s and early 1940s from near the mouth of Tampa Bay and the collections of the FIM Program beginning in 1989.

Although we located records of more than 300 species from Tampa Bay, at least 100 of them are not regular members of the ichthyofauna. We eliminated all but 195 species from the list by removing species represented by <10 individuals and by removing records for which the

identification was questionable (Table 2). Additional species could have been removed from the list due to paucity of independent occurrences, although those species are represented by more than 10 individuals (e.g., sargassumfish, *Histrio histrio*, and Atlantic cutlassfish, *Trichiurus lepturus*).

Of the 195 Tampa Bay species listed in Table 2, the majority have been known from the bay for at least several decades and have been reported from the bay recently (Table 2). Ten species have been absent in records for two or more decades, and all of those species either are regular members of the nearshore marine ichthyofauna that only occasionally wander into our study area (e.g., reef croaker, *Odontoscion dentex*) or are very difficult to collect because of their burrowing habits (e.g., spotted spoon-nose eel, *Echiophis intertinctus*). Nineteen species were first reported from the bay in the last two decades. Thirteen of those species are well-known from the freshwater tributaries to Tampa Bay but are not commonly reported from salinities >0.5 (e.g., several sunfish species, *Lepomis*). The remaining six relative newcomers are cryptic in terms of habitat or taxonomy, are recently established exotics, or are recent immigrants from other parts of Florida. The tiger goby, *Elacatinus macrodon*, was first reported in the CZM and FIM studies but was known from nearshore reefs as early as the 1960s (Springer and Woodburn 1960). This tiny species (maximum total length of 50 mm) occurs in the interstices of encrusting animal or plant communities on hard substrates such as seawalls. These habitats are difficult to sample with standard collecting gear such as seines or trawls, and the tiger goby may have been missed by most sampling programs. The spotfin mojarra, *Eucinostomus argenteus*, regularly enters many Florida estuaries in the late summer or fall but is extremely difficult to separate from the much more common species, *E. gula* and *E. harengulus*. Records of *E. argenteus* prior to 1984, when a new diagnosis for this species was published by Matheson and McEachran (1984), are likely based on *E. harengulus*, but we have well-documented records of this species from as early as the 1990s from Tampa Bay. Based primarily upon the FIM data, two exotic species appear to have recently become established in Tampa Bay. These species are pike killifish, *Belonesox belizanus*, and African jewelfish, *Hemichromis letourneuxi*, and both are represented by numerous specimens collected in multiple years. Two species appear to have immigrated to Tampa Bay and become well-established over the past two decades (the possibility of human introduction of either species cannot be discounted but seems unlikely). The emerald goby, *Ctenogobius smaragdus*, has been known from the Charlotte Harbor area since at least 1891 (Henshall, 1891) but was not reported from the Tampa Bay area until the 1990s (Ellis and Bell 2004; Krebs et al. 2007; FIM data). This species has also been recorded north of Tampa Bay in the Cedar Key area in recent years (FIM data). The emerald goby is not particularly cryptic in either habitat or taxonomy and would probably have been identified by previous workers if they had collected it. The crested goby, *Lophogobius cyprinoides*, has been known from southwest Florida since at least the 1950s (Kushlan and Lodge 1974), but we find no record of it from Tampa Bay prior to 2006 (76 specimens in the FIM database were collected from 2006 through 2008). The crested goby has also been collected north of Tampa Bay in recent years: one collection from Walton County in the Florida Panhandle (FLMNH data) and several collections from the Weeki Wachee River (FIM data). Like the emerald goby, the crested goby is not cryptic in either habitat or taxonomy.

Finally, two large species that were probably formerly abundant in Tampa Bay are now extremely rare. Atlantic sturgeon (Gulf subspecies), *Acipenser oxyrinchus*, briefly supported a commercial fishery in Tampa Bay. The following statement is from Townsend (1902):

“The catch in Tampa Bay, on the western side of the State, during the winter of 1886 and 1887 was about 1,500 sturgeon, which yielded 5,000 pounds of caviar. The following season over 2,000 fish were secured and nearly 6,000 pounds of caviar prepared. The next season only 7 fish were taken and the fishery was then abandoned.”

We have located records of fewer than 10 specimens of sturgeon from Tampa Bay since the early 1900s (Springer 1961; Hoehn 1998; FLMNH data; FIM data). The federally endangered smalltooth sawfish, *Pristis pectinata*, may have also been abundant in Tampa Bay at one time, although data are somewhat sparse. Henshall (1895) reported a single specimen from Tampa Bay but indicated that the species was common along both coasts of Florida and was “always killed by the fishermen when captured”. Springer and Woodburn (1960) collected two specimens in Old Tampa Bay in 1957 and 1959 but indicated that “Mr. John Hurlbut, operator of a commercial aquarium, told us that until a few years ago specimens (*Pristis* spp.) were frequently caught in Hillsborough Bay”. The most recent record that we find is from 1999 (FIM data).

DISCUSSION

Despite fluctuations in abundance of various species (e.g., Matheson et al. 2005), the species composition of the ichthyofauna of Tampa Bay has changed little during historical times, and approximately 15% of the current ichthyofauna has been in the area since at least the Pleistocene. Most of the species that have been absent for two or more decades or have been recorded only within the past two decades are well known from nearby marine or freshwater habitats, are cryptic in either habitat or taxonomy, or are recently established exotics. Two species appear to be new immigrants to the system, and two species that may once have been common have apparently been nearly extirpated. The two new immigrants have apparently expanded their ranges to the north in recent years, and, as for the tropical exotic species, we do not yet know how the cold kill of 2009/2010 has affected these animals. The two nearly extirpated species appear to have both fallen prey to fishing activities.

The management conclusion that we draw from these data is that, despite the numerous stresses placed on the Tampa Bay system in historical times, the core of the ichthyofauna is still present. This fact bodes well for restoration efforts because it means that a long-established, co-evolved assemblage of fishes is still in the system and is available for recolonization of restored habitats. This is especially important for estuarine residents with little dispersal ability that would have difficulty recolonizing from outside Tampa Bay.

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URLS OF MUSEUM DATABASES USED IN THIS STUDY

ANSP <http://clade.ansp.org/ichthyology/collections/fish.php>
 AMNH <http://entheros.amnh.org/db/emuwebamnh/logon.php>
 CAS <http://research.calacademy.org/redirect?url=http://researcharchive.calacademy.org/research/Ichthyology/collection/index.asp>
 FMNH <http://emuweb.fieldmuseum.org/fish/Query.php>
 FLMNH (palaeontology) http://www.flmnh.ufl.edu/scripts/DBs/vp_uf_pub.asp
 FLMNH (recent) http://www.flmnh.ufl.edu/scripts/dbs/fish_pub.asp
 MCZ <http://collections.mcz.harvard.edu/Fish/FishSearch.htm>
 UMMZ <http://141.213.176.15/UMMZ/>
 USNM <http://nhb-acsmith2.si.edu/emuwebvzfishesweb/pages/nmnh/vz/DtlQueryFishes.php>

Table 1. Prehistoric occurrences of fishes in the Tampa Bay region. All data were taken from the electronic vertebrate paleontology database of the Florida Museum of Natural History and include fossils identified to species from Hillsborough, Pinellas, and Manatee counties. Categories as follows: present = currently present in area; present (fw) = currently present in general area but limited to freshwater; present (marine) = currently present in general area but only in marine waters beyond the 10-m depth contour; absent (extant) = not present in area but extant; absent (extinct) = extinct worldwide.

Species	Miocene	Pliocene	Pleistocene	Current Status in Area
<i>Aetobatus narinari</i>			present	present
<i>Ameiurus natalis</i>			present	present (fw)
<i>Ameiurus nebulosus</i>			present	present
<i>Amia calva</i>			present	present
<i>Anguilla rostrata</i>			present	present
<i>Archosargus probatocephalus</i>			present	present
<i>Ariopsis felis</i>			present	present
<i>Atractosteus spatula</i>	present	present	present	absent (extant)
<i>Bagre marinus</i>			present	present
<i>Caranx hippos</i>			present	present
<i>Carcharhinus acronotus</i>			present	present
<i>Carcharhinus brevipinna</i>			present	present
<i>Carcharhinus limbatus</i>			present	present
<i>Carcharhinus obscurus</i>			present	present
<i>Carcharhinus plumbeus</i>			present	present (marine)
<i>Carcharias cuspidata</i>	present			absent (extinct)
<i>Carcharias macrota</i>	present			absent (extinct)
<i>Carcharias taurus</i>	present	present	present	present
<i>Carcharocles megalodon</i>	present	present	present	absent (extinct)
<i>Carcharodon carcharias</i>			present	present
<i>Chaetodipterus faber</i>			present	present
<i>Cynoscion nebulosus</i>			present	present
<i>Cyprinodon variegatus</i>			present	present
<i>Elops saurus</i>			present	present
<i>Erimyzon sucetta</i>			present	present
<i>Floridichthys carpio</i>			present	present
<i>Fundulus heteroclitus</i>			present	absent (extant)
<i>Fundulus seminolis</i>			present	present
<i>Fundulus similis</i>			present	Present
<i>Galeocерdo contortus</i>	present			absent (extinct)
<i>Galeocерdo cuvier</i>	present	present	present	present
<i>Gambusia affinis</i>	present	present	present	absent (extant)
<i>Ginglymostoma serra</i>			present	absent (extinct)

Species	Miocene	Pliocene	Pleistocene	Current Status in Area
<i>Isurus hastalis</i>	present	present	present	absent (extinct)
<i>Isurus oxyrinchus</i>			present	present (marine)
<i>Jordanella floridae</i>	present	present	present	present
<i>Lachnolaimus maximus</i>			present	present
<i>Lepisosteus oculatus</i>			present	absent (extant)
<i>Lepisosteus osseus</i>			present	present
<i>Lepomis auritus</i>			present	present
<i>Lepomis gulosus</i>	present	present	present	present
<i>Lepomis microlophus</i>			present	present
<i>Megalops atlanticus</i>			present	present
<i>Micropterus salmoides</i>	present	present	present	present
<i>Morone saxatilis</i>			present	absent (extant)
<i>Negaprion brevirostris</i>	present	present	present	present
<i>Notemigonus crysoleucas</i>			present	present
<i>Poecilia latipinna</i>	present	present	present	present
<i>Pogonias cromis</i>	present	present	present	present
<i>Pomoxis nigromaculatus</i>			present	present
<i>Rhinoptera bonasus</i>	present	present	present	present
<i>Rhizoprionodon terraenovae</i>	present	present	present	present
<i>Sciaenops ocellatus</i>			present	present
<i>Sphyraena barracuda</i>	present	present	present	present
<i>Sphyrna mokarran</i>			present	present
<i>Trachinotus carolinus</i>			present	Present

Table 2. Fishes reliably reported from Tampa Bay and known from more than 10 specimens. Abundance categories as follows: abundant = >1,000 specimens, common = 101-1,000, occasional = 11-100. Columns labeled "First" and "Last" refer to first and last records from Tampa Bay based on either literature, museum records (mostly unverified), or unpublished data from FWRI.

Species	Common Name	Abundance	First	Last
<i>Mugil cephalus</i>	striped mullet	abundant	1769	2008
<i>Sciaenops ocellatus</i>	red drum	abundant	1769	2008
<i>Archosargus probatocephalus</i>	sheepshead	abundant	1869	2008
<i>Anchoa hepsetus</i>	striped anchovy	abundant	1880	2008
<i>Harengula jaguana</i>	scaled sardine	abundant	1880	2008
<i>Poecilia latipinna</i>	sailfin molly	abundant	1880	2008
<i>Eucinostomus gula</i>	silver jenny	abundant	1880	2008
<i>Eucinostomus harengulus</i>	tidewater mojarra	abundant	1880	2008
<i>Gymnothorax saxicola</i>	honeycomb moray	occasional	1880	1960
<i>Echiophis intertinctus</i>	spotted spoon-nose eel	occasional	1880	1977
<i>Halichoeres bivittatus</i>	slippery dick	occasional	1880	2008
<i>Trachinotus carolinus</i>	Florida pompano	abundant	1882	2008
<i>Opisthonema oglinum</i>	Atlantic thread herring	abundant	1884	2008
<i>Syngnathus scovelli</i>	Gulf pipefish	abundant	1884	2008
<i>Gobiesox strumosus</i>	skilletfish	abundant	1884	2008

Species	Common Name	Abundance	First	Last
<i>Chaetodipterus faber</i>	Atlantic spadefish	abundant	1884	2008
<i>Achirus lineatus</i>	lined sole	abundant	1884	2008
<i>Symphurus plagiusa</i>	blackcheek tonguefish	abundant	1884	2008
<i>Rhinobatos lentiginosus</i>	Atlantic guitarfish	common	1884	2008
<i>Myrophis punctatus</i>	speckled worm eel	common	1884	2009
<i>Ogcocephalus cubifrons</i>	polka-dot batfish	common	1884	2008
<i>Scorpaena brasiliensis</i>	barbfish	common	1884	2008
<i>Peprilus paru</i>	harvestfish	common	1884	2007
<i>Aluterus schoepfii</i>	orange filefish	common	1884	2008
<i>Bascanichthys scuticaris</i>	whip eel	occasional	1884	1974
<i>Antennarius ocellatus</i>	ocellated frogfish	occasional	1884	1963
<i>Sphyrna tiburo</i>	bonnethead	abundant	1885	2008
<i>Ariopsis felis</i>	hardhead catfish	abundant	1885	2008
<i>Fundulus grandis</i>	Gulf killifish	abundant	1885	2008
<i>Fundulus similis</i>	longnose killifish	abundant	1885	2008
<i>Cyprinodon variegatus</i>	sheepshead minnow	abundant	1885	2008
<i>Prionotus tribulus</i>	bighead searobin	abundant	1885	2008
<i>Caranx hippos</i>	crevalle jack	abundant	1885	2008
<i>Lagodon rhomboids</i>	Pinfish	Abundant	1885	2008
<i>Chasmodes saburrae</i>	Florida blenny	abundant	1885	2008
<i>Microgobius gulosus</i>	clown goby	abundant	1885	2008
<i>Sphoeroides nephelus</i>	southern puffer	abundant	1885	2008
<i>Hyporhamphus unifasciatus</i>	Atlantic silverstripe halfbeak	occasional	1885	2008
<i>Anarchopterus criniger</i>	fringed pipefish	occasional	1886	2008
<i>Cynoscion nebulosus</i>	spotted seatrout	abundant	1887	2008
<i>Syngnathus louisianae</i>	chain pipefish	abundant	1889	2008
<i>Centropristis striata</i>	black sea bass	abundant	1889	2008
<i>Leiostomus xanthurus</i>	spot	abundant	1889	2008
<i>Menticirrhus littoralis</i>	Gulf kingfish	abundant	1889	2008
<i>Hippocampus erectus</i>	lined seahorse	common	1889	2008
<i>Citharichthys macrops</i>	spotted whiff	common	1889	2008
<i>Dasyatis say</i>	bluntnose stingray	abundant	1891	2008
<i>Mugil curema</i>	white mullet	abundant	1891	2008
<i>Strongylura marina</i>	Atlantic needlefish	abundant	1891	2008
<i>Gambusia holbrooki</i>	eastern mosquitofish	abundant	1891	2008
<i>Scomberomorus maculatus</i>	Spanish mackerel	abundant	1891	2008
<i>Chilomycterus schoepfii</i>	striped burrfish	abundant	1891	2008
<i>Notemigonus crysoleucas</i>	golden shiner	common	1891	2007
<i>Micropterus salmoides</i>	largemouth bass	common	1891	2008
<i>Dasyatis sabina</i>	Atlantic stingray	abundant	1892	2008
<i>Elops saurus</i>	ladyfish	abundant	1892	2008

Species	Common Name	Abundance	First	Last
<i>Synodus foetens</i>	inshore lizardfish	abundant	1892	2008
<i>Centropomus undecimalis</i>	common snook	abundant	1892	2008
<i>Pomatomus saltatrix</i>	bluefish	abundant	1892	2008
<i>Trachinotus falcatus</i>	permit	abundant	1892	2008
<i>Menticirrhus saxatilis</i>	northern kingfish	abundant	1892	2008
<i>Micropogonias undulatus</i>	Atlantic croaker	abundant	1892	2008
<i>Pogonias cromis</i>	black drum	abundant	1892	2008
<i>Dasyatis americana</i>	southern stingray	common	1892	2008
<i>Ancyloperetta quadrocellata</i>	ocellated flounder	common	1892	2008
<i>Albula vulpes</i>	bonefish	occasional	1892	2008
<i>Lobotes surinamensis</i>	Atlantic tripletail	occasional	1892	2009
<i>Kyphosus sectatrix</i>	Bermuda chub	occasional	1892	1977
<i>Anchoa mitchilli</i>	bay anchovy	abundant	1893	2008
<i>Opsanus beta</i>	Gulf toadfish	abundant	1893	2008
<i>Selene vomer</i>	lookdown	abundant	1893	2008
<i>Lutjanus griseus</i>	gray snapper	abundant	1893	2008
<i>Lutjanus synagris</i>	lane snapper	abundant	1893	2008
<i>Diplodus holbrookii</i>	spottail pinfish	abundant	1893	2008
<i>Bairdiella chrysoura</i>	silver perch	abundant	1893	2008
<i>Acanthostracion quadricornis</i>	scrawled cowfish	abundant	1893	2008
<i>Megalops atlanticus</i>	tarpon	common	1893	2007
<i>Sardinella aurita</i>	Spanish sardine	abundant	1896	2008
<i>Mugil gyrans</i>	whirligig mullet	abundant	1898	2008
<i>Adinia xenica</i>	diamond killifish	abundant	1898	2008
<i>Floridichthys carpio</i>	goldspotted killifish	abundant	1898	2008
<i>Prionotus scitulus</i>	leopard searobin	abundant	1898	2008
<i>Etropus crossotus</i>	fringed flounder	abundant	1898	2008
<i>Trinectes maculatus</i>	hogchoker	abundant	1898	2008
<i>Diplectrum formosum</i>	sand perch	common	1898	2008
<i>Porichthys plectrodon</i>	Atlantic midshipman	occasional	1898	1967
<i>Lactophrys trigonus</i>	trunkfish	occasional	1898	2007
<i>Echeneis neucratoides</i>	whitefin sharksucker	occasional	1904	2008
<i>Narcine bancroftii</i>	lesser electric ray	occasional	1911	1990
<i>Sphyrna barracuda</i>	great barracuda	common	1912	2008
<i>Rhizoprionodon terraenovae</i>	Atlantic sharpnose shark	occasional	1914	1993
<i>Urophycis floridana</i>	southern hake	common	1923	2008
<i>Calamus arctifrons</i>	grass porgy	common	1930	2008
<i>Fundulus confluentus</i>	marsh killifish	abundant	1934	2008
<i>Lucania parva</i>	rainwater killifish	abundant	1935	2008
<i>Menidia beryllina</i>	inland silverside	abundant	1936	2003
<i>Brevoortia smithi</i>	yellowfin menhaden	abundant	1937	2008

Species	Common Name	Abundance	First	Last
<i>Caranx crysos</i>	blue runner	common	1937	2008
<i>Epinephelus morio</i>	red grouper	occasional	1937	2008
<i>Bagre marinus</i>	gafftopsail catfish	abundant	1938	2008
<i>Membras martinica</i>	rough silverside	abundant	1938	2008
<i>Mycteroperca microlepis</i>	gag	abundant	1938	2008
<i>Chloroscombrus chrysurus</i>	Atlantic bumper	abundant	1938	2008
<i>Gobiosoma robustum</i>	code goby	abundant	1938	2008
<i>Carcharhinus acronotus</i>	blacknose shark	common	1938	2008
<i>Gymnura micrura</i>	smooth butterfly ray	common	1938	2008
<i>Echeneis naucrates</i>	sharksucker	common	1938	2004
<i>Haemulon aurolineatum</i>	tomtate	common	1938	1998
<i>Nicholsina usta</i>	emerald parrotfish	common	1938	2008
<i>Monacanthus ciliatus</i>	fringed filefish	Common	1938	2008
<i>Hemicaranx amblyrhynchus</i>	bluntnose jack	occasional	1938	2006
<i>Lachnolaimus maximus</i>	hogfish	occasional	1938	2008
<i>Gymnachirus melas</i>	naked sole	occasional	1938	1989
<i>Hippocampus zosterae</i>	dwarf seahorse	abundant	1939	2008
<i>Eugerres plumieri</i>	striped mojarra	abundant	1939	2008
<i>Cynoscion arenarius</i>	sand seatrout	abundant	1939	2008
<i>Menticirrhus americanus</i>	southern kingfish	abundant	1939	2008
<i>Negaprion brevirostris</i>	lemon shark	occasional	1939	2008
<i>Decapterus punctatus</i>	round scad	occasional	1940	1998
<i>Stephanolepis hispidus</i>	planehead filefish	abundant	1941	2008
<i>Strongylura timucu</i>	timucu	abundant	1948	2008
<i>Syngnathus floridae</i>	dusky pipefish	abundant	1948	2008
<i>Bathygobius soporator</i>	frillfin goby	abundant	1948	2008
<i>Syacium papillosum</i>	dusky flounder	occasional	1948	2010
<i>Paraclinus marmoratus</i>	marbled blenny	occasional	1951	2008
<i>Oligoplites saurus</i>	leatherjack	abundant	1952	2008
<i>Raja eglanteria</i>	clearnose skate	occasional	1953	1998
<i>Aetobatus narinari</i>	spotted eagle ray	occasional	1953	2008
<i>Gobiosoma bosc</i>	naked goby	abundant	1957	2008
<i>Odontoscion dentex</i>	reef croaker	common	1957	1976
<i>Ophichthus gomesii</i>	shrimp eel	occasional	1957	1997
<i>Histrio histrio</i>	sargassumfish	occasional	1957	1957
<i>Caranx latus</i>	horse-eye jack	occasional	1957	2008
<i>Gobionellus oceanicus</i>	highfin goby	occasional	1957	2007
<i>Rhinoptera bonasus</i>	cownose ray	abundant	1958	2008
<i>Anchoa cubana</i>	Cuban anchovy	abundant	1958	2008
<i>Ameiurus nebulosus</i>	brown bullhead	abundant	1958	2005
<i>Fundulus seminolis</i>	Seminole killifish	abundant	1958	2008
<i>Sarotherodon melanotheron</i>	blackchin tilapia	abundant	1958	2008

Species	Common Name	Abundance	First	Last
<i>Carcharhinus leucas</i>	bull shark	common	1958	2009
<i>Lepisosteus osseus</i>	longnose gar	common	1958	2008
<i>Hyporhamphus meeki</i>	false silverstripe halfbeak	common	1958	2008
<i>Sphyræna borealis</i>	northern sennet	common	1958	2008
<i>Anguilla rostrata</i>	American eel	occasional	1958	2004
<i>Tylosurus crocodilus</i>	houndfish	occasional	1958	2008
<i>Rachycentron canadum</i>	cobia	occasional	1958	2007
<i>Pareques umbrosus</i>	cubbyu	occasional	1958	1976
<i>Hypleurochilus caudovittatus</i>	zebratail blenny	occasional	1958	2008
<i>Parablennius marmoreus</i>	seaweed blenny	Occasional	1958	2007
<i>Trichiurus lepturus</i>	Atlantic cutlassfish	occasional	1958	1971
<i>Ctenogobius boleosoma</i>	darter goby	common	1959	2006
<i>Lepomis macrochirus</i>	bluegill	abundant	1960	2008
<i>Carcharhinus limbatus</i>	blacktip shark	common	1960	2008
<i>Hemiramphus brasiliensis</i>	ballyhoo	occasional	1960	2008
<i>Epinephelus itajara</i>	goliath grouper	occasional	1960	2008
<i>Gobiosoma longipala</i>	twoscale goby	occasional	1960	2008
<i>Dorosoma cepedianum</i>	gizzard shad	occasional	1961	2006
<i>Ophidion josephi</i>	crested cusk-eel	occasional	1963	1994
<i>Dactyloscopus moorei</i>	speckled stargazer	occasional	1963	2007
<i>Lupinoblennius nicholsi</i>	highfin blenny	occasional	1963	2008
<i>Dorosoma petenense</i>	threadfin shad	abundant	1965	2008
<i>Oreochromis aureus</i>	blue tilapia	occasional	1968	2008
<i>Sphyræna guachancho</i>	guaguanche	occasional	1971	2004
<i>Rivulus marmoratus</i>	mangrove rivulus	occasional	1978	2007
<i>Diapterus auratus</i>	Irish pompano	occasional	1978	2005
<i>Serraniculus pumilio</i>	pygmy sea bass	occasional	1979	2007
<i>Ameiurus catus</i>	white catfish	common	1990	2008
<i>Lucania goodei</i>	bluefin killifish	common	1990	2008
<i>Lepomis microlophus</i>	redeer sunfish	common	1990	2008
<i>Lepomis punctatus</i>	spotted sunfish	common	1990	2008
<i>Elacatinus macrodon</i>	tiger goby	occasional	1990	2008
<i>Notropis petersoni</i>	coastal shiner	abundant	1991	2008
<i>Labidesthes sicculus</i>	brook silverside	common	1991	2008
<i>Heterandria formosa</i>	least killifish	common	1991	2008
<i>Lepisosteus platyrhincus</i>	Florida gar	occasional	1991	2008
<i>Amia calva</i>	bowfin	occasional	1991	2008
<i>Notropis maculatus</i>	taillight shiner	occasional	1991	2007
<i>Eucinostomus argenteus</i>	spotfin mojarra	occasional	1991	2008
<i>Belonesox belizanus</i>	pike killifish	common	1995	2008
<i>Ictalurus punctatus</i>	channel catfish	occasional	1996	2006
<i>Ctenogobius smaragdus</i>	emerald goby	occasional	1996	2008

Species	Common Name	Abundance	First	Last
<i>Hemichromis letourneuxi</i>	African jewelfish	occasional	2000	2008
<i>Lepomis marginatus</i>	dollar sunfish	occasional	2001	2007
<i>Lophogobius cyprinoides</i>	crested goby	occasional	2006	2008

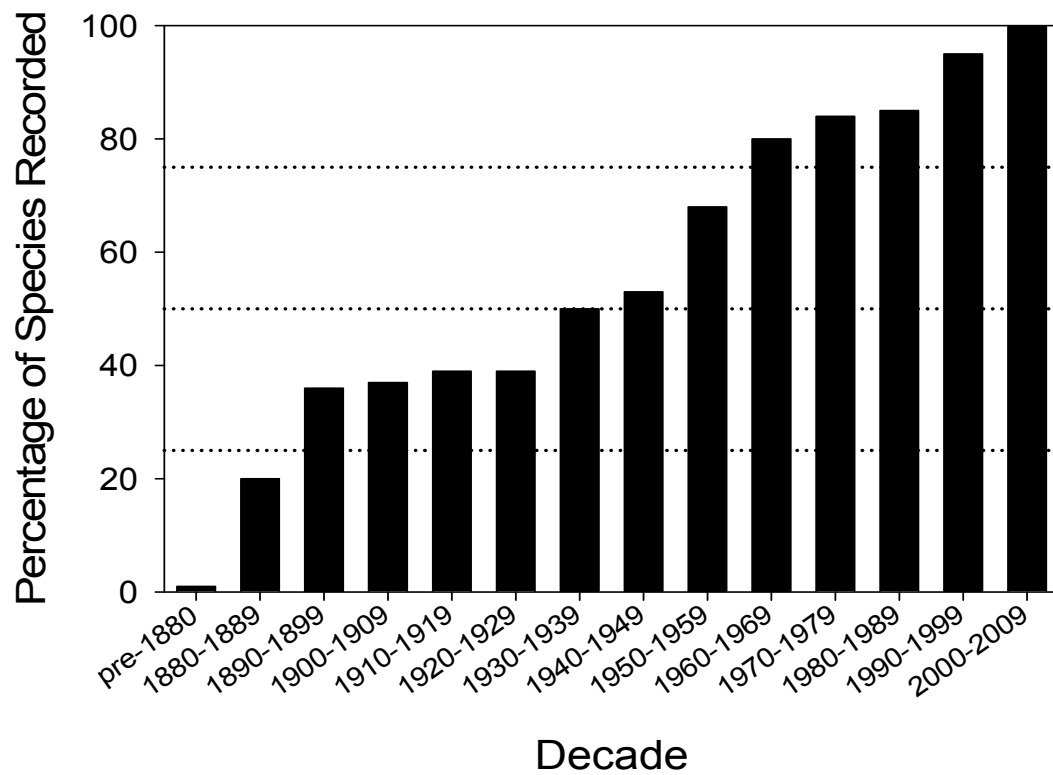


Figure 1. Species recorded from Tampa Bay during historical times. Bars represent cumulative percentages of the current total of 309 species.

ASSESSING THE INFLUENCE OF BOTTOM-UP AND TOP-DOWN PROCESSES IN TAMPA BAY USING ECOPATH WITH ECOSIM

D. Chagaris & B. Mahmoudi

ABSTRACT

Drastic changes have occurred in Tampa Bay over the last 50-60 years, such as an increase and then decline in nutrient loads to the bay, loss and recovery of seagrass habitat, and the evolution of fishing regulations in response to declining fish stocks. To explore the combined impacts of these changes on fish and invertebrate populations in Tampa Bay, we constructed a multi-species/fleet ecosystem model using Ecopath with Ecosim (EwE). Our model consists of 74 trophically-linked biomass “pools” in the form of functional groups, single species, or age classes of single species and ranging from detritus to top predators. In Ecosim, time series of fishing mortalities and fishing effort were used to calibrate the model for the period 1950-2007 in order to replicate historical trends in abundance and catch. Nitrogen load estimates to Tampa Bay were included in Ecosim as an environmental forcing function. The relative influence that bottom-up and top-down control has on certain species was evaluated using the weighted sum of squares of deviations (SS) between observed and predicted biomass and comparing this measure to a baseline scenario with neither environmental forcing nor fishing mortality included. The results varied across species with some undergoing stronger top-down control (spotted seatrout), others influenced more by bottom-up processes (anchovies), and several functioning under mixed control (blue crab). This exercise provides a relatively simple approach to identify whether top-down or bottom-up forces are influencing the abundance of various species in Tampa Bay.

INTRODUCTION

Since the mid 1900s, drastic changes in fishing activity and external nutrient loading have taken place in Tampa Bay. Zarbock et al (1994) estimated nitrogen loads to be 1,915 tons/yr in 1938; in 1976 following rapid population growth and development in the Tampa Bay, the estimate was nearly 10,000 tons/yr. The large increase of nitrogen entering the bay, mostly from raw sewage, was believed to be the cause of poor water quality, algal blooms, and foul odors. As a result, pollution abatement measures, such as advanced wastewater treatment and stormwater management permitting, were taken to reduce the amount of point-source nitrogen entering Tampa Bay (Lewis et al 1998). These measures resulted in an almost 60% decrease in total nitrogen from 1976 to the period of 1985-1991 (Zarbock et al. 1994) where they have remained relatively stable, with fluctuations largely associated with rainfall amounts.

Commercial finfish landings in Tampa Bay, for fish commonly found in bay waters (i.e. excluding offshore species) peaked in 1965 with over 13,000,000 lbs/yr and then steadily declined to around 7,000,000 lbs/yr in the early 1970s (Lombardo et al. 1985) with current levels at approximately 4,000,000 lbs/yr. Shellfish landings in Tampa Bay peaked in the mid-1950s at around 19,000,000 lbs/yr, drastically declined through the 1960s to about 5,000,000 lbs/yr in the 1970s (Lombardo et al. 1985) and are currently at a level less than 2,000,000 lbs/yr. While some of the declines in commercial landings are due to decreased stock sizes, much of the reduction in commercial catch over the last 2-3 decades is a result of increasing regulations in the 1980s and especially the 1994 Florida net ban that limited the use of entangling nets in Florida waters and greatly restricted the use of other commercial gear. During the same time period, recreational fishing effort increased greatly, as indicated by at least a 300% rise in boat registrations in Tampa Bay from 1964 to 1980 (Lombardo et al. 1985). Since recreational fishing data collection began in 1982, the estimated number of angler trips taken in Tampa Bay has increased by nearly 5% each year.

Nutrient loading, a type of bottom-up process, drives primary production in estuaries and alters the amount of food available at the bottom of the food-web, leading to noticeable changes throughout the ecosystem. Phytoplankton generally respond positively to an increase in nutrients, then zooplankton and their fish predators benefit from this rise in food production. Phytoplankton that is not consumed is deposited to the sediment where it can stimulate benthic production (Pinckney et al. 2001). Increased nutrient loading can also have serious adverse effects on an ecosystem such as harmful algal blooms, hypoxia, poor water quality, and loss of submerged aquatic vegetation. In Tampa Bay, 46% of seagrass acreage was lost between 1950 and 1982 as a result of low light penetration caused by poor water quality due to nutrient loading (Greening et al. 2006).

The removal of biomass due to fishing not only affects the target species, but is almost certain to have an impact on their predators and prey. Thus, non-targeted species can be expected to respond to changes in the abundance of other species as a result of fishing. While top-down theory traditionally refers to the regulation of food-web components by one or more top predators via trophic cascades, in this study the term ‘top-down’ is also applied to effects of fisheries even if they target lower trophic level species. For example, the harvesting of major prey species such as blue crab or mullet would not result in a trophic cascade per se, but may have effects on the growth of their predators. These effects are nonetheless considered top-down, driven by the fishery.

With regard to ‘ecosystem-based fisheries management’, it is important to understand whether bottom-up (nutrient loading, climate) or top-down effects (fishing, predation) might be more important for one species than another. Accounting for the multiple and confounding factors between species, their environment, and fisheries production requires the use of ecosystem models that at least provide a simplified representation of the complex food-web interactions. Recently, the Ecopath with Ecosim modeling approach (Christensen and Walters 2004) has been used to tease apart the effects of fishing and environmental changes by attempting to reconstruct historical biomass trends. Results generally indicate that it is a combination of fishing and environmental forcing that explain most of the variation in observed trends of abundance (Araujo et al. 2006; Field et al. 2006; Guenette et al. 2006; Mackinson et al. 2008). The objective of this study was to assess the relative influence of nutrient loading and fishing across several species in Tampa Bay using Ecopath with Ecosim.

METHODS

Ecopath with Ecosim is a modeling framework that integrates a wide range of biological and fisheries information for multiple species in a system. A detailed description of the model formulations is given by Christensen et al (2004). Briefly, Ecopath with Ecosim combines a trophic, mass-balance analysis with a dynamic model to evaluate fisheries resources in an ecosystem context and facilitate overall ecosystem characterization and comparison. The model requires input for biomass, production, consumption, diet, and fishery harvests/discards. The major assumption of the Ecopath model is that each species must be thermodynamically mass-balanced. That is, biomass production cannot be less than biomass losses due to fishery harvests, predation, emigration, and other mortality. Further, a group cannot produce, respire, or excrete more biomass than it consumes. Once these assumptions are met, the model is passed to the time dynamic simulation, Ecosim, where it is calibrated to fit historical time series of catch and abundance.

In Ecosim, the energy flows defined by the underlying Ecopath model are turned into dynamic, time-varying predictions using differential equations that predict biomass from changes in the consumption rate of predator j on prey i according to foraging arena theory (Walters et al. 1997; Christensen et al. 2004). Foraging arena theory essentially claims that prey can be in states that are or are not vulnerable to predation. The movement of prey i , between invulnerable and vulnerable states to predator j , is known as the vulnerability flow parameter v_{ij} and can range from 1 to ∞ . The consumption rates, and resulting biomasses predicted in Ecosim are very sensitive to the v_{ij} settings, where low values imply bottom-up control and high values imply top-down control. Additionally, the vulnerabilities can be modified by seasonal or long-term forcing functions such as nutrient loading and also by including mediation effects where a third party organism may facilitate or provide protection from predation.

Changes in nutrient loading in Ecosim are simulated by assigning a user-supplied time-multiplying forcing function, f_t , that alters the total nutrient concentration N_T of the system such that $f_t = 1$ implies the base value of N_T . The total nutrient concentration is partitioned between that 'bound' in biomass versus free in the environment for uptake by primary producers. Primary production rates for producer pools are linked to the free nutrient concentration during each simulation through assumed Michaelis-Menten uptake relationships. The free nutrient concentration is available to all primary producers in the model, implying competition among all plant types for those nutrients. The sensitivity to changes in nutrient concentrations was modeled by adjusting the maximum production rate parameter (PB_{max}) to specify which primary producer will out-compete the others for the free nutrients. For this analysis the PB_{max} values for phytoplankton, algae, and seagrass were set to 3.0, 1.25, and 1.1 respectively.

The analysis presented here is based on an existing, balanced and calibrated, Ecopath with Ecosim model of Tampa Bay created by the current authors. For the sake of brevity, a detailed description of the derivation of biomass, production, consumption, diet parameters, and time series is not included. The Ecopath model input is based on averages over the time period of 2003-2007. The model consists of 74 biomass pools including: 2 detritus groups, 3 primary producers, 2 zooplankton groups, 1 dolphin group, 1 bird group, 13 invertebrate pools with blue crab and shrimp represented by juvenile and adult stanzas, and 51 fish pools ranging from sharks to gobies with some representing a specific life stage of a single species and others an aggregate of similar species. Eight fishing "fleets" represented by 7 different commercial gear types and one recreational fishery were also included in the model. In general, stock assessments were considered the most reliable and complete sources of information for fish groups. For those fish groups without stock assessments, biomass was either calculated from fisheries independent monitoring (FIM) survey data or estimated by the Ecopath model. Parameter estimates for non-fish components (detritus, primary producers, zooplankton, benthic fauna, dolphins, birds), were based on a compilation of survey estimates, literature values, or empirical relationships. Ecopath consumption rates for all consumers were either obtained directly from literature values or derived using empirical relationships. Initial parameter estimates were adjusted slightly to achieve mass-balance before proceeding to the Ecosim module.

The time-series data of relative biomass, catch, fishing mortality, and fishing effort used to calibrate the Ecosim model and estimate the vulnerability flow rates were compiled from a variety of sources. When available, biomass and fishing mortality trends generated from stock assessment models were used. For species without stock assessments, indices of abundance were generated using FIM survey data. Time series of catch was gathered using Florida Trip Ticket and Marine Recreational Fisheries Statistical Survey (MRFSS) data, reduced to landings made in Hillsborough, Pinellas, Manatee, and Sarasota counties. Fishing effort was also estimated for each fleet as the number of trips taken each year using Florida Trip Ticket and MRFSS data. The

nutrient loading time series was estimated in monthly time steps using the hydrodynamic Florida Ecosystem Model (FLEM). The FLEM model required inputs of total annual nitrogen loads (Zarbock et al. 1994; Pribble et al. 2001; Poe et al. 2005; Janicki et al. 2008), mean monthly rainfall, mean monthly river discharge, and average monthly directional wind velocities. The model provided monthly estimates of relative nutrient loading that were then supplied as a forcing function in Ecosim after being smoothed using a 12-month running average.

The calibrated Ecosim model was forced top-down by fishing mortality (or fishing effort) data, and bottom-up by nutrient-driven primary production, and simulations were run from 1950-2007. To obtain a 'best fit' Ecosim model, the vulnerability flow rates were first manually adjusted to correct for any gross divergences in predicted and observed trends in abundance and then estimated using an optimization routine in Ecosim. The optimization routine first identifies the vulnerability flow rates that have the most influence on the sum of squared differences (SS) between log-observed and log-predicted biomass by adjusting each one by 1% and rerunning the model. The routine then estimates the v_{ij} value for the n most sensitive predator-prey combinations using a Marquardt nonlinear search algorithm that minimizes the SS (Christensen et al. 2004). In order to further improve model fits, minor adjustments were also made to other, less sensitive parameters in the underlying Ecopath model and those in the Ecosim foraging arena equation.

To investigate the influence of top-down and bottom-up processes on species in Tampa Bay, we used the SS measure to evaluate the degree to which model predictions improved when fishing and nutrient loading were included as forcing factors in model simulations. Using the 'best fit' model, we first created a 'baseline' scenario to compare with subsequent runs. The 'baseline' scenario was essentially a 'flatline' run where neither fishing nor nutrient loading were included. We then ran three scenarios, one where only fishing mortality (or effort) was included, one where only nutrient loading was included, and one with both fishing and nutrient loading. The results are presented as the percent reduction of residual variance from the baseline scenario ($1 - [SS_{\text{scenario}}/SS_{\text{baseline}} \times 100]$) under each of the three scenarios.

RESULTS

The biomass, production, and consumption values used for input into the underlying, mass-balanced Ecopath model are provided in Table 1, along with estimates for the various components of mortality. For the 'baseline' and 'nutrient only' scenarios described above, the fishing mortality rate is held constant across all years, from 1950-2007, at the Ecopath baseline values provided in Table 1. Fishing mortality rates calculated by Ecopath, as catch/biomass, were consistent with fishing mortality rates calculated in the single species stock assessments. Nutrient loading calculated by the FLEM model is illustrated in Figure 1, and shows the increase in nutrient loading from 1950 to the mid 1970s and the subsequent decline resulting from pollution abatement measures. The nutrient loading forcing function shown here is the value for the nutrient multiplier f_t used to alter the total nutrient concentration in the system on a monthly time step. The pattern predicted by the FLEM model, smoothed over a 12 month running average, closely resembled the trend annual nitrogen load estimates provided by the Tampa Bay Estuary Program (Zarbock et al. 1996; Pribble et al. 2001; Poe et al. 2005; Janicki Environmental 2008).

The relative influence of top-down versus bottom-up processes was assessed for 15 species/groups and results are presented in Figure 2. The effects of fishing and nutrient loading were variable, although similar species did generally show similar responses. In most cases the scenario which explained the largest amount of variation in the data included both fishing and nutrient loading forcing factors (fishing + primary production scenario). A negative value indicates that the model fit worsened under that forcing scenario. Lower trophic level species

such as anchovies, silversides, clupeids, and small carangids were controlled more by bottom up processes. Inclusion of the nutrient loading forcing function explained approximately 10-50% more of the variation than the baseline scenario for these species. Higher trophic level or targeted species like spotted seatrout, red drum, black drum, pompano, and sharks appeared to be influenced more by fishing. When fishing mortality was included, the SS was reduced by 50-75% from the baseline measure for these species. Other species, such as blue crab, sheepshead, and mullet are influenced strongly by both bottom-up and top-down factors. For these species, the 'fishing + primary production' scenario explained 10-40% more of the variation than the next best scenario showing that biomass predictions improved greatly only when both forcing factors were included.

Annual time series of observed and predicted biomass trends for the same 15 species/groups are given in Figure 3. The first panel illustrates model predictions when only fishing mortality and relative fishing effort are presumed to be the major driver of population change. This is true for species such as red drum, pompano, seatrout, and snook that have historically been under considerable fishing pressure. While blue crab and mullet have also been heavily fished in the past, their observed biomass trend is not captured well in this scenario. When the fishing only scenario is compared with the nutrient loading only scenario (panel B), several important patterns emerge. The model suggests that nutrient loading was not responsible for low abundances of red drum, spotted seatrout, and snook during the early 1980s. However, when primary production is forced the model attempts to capture the more recent variation in red drum biomass. Mullet and blue crab predictions are clearly improved when bottom-up forcing is turned on. Furthermore, the model fits for the small pelagic baitfishes (anchovies, silversides, clupeids) are greatly improved under the bottom-up scenario as more of the high frequency variability is captured.

DISCUSSION

The Ecopath with Ecosim model was effective at assessing the relative influence of bottom-up versus top-down control in Tampa Bay. Our model suggests that Tampa Bay is largely controlled by nutrient-driven primary production. This is consistent with the findings of Frank et al (2007) that suggest systems with high species richness and warmer temperatures are usually bottom-up driven and more resilient to exploitation than colder regions. Because nutrient loading in Tampa Bay is now dominated by non-point source runoff due to rainfall, the abundance of many species is closely linked to changes in climate patterns and meteorological events. For example, it is intuitive that species such as anchovies, silversides, and clupeids respond positively to an increase in nutrients and our model captures this process well. In estuaries, nutrient uptake is dominated by the resident phytoplankton community, which is usually nitrogen limited, and leads to an increase in zooplankton and particulate organic material that feeds small pelagic baitfishes. Thus, the nutrient-phytoplankton-zooplankton-fish food web is relatively short and well understood. For the last three to four years, the Tampa Bay area has been under drought conditions and this period of low rainfall coincides with recent declines in abundance of small pelagic baitfishes. The Ecosim model strongly suggests that these declines are due to a decrease in primary production.

The impact of nutrient loading on the benthos is less clear. Phytoplankton production that goes ungrazed settles to the bottom where it can stimulate benthic production. Or, when the water becomes stratified and heats up, phytoplankton sedimentation and subsequent decomposition can lead to hypoxic conditions. Such internal feedback loops can further stimulate or sustain algal blooms, cause relatively widespread and persistent low dissolved oxygen conditions, and have a negative impact on the benthos. Confounding this is the loss of seagrass and other submerged aquatic vegetation that serves as refuge habitat and safe feeding grounds for many benthic invertebrates. The model presented here assumes that the benthic community will respond

positively to an increase in nutrients, and this production is transferred up the food web to demersal fishes. It is critical that the model includes the appropriate response of benthic invertebrates to external nutrient loading either through adjustments to the vulnerability flow rate parameter or by including trophic mediation functions. We have explored the use of a seagrass mediation function to simulate declines in predation vulnerability with increases in seagrass biomass for several fish groups. Data from the Environmental Protection Commission of Hillsborough County's (EPCHC) benthic monitoring program (David Karlen, personal communication) have been provided to investigate similar relationships for the benthos. Inclusion of these data as reference time series of abundance is expected to improve the reliability of the model and could potentially provide different results.

Shifts from bottom-up to top-down control have been observed on the Scotian Shelf and Gulf of Maine (Frank et al. 2007). Although Tampa Bay as a whole appears to be strongly bottom-up driven over the entire time period modeled, there is evidence that some species may have changed from being top-down to bottom-up controlled. A closer look at the biomass time series plots for red drum show that including fishing mortality alone allowed the model to estimate the biomass rather well from 1950-1984. In 1987 rules went into effect that eventually led to the prohibition of the commercial sale of red drum, as a result fishing mortalities declined and the stock rebounded. After 1986, the fishing only scenario begins to diverge from the higher frequency variation in the observed data while scenarios that included environmental forcing attempt to capture that variation. One interpretation of this is that the top-down control was reduced following strict regulations during the 1980s and the population is now experiencing a stronger bottom-up influence. Further investigations into possible shifts in direction of trophic control in Tampa Bay are forthcoming.

While the ultimate goal is to explain the historical trends in abundance for all species and then use the model to make predictions about policy issues, much is learned through the actual fitting process. Assuming the underlying Ecopath model is correctly parameterized, poor fits in the Ecosim model can be due to several factors including, but not limited to: incorrect settings of foraging arena parameters (especially the vulnerability flow rates), the omission of important ecological processes, and inappropriate or missing time series data. In this analysis, observed trends in abundance for assessed species were taken from the assessments themselves and represent trends for the entire Gulf of Mexico, west coast of Florida, or southwest region of Florida. Although biomass trends from broader geographic areas usually go back further in time and can give a good historical perspective on biomass levels, they most likely fail to capture the high frequency variability specific to Tampa Bay populations. Though not shown here, in some cases the model fit to observed indices of abundance from FIM data was better than that from the broader area assessments. Because it is difficult to measure the vulnerability flow rates in the field, longer time-series of both abundance and diet are needed to improve our estimates of this parameter. Tampa Bay is a data-rich system and many of the necessary monitoring programs are already in place (FIM, EPCHC, various habitat and water quality monitoring programs), however time-series data are lacking for the critically important zooplankton component. Initial investigations into likely mediation processes, such as seagrass as refuge habitat and water quality impacts on visual predators, have only been qualitative and linear but do appear to improve the fit of the model. By determining the nonlinear relationship between a predator, some mediating third party organism, and their prey we hope to incorporate all major trophic processes into the model.

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Table 7. Biomass, production, and consumption parameters used as model input and resulting trophic level and mortality estimates made by the Ecopath model. * indicates biomass values estimated by Ecopath.

Group name	Biomass (t/km ²)	Prod./ biom. (/year)	Cons./ biom. (/year)	Trophic Level	Fishing Mortality rate (F)	Predation Mortality rate (M2)	Other Mortality rate (M0)
Seabirds	0.0372	4.2	29.7383	3.97	0	0	4.2
Dolphins	0.0947	0.12	14.6	3.83	0	0.0182	0.1018
large coastal sharks	1.7059	0.3293	3.4798	4.26	0.0439	0.0014	0.284
small coastal sharks	0.7319	0.5392	4.6517	3.75	0.0367	0.1345	0.368
0-6 Snook	0.0622	6	29.3185	3.74	0	5.0731	0.9269
6-18 Snook	0.4499	2	10.2844	3.83	0.0026	1.1031	0.8942
18-66 Snook	1.0704	1	5.3199	3.91	0.0218	0.2708	0.7074
66+ Snook	0.0912	0.6	3.85	4.01	0.4396	0.0074	0.153
0-6 Red Drum	0.0356	5	21.2105	3.17	0.0446	4.2847	0.6707
6-18 Red Drum	0.3506	1.75	7.584	3.51	0.063	0.9629	0.7241
18+ Red Drum	1.2172	0.87	3.8344	3.9	0.1952	0.0038	0.671
juv Black Drum	0.0015	5.1	29.1567	3.12	0.0007	4.4541	0.6452
adult Black Drum	0.2401	0.65	4	3.42	0.4554	0.0275	0.1671
juv Spanish mackerel	0.0122	5	34.1559	3.84	0.3017	3.8045	0.8938
adult Spanish mackerel	1.0018	0.96	7.2273	4.03	0.4815	0.2712	0.2072
juv Jacks	0.0161	4	27.925	3.8	0.0838	3.1714	0.7447
adult Jacks	0.7846	0.71	5.4618	3.95	0.1906	0.0683	0.4512
juv Pompano	0.00435	4	29.5374	3.12	0.0459	3.4465	0.5076
adult Pompano	0.2107	0.82	7.0036	3.34	0.1804	0.4592	0.1804
0-6 Sea Trout	0.0251	5.5	28.9227	3.64	0.016	4.7576	0.7265
6-18 Sea Trout	0.2548	1.75	9.736	3.8	0.0286	1.221	0.5004
18+ Sea Trout	1.5028	0.75	4.2762	4	0.25	0.3056	0.1944
juv Sand Trout	0.0385	3	19.254	3.57	0.0243	2.5867	0.389
adult Sand Trout	0.5183	0.85	5.7958	3.85	0.0796	0.1673	0.6031
juv snappers	0.0184	3.5	21.8011	3.67	0.0007	2.8742	0.6251
adult snappers	0.4895	0.73	4.7579	3.95	0.1598	0.1511	0.4191
juv Flounders	0.00153	5	30.6905	3.71	0.0065	4.1324	0.8611
adult Flounders	0.2232	0.77	5.739	3.94	0.4283	0.179	0.1627
juv Mullet	0.1247	5	33.1952	2.19	0	4.2412	0.7588
adult Mullet	4.7904	0.75	7.7787	2.13	0.2357	0.3196	0.1947
juv Ladyfish	0.0378	4	26.9109	3.65	0	3.4607	0.5393
adult Ladyfish	0.7986	0.9	6.6386	3.82	0.2575	0.4074	0.2351
juv Silver Perch	0.0413	5	21.8803	3.46	0.0015	4.3273	0.6712
adult Silver Perch	0.8523	1.32	6.944	3.78	0.0042	0.5082	0.8076
juv Sheepshead	0.0379	5	33.912	3.15	0.0356	4.1756	0.7888
adult Sheepshead	1.8298	0.93	6.5046	3.45	0.1834	0.0446	0.702
Spadefish	0.2703	0.97	7.7265	2.49	0.0671	0.0535	0.8494

Group name	Biomass (t/km ²)	Prod./ biom. (/year)	Cons./ biom. (/year)	Trophic Level	Fishing Mortality rate (F)	Predation Mortality rate (M2)	Other Mortality rate (M0)
Juv Groupers	0.1085	1.74	6.5988	3.75	0.006	1.2702	0.4638
Kingfish (sciaenid)	0.9911	1.16	6.9203	3.11	0.3058	0.317	0.5372
Spot	1.0448	1.47	8.7304	3.27	0.0104	0.9616	0.498
Pinfish	3.1867	2.376	10.3455	2.62	0.0588	2.2159	0.1013
Mojarra	*2.1548	1.32	9.2894	3.36	0.0218	1.2322	0.066
Catfish	2.8253	0.78	5.0114	3.32	0.0655	0.0582	0.6563
Rays	3.6564	0.53	6.0197	3.19	0.0166	0.1148	0.3986
other small demersals	3.7866	3.18	9.92	3.32	0.0835	1.3285	1.768
Needlefish	1.0598	1.97	7.7657	3.63	0	0.1036	1.8664
small carangids	1.6312	2.08	9.0666	3.12	0.0202	1.0444	1.0154
Menhaden	*1.3519	2.9583	10.0965	2.58	0.0666	2.7438	0.1479
Clupeids	*4.9925	3.2541	10.4159	2.89	0.0766	3.0148	0.1627
Silversides	*0.9381	3.61	15.6948	2.97	0.1058	3.3237	0.1805
Anchovies	*2.9428	3.73	15.9066	2.78	0.0391	3.5044	0.1865
killifish and livebeares	*1.6945	3.35	16.2628	3.17	0.0001	3.1824	0.1675
Gobies	*0.4679	3.88	19.7745	3.13	0	3.686	0.194
juv Blue Crab	0.6989	3.872	15.8382	3.11	0.027	3.4019	0.4431
Blue Crab	1.0672	1.9	8	3.42	0.428	0.9892	0.4828
juv Shrimp	0.4702	7	29.3511	2.52	0	6.3314	0.6686
Shrimp	1.8073	3.5	14.7	2.99	0.0818	3.2779	0.1404
Caridean Shrimp	*2.5601	5.475	18.25	2.46	0	5.4203	0.0547
bivalves	82.1346	3.5805	9.5	2.1	0	1.0043	2.5762
echinoderms	12.723	1.7652	4.6656	2.32	0	0.2678	1.4974
gastropods	17.0829	3.7189	9.3667	2.16	0	1.6014	2.1175
larger benthic crustaceans	15.5973	4.689	17.394	2.53	0	3.68	1.009
small benthic crustaceans	16.1283	9.5254	24.426	2.34	0	9.106	0.4194
predatory worms	12.8723	4.3722	11.13	3.01	0	1.5276	2.8446
Susp./Deposit feed. worms	14.2261	5.8306	15.105	2.1	0	4.228	1.6026
Sessile Filter Feeders	39.6484	1.6207	8.2854	2.09	0	0.805	0.8157
Mesoplankton	5.4846	17.3	60	2.5	0	8.4871	8.8129
Holoplankton	17.2752	17.3333	57.7777	2.05	0	14.7027	2.6306
Bacteria	70.4305	119	280.6356	2	0	4.7649	114.2351
Phytoplankton	51.8447	218.5798	-	1	0	22.0582	196.5216
Sea Grass	239.1822	10.2231	-	1	0	0.3126	9.9105
Macro and Microalgae	88.9377	19.8695	-	1	0	17.1508	2.7187
DOC	146.0922	-	-	1			
Detritus	551.745	-	-	1			

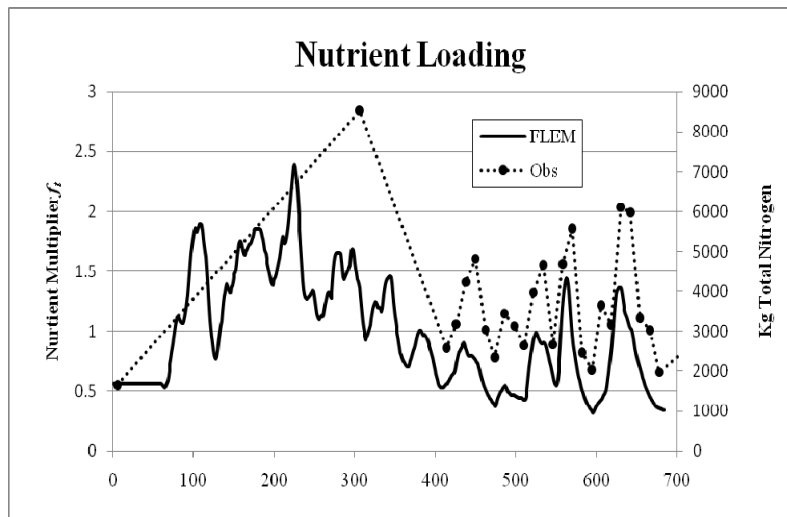


Figure 6. Nutrient loading pattern estimated by the FLEM model (solid line) and provided by the Tampa Bay Estuary Program (dotted line).

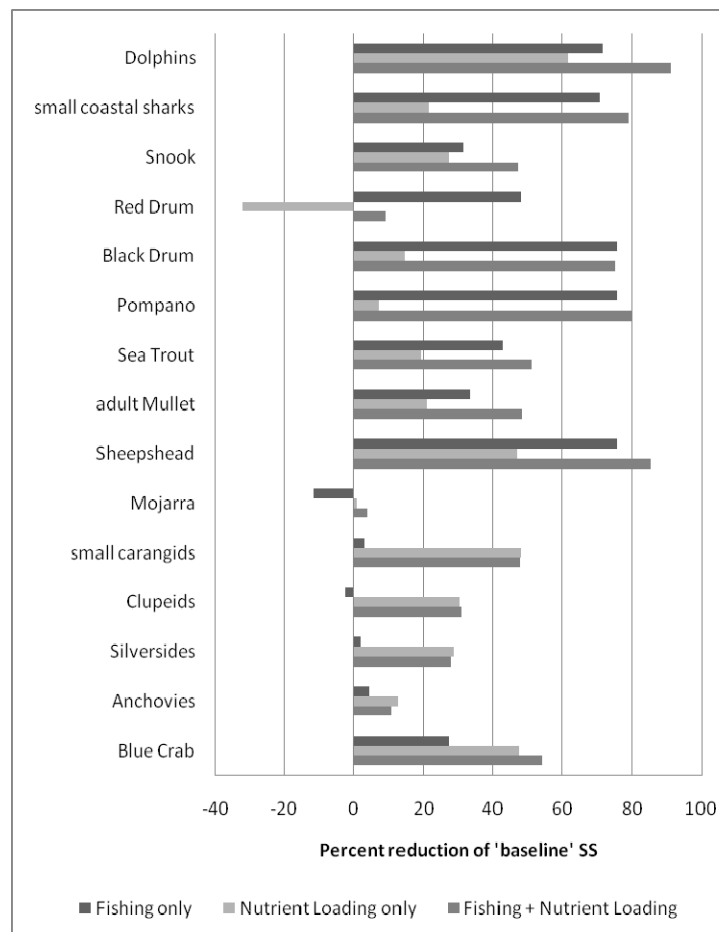
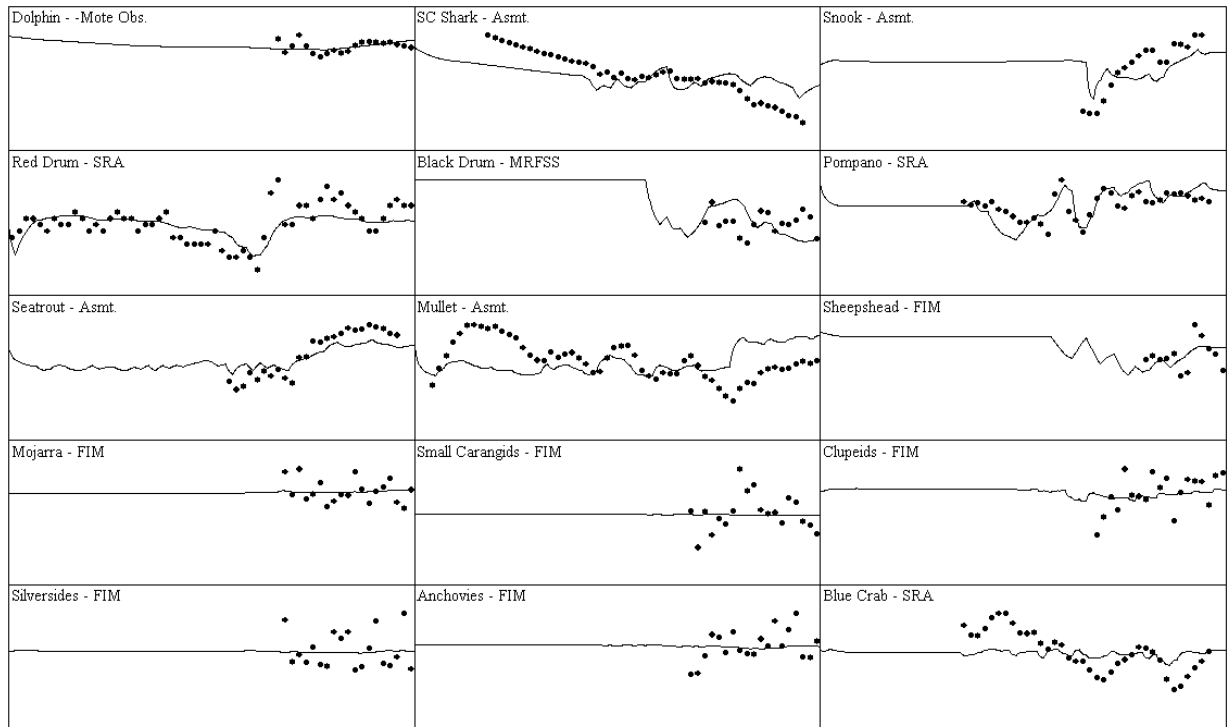
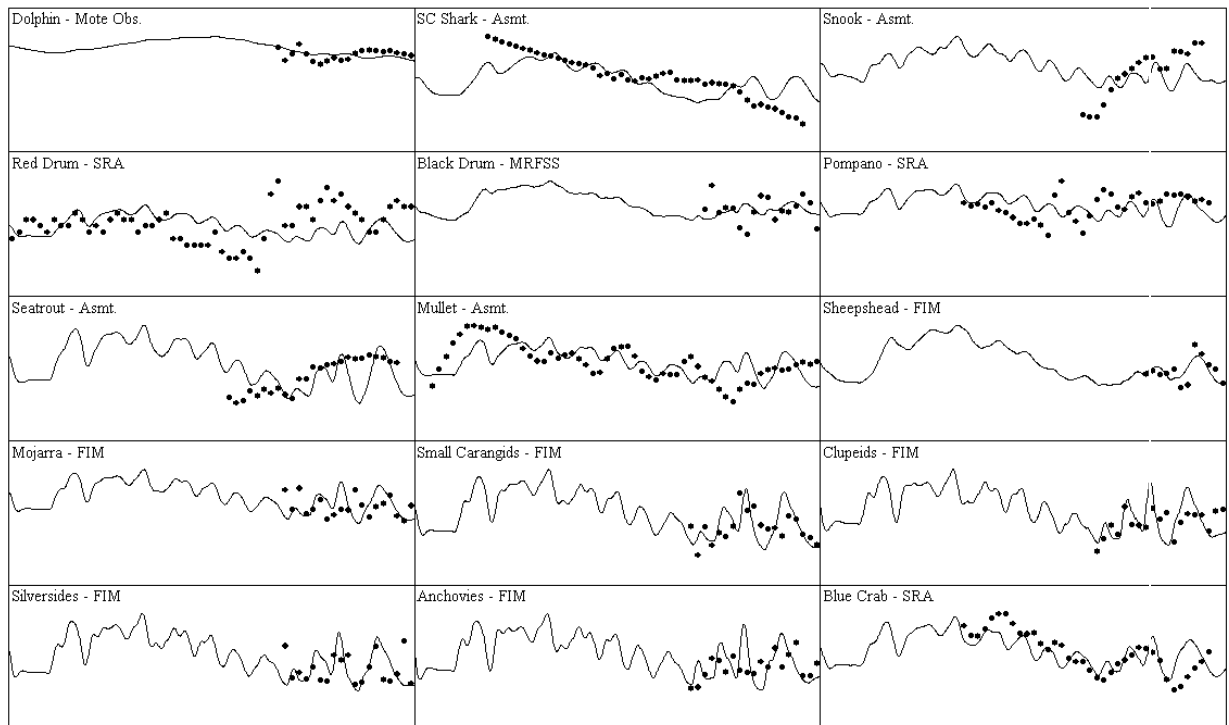


Figure 7. Percent improvement in model fit from the 'baseline' scenario.

A



B



C

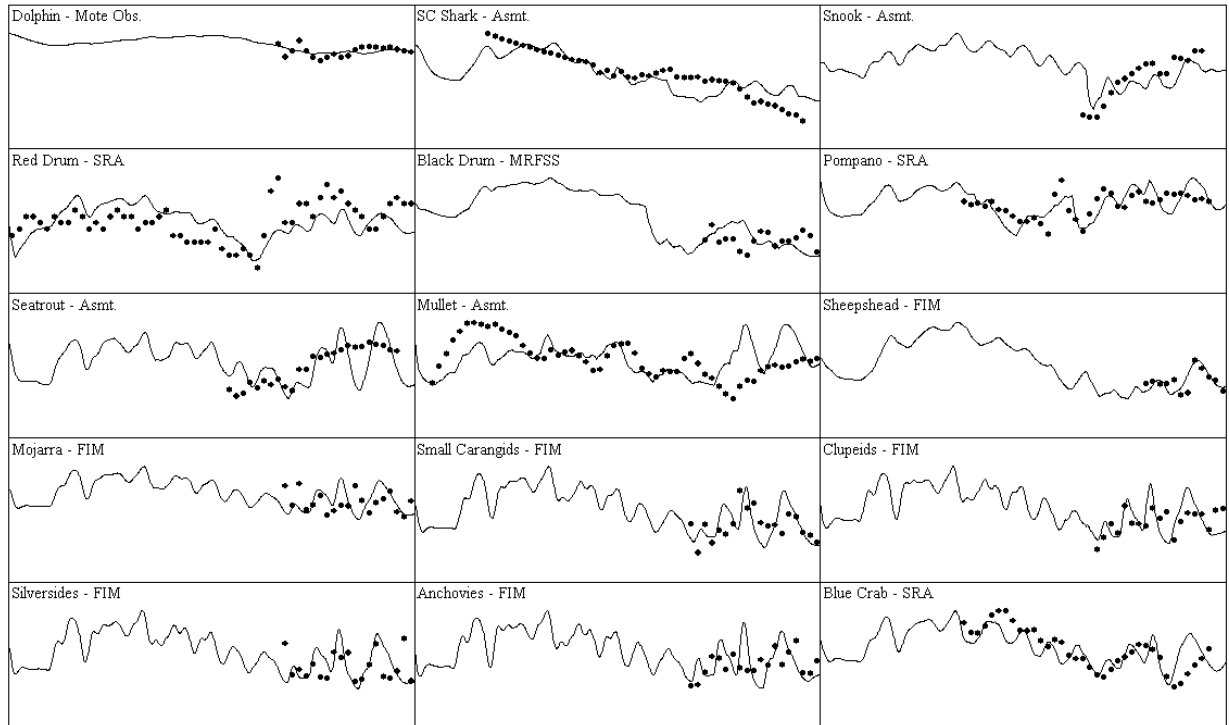


Figure 8. Time series of biomass trends (t km⁻²) estimated by Ecosim (line) and from observed data (dots) for 15 species in three model scenarios: A-fishing forcing factor only, B-nutrient loading forcing factor only, and C-fishing and nutrient loading forcing factors. In each plot is the name of the species/group and the source type of observed data. Asmt. = stock assessment; SRA = stock reduction analysis; MRFSS = recreational catch per unit effort; FIM = fisheries independent monitoring catch per unit effort.

NEKTON COMMUNITIES ASSOCIATED WITH SEAGRASS IN TAMPA BAY: THE EFFECTS OF SEAGRASS BED ARCHITECTURE, SEAGRASS SPECIES COMPOSITION, AND VARYING DEGREES OF FRESHWATER INFLUENCE

K.E. Flaherty, R.E. Matheson Jr., F.X. Courtney & R.F. Jones

ABSTRACT

Seagrass beds provide refuge and feeding areas for various species of nekton. Several studies have indicated that seagrass bed architecture, seagrass species composition, location, and water quality can affect the use of seagrass habitats by nekton. To document patterns of spatiotemporal distribution and abundance of nekton in Tampa Bay seagrass beds and to evaluate any changes in the nekton community associated with variations in the vegetation community and the influence of freshwater, we analyzed long-term fisheries-independent data (1989–2007) and conducted a short-term synoptic study of small-bodied seagrass-associated nekton in Tampa Bay. Long-term community structure was strongly seasonal, partly due to recruitment patterns of abundant species such as *Lagodon rhomboides* and *Orthopristis chrysoptera*. Short-term synoptic data showed a clear difference in community structure associated with historical salinity variation, which was used as a proxy for freshwater influence. The aboveground biomass of seagrass beds sampled during the synoptic study was less in areas influenced by freshwater than in those not influenced by freshwater. Despite this reduced seagrass cover some species, such as *Cynoscion nebulosus* and *Farfantepenaeus duorarum*, preferred seagrass beds in areas with greater freshwater influence. The value of seagrass as habitat transcends the mere fact that it is present or absent, and some nekton may preferentially inhabit seagrass beds with a particular seagrass species composition or salinity regime. Therefore, management decisions regarding seagrass beds, especially decisions involving mitigation by acre-for-acre substitution of one seagrass bed for another, need to be made with full consideration of the complex relationships governing the value of a particular seagrass bed for nekton.

INTRODUCTION

Seagrass beds are known to provide refuge and feeding areas for various species of nekton, and several studies have indicated that bed architecture, seagrass species composition, location, and water quality can affect the value of seagrass as habitat (Bell et al. 1987, 1988; Robbins and Bell 1994). Bell et al. (2001) found little evidence that seagrass patch size has any consistent effect on faunal communities in Tampa Bay and suggested that other landscape-level factors (e.g., location within the estuary, proximity to other habitats) may be more important in determining faunal distribution and abundance. Rydene and Matheson (2003) documented significant relationships between the density of seagrass cover and the abundance of nine species of fish in Tampa Bay seagrass beds.

Tampa Bay and its tidal rivers are important nursery habitat for many species of fish, including several that support important recreational fisheries (Peters and McMichael 1987; McMichael and Peters 1989; Peters et al. 1998; Matheson et al. 2003; Rydene and Matheson 2003). Freshwater inflow from rivers can have a broad influence on estuarine systems by affecting factors such as transport of larval and juvenile fishes, nekton community structure, salinity regimes, and nutrient dynamics (Norcross and Shaw 1984; Livingston 1997; Livingston et al. 1997; Tsou and Matheson 2002; Greenwood et al. 2007). Water quality, as estimated by salinity and turbidity data, may affect seagrass-associated fish communities directly, through species' tolerances, or indirectly, through effects on seagrass habitat (Blaber and Blaber, 1980; Tomasko et al. 2005).

In many estuaries, including Tampa Bay, management decisions often affect the fate of seagrass beds, and these decisions are often based on limited information. Available data may be

applicable only at the baywide level and may not be specific enough to characterize a specific seagrass bed as habitat for a given nekton species. For some nekton species, seagrass beds in a particular portion of the bay or of a particular architecture may be much more important than beds with other characteristics. Nekton preferences for seagrass beds of a particular species of seagrass are also unknown. Therefore, for a given nekton species, mitigation for the loss of a seagrass bed in one region of the bay by creation of a new bed of the same size in another region may not be effective, because the new bed would not necessarily provide habitat for the nekton species that had inhabited the bed that was lost.

Determining the relative value of different types of seagrass beds as faunal habitat is critical in prioritizing the seagrass beds to be preserved and restored within the Tampa Bay system, and the purpose of this study is to use existing long-term nekton data and an intensive synoptic survey to determine these habitat values. The objectives of this study were 1) to document patterns of spatiotemporal distribution and abundance of nekton in seagrass beds in different parts of Tampa Bay based on analyses of 18 years of fisheries-independent monitoring data; 2) to investigate whether these patterns vary as a function of freshwater inflow to Tampa Bay; and 3) to provide data not found in the historical data set via additional synoptic sampling that would allow detailed characterization of seagrass bed architecture (e.g., blade density, percent cover) and a large-scale evaluation of the influence of low-salinity tributaries and of turbidity on the distribution and abundance of seagrass-associated nekton.

METHODS

Small-bodied nekton were collected in the Tampa Bay estuary, on the west coast of Florida, from 1989 through 2007 as part of a statewide monitoring program (McMichael 2007). For the purposes of regional analyses, Tampa Bay was divided into four regions (Hillsborough Bay, Old Tampa Bay, Middle Bay, and Lower Bay; Figure 1), based on salinity and habitat types, after Lewis and Estevez (1988). Sampling sites were selected using a grid that divided the estuary into cells 1 nm on a side; these were further divided into a microgrid of cells 0.1 nm on a side. Microgrid cells were selected for sampling in a stratified-random manner throughout the study, but other details of methodology changed from 1989 to 2007 (detailed description of site selection can be found in the procedural manual of the Florida Fish and Wildlife Conservation Commission's Fisheries Independent Monitoring Program, hereafter referred to as the FIM Program). We do not believe, however, that any of the programmatic changes significantly affected our analyses. For this paper, we analyzed data collected during long-term seasonal sampling throughout this 18-year time period (spring: March–June; fall: September–December) and during short-term synoptic sampling in September 2007.

All nekton were collected with a 21.3-m, 3.2-mm-mesh bag seine. Seines were set and retrieved offshore, and a 15.5-m line between the seine poles was used to maintain the net opening at a consistent width. Seines were pulled over a distance of 9.1 m and sampled an area of approximately 140 m² of bottom. Location, salinity (ppt), water temperature (°C), dissolved oxygen (mg l⁻¹), water depth (m), and habitat descriptions were recorded for all sampling sites. Fish and selected invertebrates (blue crabs, stone crabs, horseshoe crabs, and penaeid shrimp) were identified to the lowest practical taxonomic level and counted, and a subsample of randomly selected individuals were measured. Catch per unit effort was calculated as number of individuals 100 m⁻².

Seasonal sampling (1989–2007)—To focus the study on long-term seasonal nekton samples that were collected in seagrass beds, a subset of the data was used in community structure analyses: only seine sites that had at least 50% seagrass or macroalgal coverage were included. Seagrass species encountered during sampling included *Ruppia maritima*, *Halodule wrightii*, *Syringodium*

filiforme, and *Thalassia testudinum* and are referred to by their genus names for the rest of the paper. Macroalgal species such as *Caulerpa* spp. and *Halophila* spp. were also encountered. Sites at which a single seagrass or macroalgal species accounted for at least 50% of the bottom vegetation were considered to be dominated by that species. Sites at which the primary seagrass or macroalgal species accounted for less than 50% of the bottom vegetation were considered to be mixed. ArcGIS software was used to display the type of seagrass species encountered at each sampling site spatially.

Levels of annual freshwater input into Tampa Bay from 1989 through 2007 were estimated using discharge data collected by water-stage recorders at two locations: the Alafia River at Lithia and the Little Manatee River at Wimauma (USGS 2008). Years were categorized into those with above average, average, or below average river inflows by using 95% confidence intervals of mean inflow for the study period.

Synoptic sampling (September 2007)—Short-term sampling conducted specifically for this study was designed to refine the distributions and habitat relationships documented by analyses of long-term seasonal data. We conducted one synoptic survey in Tampa Bay seagrass beds during September 2007. This survey consisted of 54 collections made using standard FIM Program collecting gear and methodology (21.3-m offshore seine, as described above). Samples were equally divided among seagrass beds near sources of freshwater inflow (creeks or rivers) and those distant from sources of freshwater inflow. Sampling sites were selected from 1-nm² grid cells with the greatest or least freshwater influence in the bay based on historical surface salinity means, standard deviations, and ranges (the 30 cells with greatest and the 30 cells with least freshwater influence). These cells of greatest and least freshwater influence were displayed spatially with ArcGIS to assist in field sampling, and sampling sites were chosen within these cells that could be sampled by 21.3-m seines and were characterized with continuous, dense cover (i.e., at least 90% cover) of any seagrass species. Seagrass composition at each site was determined in the same manner as for seasonal sampling and was displayed spatially with ArcGIS. Synoptic sampling was conducted within one week to minimize temporal effects in favor of spatial and microhabitat effects.

At each synoptic sampling site, we further characterized seagrass by collecting three 15.2-cm seagrass cores. The area sampled by each 21.3-m seine was marked and within that sampling area three randomly placed 15.2-cm-diameter seagrass cores were taken to a depth of 30 cm. All plant material was washed from the core tube into mesh bags and rinsed of sediment on-site. Seagrass core samples were stored in a cold room and later frozen. For processing, samples were defrosted and the seagrass sorted by species. If possible, we counted the blades of at least 20 intact shoots (i.e., those comprising blades, short shoots, roots, and rhizomes) of each seagrass species present, and blade length and width (for *Thalassia*) were recorded. For each species, shoots and attached blades were counted. Unattached blades were also counted. Shoot density (shoots m⁻²) and blade density (attached blades m⁻²) were then calculated. Blade density was a conservative estimate because unattached blades were not included in the calculation. The average length (cm) of intact (for *Thalassia*, having the rounded tip present) attached blades was calculated for each site as a conservative measure of canopy height. Canopy cover (individual blade areas x blade density [cm² m⁻²]) was calculated by using blade length and width measurements. Very few measurements of blade width were taken for *Syringodium* and *Halodule*, so blade width was estimated at 1.0 mm.

The seagrass material was then sorted by species and fraction, and dried at 60° C for five days. The dried seagrass fractions were weighed with a BrainWeigh B1600D top-loading electronic balance, and dry weights were converted to grams m⁻² as shoot bases, attached blades, unattached

blades, and root/rhizome. Aboveground biomass was calculated as grams dry weight m⁻² of attached blades.

In addition to standard water-quality measurements made during seasonal sampling (temperature, salinity, dissolved oxygen), turbidity was also measured at each synoptic sampling station. A 300-mL water sample was taken and stored on ice, and turbidity was later measured in the lab with a Hach Model 2100P Portable Turbidimeter. The water sample was inverted twice, and then read in 15-second increments over 60 seconds. The mean of these readings was used as the turbidity value for each site.

Nekton community structure analyses—Differences in nekton community structure as observed during seasonal sampling from 1989 through 2007 and synoptic sampling in September 2007 were analyzed using PRIMER V.6 software (Plymouth Routines in Multivariate Ecological Research; Field et al. 1982; Clarke 1993; Clarke and Warwick 1998) to ascertain whether any changes in nekton communities were attributable to seagrass composition and/or freshwater influence. Catch per unit effort (CPUE; individuals 100 m⁻²) data were fourth-root transformed to reduce the influence of extremely abundant taxa on analyses. Seasonal sampling data (1989–2007) were averaged over season, region, seagrass composition, and river inflow (above average, average, or below average); each data point from synoptic sampling represents one net set in September 2007. Comparisons of nekton assemblages were conducted using nonmetric multidimensional scaling (MDS) and analyses of similarity (ANOSIM). The usefulness of spatial ordination of sample data using MDS is indicated by a stress value. Generally, stress values less than or equal to 0.2 are considered useful in determining a 2-dimensional representation of sample data, and the lower the stress value, the better the representation. Differences in community structure by season, seagrass composition, river inflow, and region (for seasonal data) and by region, seagrass composition, and freshwater influence (for synoptic data) were tested at $\alpha = 0.05$ using the ANOSIM procedure, and a nonparametric permutation test applied to the Bray-Curtis similarity matrix (Bray and Curtis 1957). The value of the test statistic, R, was calculated based on ranked similarities within and among groups (ranging from 0 to 1) under the null hypothesis that community structure did not differ with respect to these variables. ANOSIM results were interpreted by directly examining R-values to determine whether groups were well separated ($R > 0.75$), clearly separated with some overlap ($0.75 \geq R > 0.5$), moderately separated ($0.5 \geq R > 0.25$), or barely separated ($R \leq 0.25$). Groupings between samples were also determined based on a 50% similarity level calculated using CLUSTER analysis. Environmental matching (BEST routine) was used to determine the environmental factors (temperature, salinity, dissolved oxygen, bottom vegetation cover, and, for synoptic data only, aboveground biomass of seagrasses) that best explained differences in community structure. Nekton taxa contributing to differences in community structure between beds composed of the two dominant seagrass communities (*Halodule*-, *Thalassia*-dominated) and, for synoptic data only, between the two levels of freshwater influence were determined using the similarity percentages routine (SIMPER).

Water quality and seagrass core parameters collected at synoptic-sampling sites were compared by freshwater-influence category using Student's t-tests. Standard errors were pooled when variances between samples were equivalent, and a Satterthwaite approximation of standard error was used when variances were not equal.

RESULTS

Annual patterns of river inflow varied significantly over the 18-year study period (Figure 2). Mean daily river inflow of the Alafia and Little Manatee rivers in 1994, 1995, 1997, 2003, and 2004 was extremely high. The high 2004 inflows were related to heavy rainfall in the summer

due to hurricanes, whereas the 1997 inflows were associated with El Niño weather patterns in autumn and winter (Schmidt et al. 2001). In 1989, 1990, 1993, 1996, 1999, 2000, and 2007 inflows were very low, at or below the 95% confidence interval of study-period mean inflows; some of these low inflows were attributable to La Niña conditions. In the remaining years, inflows were closer to historic means. Only June through December were included to correlate to the fall season (data used in most community structure analyses; see the following results).

In the synoptic data, some of the physical factors varied based on freshwater influence (Figure 3). Temperature was significantly higher and turbidity was significantly greater at sites with freshwater influence (t-tests; [equal variance; pooled] $p = 0.0294$ and [unequal variance; Satterthwaite] $p = 0.0062$, respectively). Salinity was significantly greater at sites without freshwater influence ([equal variance; pooled] $p < 0.0001$). Dissolved oxygen did not differ significantly by freshwater-influence category ([equal variance; pooled] $p = 0.6870$).

In the long-term data, 1,405 seasonal (1989–2007) observations fit the criteria for analysis. Sampling sites that contained *Halodule*-dominated seagrass beds were the most common (47.5%), followed by *Thalassia*-dominated beds (29.4%). *Syringodium* and mixed seagrass beds were less common. Sites with seagrass beds dominated by *Ruppia* and other seagrasses or macroalgae were relatively rare and were omitted from further analysis (Table 1, Figure 1). Seagrass communities at synoptic sites (54 total) mirrored these trends, but no seagrass beds dominated by *Ruppia* or other seagrasses or macroalgae were sampled (Table 1, Figure 4). In both datasets, seagrass species composition differed between the four bay regions (Figures 1, 4, and 5). The overwhelming majority of samples collected in the upper regions of the bay (Hillsborough Bay and Old Tampa Bay) were dominated by *Halodule*. Middle Bay exhibits a gradual change from *Halodule*-dominated beds in the upper bay to *Thalassia*-dominated beds closer to the Lower Bay. In the Lower Bay, more samples were collected in seagrass beds dominated by *Thalassia* than in other regions (Figure 5). Seagrass community metrics (canopy height, shoot density, aboveground biomass, and canopy cover) as determined from core samples differed by dominant species and by whether the site was influenced by fresh water (Figure 6). In almost all seagrass bed types, those sites that were not influenced by fresh water were the most productive. Small sample size precluded statistical comparisons regarding freshwater influence in mixed and *Syringodium*-dominated seagrass beds, but *Halodule*- and *Thalassia*-dominated beds at sites with low levels of freshwater influence had significantly higher shoot density (t-tests [unequal variance; Satterthwaite] for all; $p = 0.0226$ and $p = 0.0401$, respectively), aboveground biomass ($p = 0.0074$ and $p = 0.0078$, respectively), and canopy cover ($p = 0.0287$ and $p = 0.0111$, respectively); canopy height was not significantly different in either seagrass bed type.

During seasonal sampling (spring and fall, 1989–2007), 120 nekton taxa were collected; of these, 49 were collected during synoptic sampling in 2007 (Table 2). One rare species, *Monacanthus ciliatus*, was collected only during synoptic sampling. The catches were numerically dominated in seasonal sampling by *Anchoa mitchilli*, *Lagodon rhomboides*, *Bairdiella chrysoura*, and *Orthopristis chrysoptera* and in synoptic sampling by *A. mitchilli*, *Lucania parva*, *Farfantepenaeus duorarum*, and *Harengula jaguana*. Differences in CPUE between the two datasets were generally $< 5\%$ for most species, but there were exceptions. In the two studies, differences in the CPUE of some species can be explained by seasonality (e.g., spring recruitment for *L. rhomboides* and *O. chrysoptera*) or by the extremely patchy distributions typical of some schooling species (e.g., *A. mitchilli*, *H. jaguana*, *Menidia* spp., and *Sardinella aurita*).

Long-term data showed a clear separation of community structure by season (Figure 7; ANOSIM (season): $R = 0.552$, $p = 0.001$). Almost 20% of the differences in community structure were due to *L. rhomboides* and *O. chrysoptera*, which recruit in the spring. For this reason and to compare

these data directly with the synoptic data, only samples collected in the fall were used in the remaining long-term data analyses. In the fall samples, region (ANOSIM: $R = 0.371$, $p = 0.001$) and seagrass composition (ANOSIM: $R = 0.214$, $p = 0.005$) strongly influenced community structure, but level of annual river inflow did not (ANOSIM: $R = -0.008$, $p = 0.586$). The MDS plot of fall samples shows a gradient of change in nekton assemblages collected from *Halodule*-dominated seagrass beds found in the upper bay to *Thalassia*- and *Syringodium*-dominated beds found in the Lower Bay (Figure 8). When mixed beds were removed from the analysis, seagrass composition had an even stronger influence on community structure (ANOSIM: $R = 0.328$, $p = 0.001$). Taken together, the environmental factors measured (temperature, salinity, dissolved oxygen, bottom vegetation cover) had the highest correlation with community structure (BEST = 0.524).

Freshwater influence (Figure 9; ANOSIM: $R = 0.431$, $p = 0.001$), region (ANOSIM: $R = 0.319$, $p = 0.001$), and seagrass composition (ANOSIM: $R = 0.323$, $p = 0.001$) were strongly associated with nekton community structure in the synoptic data set. The MDS plot again showed a gradient of change in nekton assemblages from *Halodule*-dominated seagrass beds in the upper bay to *Thalassia*- and *Syringodium*-dominated beds in the lower bay (Figure 9). Salinity and aboveground seagrass biomass had the highest correlation with community structure (BEST = 0.524).

Based on SIMPER analyses, 8 of the top 10 species influencing differences in community structure between *Halodule*- and *Thalassia*-dominated beds were the same for seasonal and synoptic data (*A. mitchilli*, *B. chrysoura*, *Cynoscion nebulosus*, *F. duorarum*, *H. jaguana*, *L. rhomboides*, *L. parva*, and *Menidia* spp.; Table 3). Most of these species exhibited the same trends in abundance versus seagrass habitat in both datasets. For example, *A. mitchilli*, *F. duorarum*, and *C. nebulosus* were consistently more abundant in *Halodule*-dominated seagrass beds, while *L. rhomboides* was more often found in *Thalassia*-dominated beds. The strong differences in community structure between freshwater-influence categories seen in the synoptic data were also due to differences in abundance of some of the same species (Table 4). For example, *A. mitchilli*, *F. duorarum*, and *C. nebulosus* were all more abundant in areas influenced by freshwater, while *L. rhomboides* was more abundant in areas without significant freshwater influence.

DISCUSSION

Based on 18 years of historical data and a synoptic sampling event in September 2007, the geographic and proportional extent of types of seagrass beds remains relatively stable within Tampa Bay. *Halodule*-dominated seagrass beds were the most common, followed by *Thalassia*-dominated beds and beds of mixed species of seagrass. *Halodule*-dominated beds also occurred more often in areas of freshwater influence, while *Thalassia*-dominated beds occurred more often in areas without freshwater influence. Estimates of shoot density and aboveground biomass (blade dry weight) were comparable to those in other studies conducted in Tampa Bay (Hall et al. 1991; Mangrove Systems Inc., unpubl. data; Tomasko and Dawes 1990), and canopy cover and canopy height estimates in *Thalassia*-dominated beds were comparable to those of Florida Bay beds (Matheson et al. 1999). *Halodule*-dominated beds were found throughout Tampa Bay, while *Syringodium*- and *Thalassia*-dominated beds were found more often in the Middle and Lower Bay regions.

Season, region of the bay, seagrass composition, and freshwater influence were associated with the distribution and abundance of nekton. Seasonality has a large impact on the community structure of estuarine nekton (Potter et al. 1986; Lazzari et al. 1999; Tsou and Matheson 2002),

and the long-term data showed a strong seasonal influence in community structure partly caused by recruitment patterns of *Lagodon rhomboides* and *Orthopristis chrysoptera*.

Variability in habitat characteristics (Blaber and Blaber 1980; Bell et al. 1988; Raposa and Oviatt 2000; Paperno et al. 2001) and variation in abiotic factors between regions of an estuary can also contribute to differences in nekton community structure. The covariation between regions within the bay and species-composition of seagrass beds complicates the interpretation of regional differences in abundance of seagrass-associated fishes (i.e., separating the importance of seagrass species from that of region). *Cynoscion nebulosus* was most abundant in *Halodule*-dominated seagrass beds in both the long-term and synoptic data and was also more abundant in Old Tampa Bay, which has a higher proportion of *Halodule*-dominated beds. These trends were static over time in that the majority of nekton species contributing to differences between *Halodule*- and *Thalassia*-dominated beds showed the same trends in both long-term and synoptic data.

Analyses of synoptic data revealed a clear difference in community structure based on the amount of freshwater influence, but years of high, moderate, and low river inflow did not separate communities in long-term data. Considering that we found more differences in community structure based on historical salinity regimes (i.e., in the process of choosing freshwater-influenced sites for the synoptic study, we used historical surface-salinity data from the entire long-term sampling period) than on yearly levels of river inflow, long-term freshwater influence appears to be more important in influencing abundance than recent conditions are.

We did not find significant associations between community structure and turbidity, even though sites with freshwater influence were significantly more turbid. Tomasko et al. (2005), in a long-term study that began in the 1980s, found that increased water clarity, and therefore higher photosynthetic potential, led to increased seagrass coverage in Tampa Bay. We can only speculate that associations may have been found if we had turbidity measurements for the long-term data. Other studies have found that turbidity is an important factor in estuarine fish distribution because of the reduced predation pressure and increase in the number of filter-feeding fishes in turbid waters (Moore 1973; Swenson 1978; Blaber and Blaber 1980).

The supplemental data collected during synoptic sampling were a valuable addition to the data associated with each collection. Based on seagrass parameters measured during our synoptic study, beds influenced by fresh water may provide less cover than those that are less strongly influenced by fresh water. This factor does not mean that all nekton prefer high-salinity seagrass beds: some species preferred low-salinity beds dominated by *Halodule* (i.e., *A. mitchilli*, *C. nebulosus*, *F. duorarum*). Aboveground biomass was also a significant environmental factor influencing nekton community structure. Seagrass architecture has not been found to be an important factor in whether fish larvae initially settle in a particular area (Bell et al. 1987), but canopy cover can be an indication of greater protection for fish from predators in these environments (Orth et al. 1984). Seagrass leaf density, however, varies seasonally, and juvenile fish density may be more strongly correlated with the location of the seagrass bed in relation to sources of larvae than with the architecture of the bed (Bell et al. 1988).

CONCLUSIONS

The value of seagrass as nekton habitat in Tampa Bay transcends the simple presence or absence of vegetation and can differ based on factors such as location within the bay, seagrass-species composition, and amount of freshwater influence. Relationships between nekton abundance and type and amount of vegetation cover are species-specific, and these microhabitat differences should be documented for each species of interest. Therefore, management decisions regarding seagrass beds, especially those involving mitigation by acre-for-acre substitution of one seagrass

bed for another, need to be made with full consideration of the complex relationships governing the value of a particular seagrass bed for nekton.

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Table 1. Numbers and percentages of 21.3-m seine samples taken, by type of seagrass habitat, for seasonal (spring and fall, 1989–2007) and synoptic (September 2007) sampling.

Seagrass Composition	Seasonal sampling		Synoptic sampling	
	Number	Percent	Number	Percent
<i>Ruppia</i> -dominated	25	1.8%	-	-
<i>Halodule</i> -dominated	668	47.5%	32	59.3%
<i>Syringodium</i> -dominated	146	10.4%	3	5.6%
<i>Thalassia</i> -dominated	413	29.4%	12	22.2%
Mixed	134	9.5%	7	13.0%
Dominated by other seagrass or macroalga	19	1.4%	-	-
Totals	1405	100.0%	54	100.0%

Table 2. Total number and catch per unit effort (CPUE; number 100 m⁻²) of each nekton species collected during seasonal (spring and fall, 1989–2007) and synoptic (September 2007) sampling. Differences in CPUE between seasonal and synoptic data for each species are presented as percentages (% Change).

Species	Seasonal sampling		Synoptic sampling		
	Number	CPUE	Number	CPUE	% Change
<i>nchoa mitchilli</i>	309,036	157.110	10,456	138.307	18.803%
<i>Lagodon rhomboides</i>	206,503	104.984	949	12.553	92.431%
<i>Bairdiella chrysoura</i>	34,621	17.601	468	6.190	11.410%
<i>Orthoprists chrysoptera</i>	34,505	17.542	9	0.119	17.423%
<i>Lucania parva</i>	29,680	15.089	3,078	40.714	25.625%
<i>Harengula jaguana</i>	26,901	13.676	1,589	21.019	7.342%
<i>Farfantepenaeus duorarum</i>	21,007	10.680	2,505	33.135	22.455%
<i>Menidia</i> spp.	18,618	9.465	301	3.981	5.484%
<i>Eucinostomus gula</i>	11,439	5.815	547	7.235	1.420%
<i>Microgobius gulosus</i>	11,133	5.660	205	2.712	2.948%
<i>Sardinella aurita</i>	10,871	5.527	1	0.013	5.513%
<i>Anchoa hepsetus</i>	7,443	3.784			3.784%
<i>Syngnathus scovelli</i>	6,165	3.134	139	1.839	1.296%
<i>Floridichthys carpio</i>	6,026	3.064	112	1.481	1.582%
<i>Cynoscion nebulosus</i>	4,367	2.220	341	4.511	2.290%
<i>Gobiosoma robustum</i>	3,807	1.935	28	0.370	1.565%
<i>Leiostomus xanthurus</i>	3,470	1.764			1.764%
<i>Cyprinodon variegatus</i>	1,678	0.853	2	0.026	0.827%
<i>Opisthonema oglinum</i>	1,506	0.766	157	2.077	1.311%

Species	Seasonal sampling		Synoptic sampling		
	Number	CPUE	Number	CPUE	% Change
<i>Syngnathus louisianae</i>	1,277	0.649	11	0.146	0.504%
<i>Sciaenops ocellatus</i>	1,232	0.626			0.626%
<i>Sphoeroides nephelus</i>	1,012	0.514	9	0.119	0.395%
<i>Haemulon plumierii</i>	953	0.484	3	0.040	0.445%
<i>Eucinostomus harengulus</i>	890	0.226	22	0.291	0.065%
<i>Archosargus</i>					
<i>probatoccephalus</i>	855	0.435	2	0.026	0.408%
<i>Stephanolepis hispidus</i>	849	0.432	3	0.040	0.392%
<i>Chasmodes saburrae</i>	810	0.412	75	0.992	0.580%
<i>Synodus foetens</i>	594	0.302	3	0.040	0.262%
<i>Chilomycterus schoepfii</i>	569	0.289	6	0.079	0.210%
<i>Diplodus holbrookii</i>	567	0.288			0.288%
<i>Syngnathus floridae</i>	541	0.275	20	0.265	0.010%
<i>Lutjanus griseus</i>	513	0.261	32	0.423	0.162%
<i>Callinectes sapidus</i>	511	0.260	17	0.225	0.035%
<i>Hippocampus zosterae</i>	420	0.214	2	0.026	0.187%
<i>Membras martinica</i>	420	0.214	1	0.013	0.200%
<i>Achirus lineatus</i>	339	0.086	23	0.304	0.218%
<i>Hyporhamphus</i> spp. ^a	331	0.084	17	0.225	0.141%
<i>Menticirrhus americanus</i>	330	0.168			0.168%
<i>Oligoplites saurus</i>	310	0.158	21	0.278	0.120%
<i>Strongylura notata</i>	287	0.146	5	0.066	0.080%
<i>Lutjanus synagris</i>	282	0.143	2	0.026	0.117%
<i>Ariopsis felis</i>	248	0.126			0.126%
<i>Acanthostracion</i>					
<i>quadricornis</i>	216	0.055	2	0.026	0.028%
<i>Anchoa cubana</i>	193	0.098	3	0.040	0.058%
<i>Cynoscion arenarius</i>	158	0.080			0.080%
<i>Paralichthys albigutta</i>	147	0.075			0.075%
<i>Gobiosoma bosc</i>	144	0.073			0.073%
<i>Prionotus scitulus</i>	140	0.071			0.071%
<i>Mugil gyrans</i>	123	0.063			0.063%
<i>Symphurus plagiusa</i>	121	0.062	2	0.026	0.035%
<i>Chaetodipterus faber</i>	109	0.055	2	0.026	0.029%
<i>Brevoortia</i> spp.	103	0.052			0.052%
<i>Opsanus beta</i>	76	0.039	4	0.053	0.014%
<i>Fundulus grandis</i>	72	0.037	2	0.026	0.010%
<i>Fundulus similis</i>	67	0.017			0.017%
<i>Menticirrhus saxatilis</i>	64	0.033			0.033%
<i>Strongylura timucu</i>	53	0.027			0.027%
<i>Nicholsina usta</i>	51	0.026	1	0.013	0.013%
<i>Strongylura marina</i>	47	0.024			0.024%
<i>Hippocampus erectus</i>	41	0.021			0.021%

Species	Seasonal sampling		Synoptic sampling		
	Number	CPUE	Number	CPUE	% Change
<i>Dasyatis sabina</i>	38	0.019	1	0.013	0.006%
<i>Prionotus tribulus</i>	38	0.019			0.019%
<i>Sphyræna borealis</i>	37	0.019			0.019%
<i>Centropristis striata</i>	36	0.018	4	0.053	0.035%
<i>Calamus arctifrons</i>	35	0.018			0.018%
<i>Chloroscombrus chrysurus</i>	33	0.017			0.017%
<i>Hypsoblennius hentz</i>	32	0.016			0.016%
<i>Mycteroperca microlepis</i>	32	0.016	15	0.198	0.182%
<i>Mugil curema</i>	31	0.016			0.016%
<i>Mugil cephalus</i>	23	0.012			0.012%
<i>Gobiesox strumosus</i>	22	0.011			0.011%
<i>Bathygobius soporator</i>	14	0.007			0.007%
<i>Eugerres plumieri</i>	14	0.007	1	0.013	0.006%
<i>Rhinoptera bonasus</i>	14	0.007			0.007%
<i>Trinectes maculatus</i>	12	0.006			0.006%
<i>Aluterus schoepfii</i>	11	0.006			0.006%
<i>Ctenogobius boleosoma</i>	10	0.005			0.005%
<i>Diplectrum formosum</i>	9	0.005			0.005%
<i>Microgobius thalassinus</i>	9	0.005			0.005%
<i>Myrophis punctatus</i>	9	0.005			0.005%
<i>Pogonias cromis</i>	9	0.005			0.005%
<i>Serraniculus pumilio</i>	9	0.005			0.005%
<i>Elops saurus</i>	7	0.004			0.004%
<i>Pomatomus saltatrix</i>	7	0.004			0.004%
<i>Limulus polyphemus</i>	6	0.003			0.003%
<i>Rhinobatos lentiginosus</i>	6	0.003			0.003%
<i>Gymnura micrura</i>	5	0.003			0.003%
<i>Scomberomorus maculatus</i>	5	0.003			0.003%
<i>Sphoeroides spengleri</i>	5	0.003			0.003%
<i>Sphyræna guachancho</i>	5	0.003			0.003%
<i>Anarchopterus criniger</i>	4	0.002	1	0.013	0.011%
<i>Fundulus confluentus</i>	4	0.002			0.002%
<i>Menippe</i> spp.	4	0.002			0.002%
<i>Poecilia latipinna</i>	4	0.002	2	0.026	0.024%
<i>Scorpaena brasiliensis</i>	4	0.002			0.002%
<i>Haemulon aurolineatum</i>	3	0.002			0.002%
<i>Paraclinus marmoratus</i>	3	0.002			0.002%
<i>Stephanolepis setifer</i>	3	0.002			0.002%
<i>Trachinotus falcatus</i>	3	0.002			0.002%
<i>Albula vulpes</i>	2	0.001			0.001%
<i>Aluterus scriptus</i>	2	0.001			0.001%
<i>Astroscopus y-graecum</i>	2	0.001			0.001%
<i>Caranx hippos</i>	2	0.001			0.001%

Species	Seasonal sampling		Synoptic sampling		
	Number	CPUE	Number	CPUE	% Change
<i>Dasyatis say</i>	2	0.001			0.001%
<i>Hemiramphus brasiliensis</i>	2	0.001			0.001%
<i>Selene vomer</i>	2	0.001			0.001%
<i>Ancylopsetta quadrocellata</i>	1	0.001			0.001%
<i>Centropomus undecimalis</i>	1	0.001			0.001%
<i>Decapterus punctatus</i>	1	0.001			0.001%
<i>Dorosoma petenense</i>	1	0.001			0.001%
<i>Epinephelus morio</i>	1	0.001			0.001%
<i>Fistularia tabacaria</i>	1	0.001			0.001%
<i>Gambusia holbrooki</i>	1	0.001			0.001%
<i>Halichoeres bivittatus</i>	1	0.001			0.001%
<i>Hypleurochilus</i>					
<i>caudovittatus</i>	1	0.001			0.001%
<i>Rimapenaeus constrictus</i>	1	0.001			0.001%
<i>Serranus subligarius</i>	1	0.001			0.001%
<i>Sphyrna barracuda</i>	1	0.001			0.001%
<i>Sphyrna picudilla</i>	1	0.001			0.001%
<i>Sphyrna tiburo</i>	1	0.001			0.001%
<i>Urophycis floridana</i>	1	0.001			0.001%
<i>Monacanthus ciliatus</i>			1	0.013	0.013%
Number of taxa	120		50		

^a The majority of *Hyporhamphus* spp. were *H. meeki*.

Table 3. Catch per unit effort and percentage contribution of the top 10 species contributing to differences in nekton community structure seen in seasonal (fall, 1989-2007) and synoptic (September 2007) sampling, by seagrass composition. Economically important species are set in boldface type. If no percentage contribution is listed, then that species is not in the 10 ten species contributing to differences. An 'X' is indicated by each species that showed the same trend in abundance by seagrass composition for both datasets.

Species	Seasonal sampling				Synoptic sampling				Same trend?
	<i>Halodule</i>	<i>Thalassia</i>	%	Contrib.	<i>Halodule</i>	<i>Thalassia</i>	%	Contrib.	
<i>Anchoa mitchilli</i>	129.84	40.54	9.04	202.59	81.61	8.26			X
<i>Bairdiella chrysoura</i>	6.33	5.08	3.50	4.51	8.27	5.36			
<i>Cynoscion nebulosus</i>	3.52	1.49	3.39	6.65	0.71	6.19			X
<i>Eucinostomus gula</i>	10.62	11.77	-	9.49	4.88	4.51			
<i>Farfantepenaeus duorarum</i>	21.86	14.94	5.67	51.65	2.44	8.41			X
<i>Harengula jaguana</i>	13.15	16.04	4.76	26.03	2.2	6.77			
<i>Lagodon rhomboides</i>	3.7	6.52	3.83	6.94	23.33	6.19			X
<i>Lutjanus griseus</i>	0.31	0.9	-	0.45	0.54	3.26			X
<i>Lucania parva</i>	15.28	10.43	3.97	48.01	28.45	10.09			X
<i>Menidia</i> spp.	9.6	4.64	3.8	6.27	0.89	3.6			X
<i>Microgobius gulosus</i>	12.86	1.46	5.59	4.58	0	-			X
<i>Syngnathus scovelli</i>	3.03	3.02	3.45	1.9	2.2	-			

Table 4. Catch per unit effort and percentage contributions of the top 10 species contributing to differences in nekton community structure observed during synoptic sampling (September 2007) for two categories of historical freshwater influence. Economically important species are set in boldface type.

Species	CPUE		
	Freshwater-influenced	Not freshwater-influenced	% Contribution
<i>Anchoa mitchilli</i>	153.15	123.47	10.88
<i>Lucania parva</i>	46.08	35.34	9.18
<i>Farfantepenaeus duorarum</i>	60.19	6.08	7.45
<i>Harengula jaguana</i>	15.61	26.43	5.75
<i>Microgobius gulosus</i>	5.40	0.03	5.32
<i>Lagodon rhomboides</i>	6.22	18.89	5.06
<i>Syngnathus scovelli</i>	3.47	0.21	4.70
<i>Bairdiella chrysoura</i>	10.05	2.06	4.67
<i>Cynoscion nebulosus</i>	7.57	1.46	3.85
<i>Chasmodes saburrae</i>	1.98	0	3.72

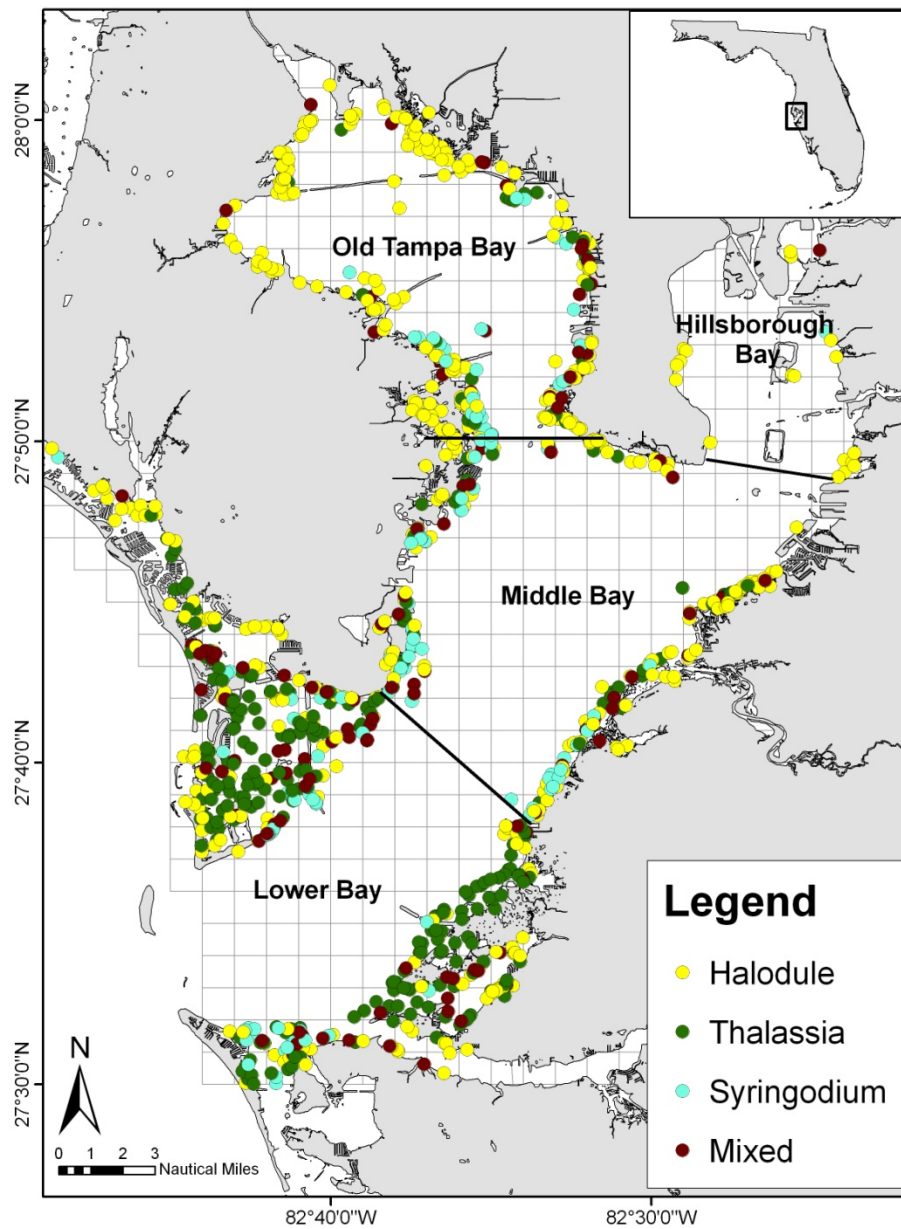


Figure 1. Seagrass species composition at seasonal sampling sites in Tampa Bay (spring and fall, 1989-2007) having $\geq 50\%$ total seagrass coverage. Sites composed of at least 50% of one seagrass species were considered dominated by that species (e.g., *Halodule*-, *Thalassia*- or *Syringodium*-dominated). If the dominant seagrass species accounted for $< 50\%$ of coverage in a bed, then the bed was considered mixed. Samples collected in areas dominated with *Ruppia* or other species of seagrasses are not included due to low frequency. Regional divisions and the one nautical-mile squared sampling grid are also shown.

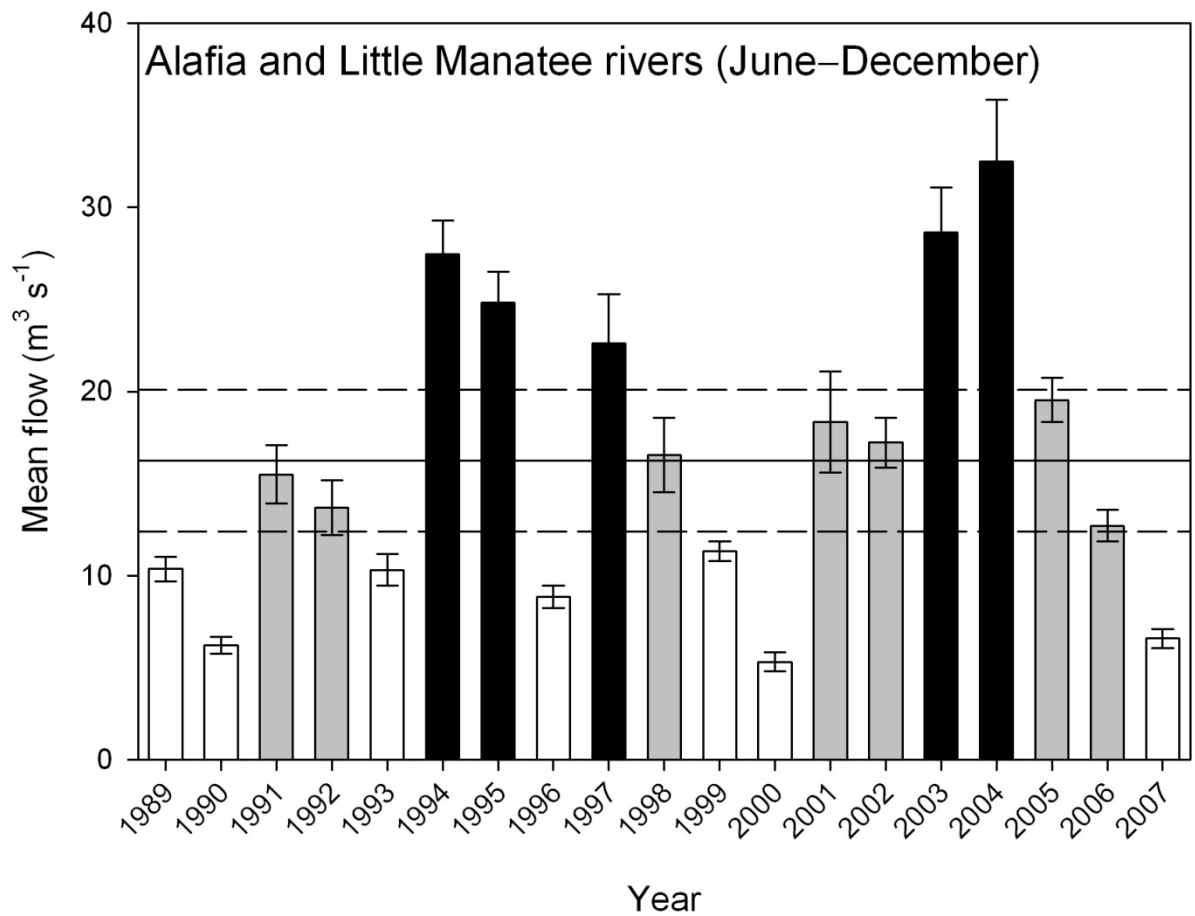


Figure 2. Mean seasonal river inflow for the Alafia and Little Manatee rivers combined (June–December, 1989–2007). The solid horizontal line indicates the mean annual river flow over the study period; the dashed lines indicate the 95% confidence intervals. White bars indicate years with below-average inflow; gray bars indicate years with average inflow; black bars indicate years with above-average inflow. Error bars represent the standard error of the mean.

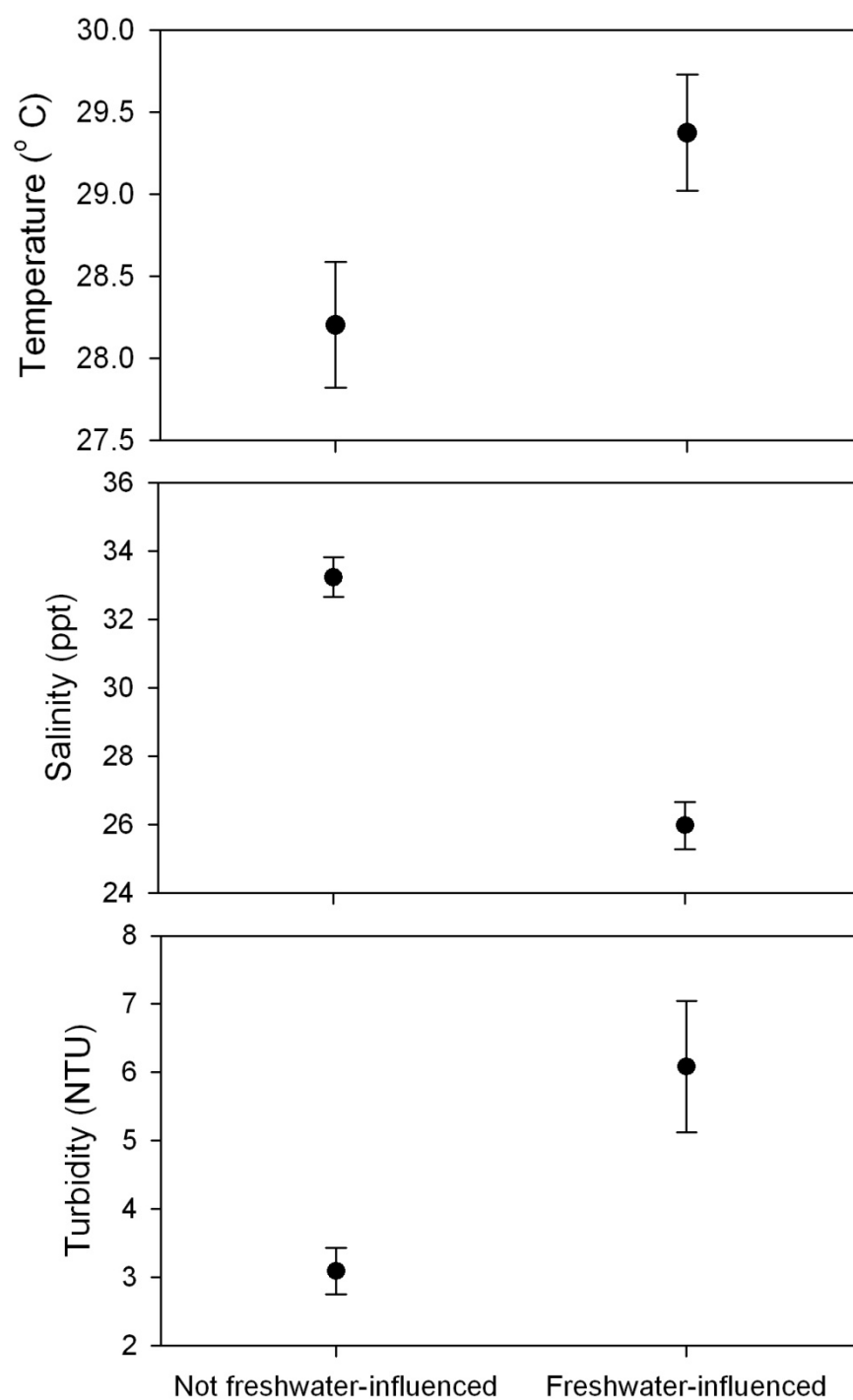


Figure 3. Means and standard errors for temperature, salinity, and turbidity data collected during synoptic sampling (September 2007), by freshwater-influence category.

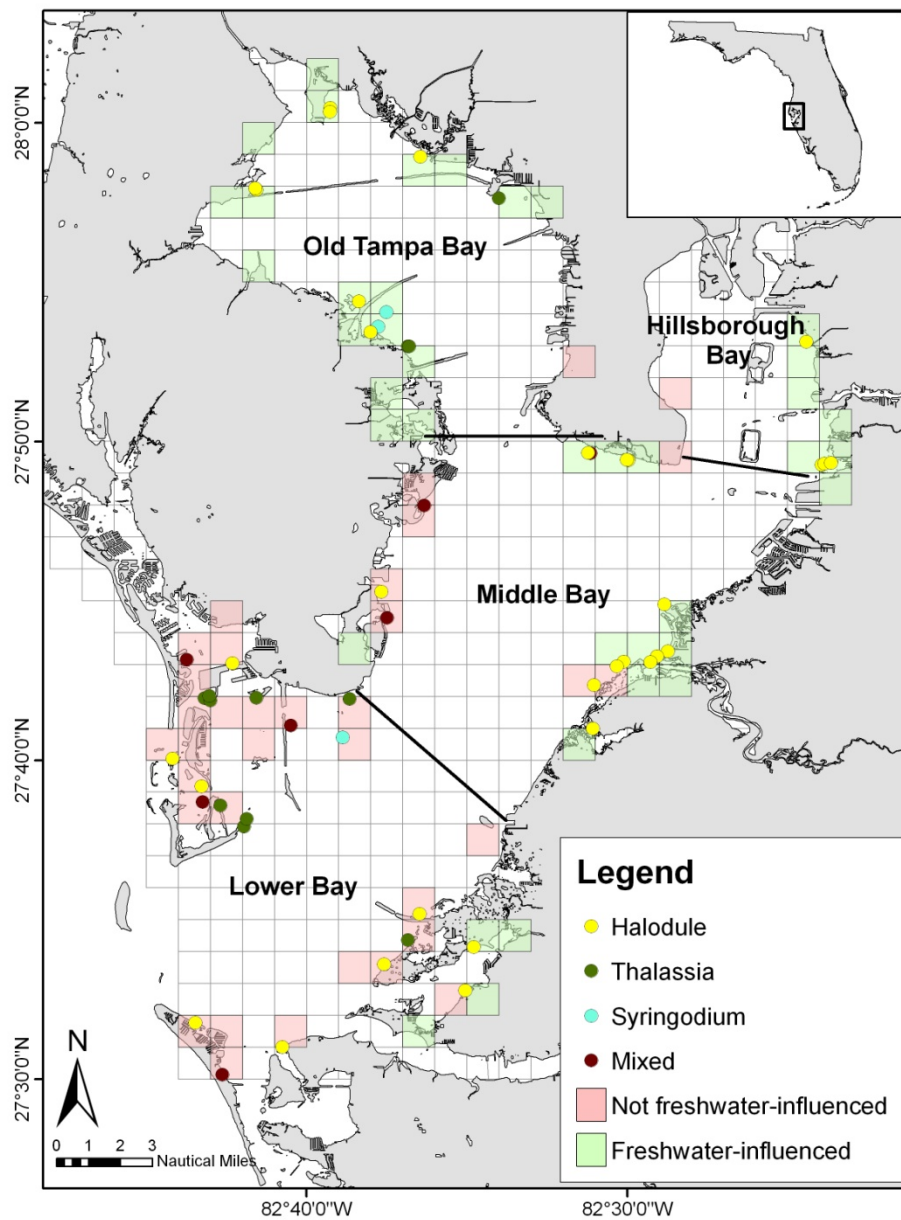


Figure 4. Seagrass species composition at synoptic sampling sites in Tampa Bay (September 2007). Sites composed of at least 50% of a one seagrass species were considered dominated by that species (e.g., *Halodule*-, *Thalassia*- or *Syringodium*-dominated). If the dominant seagrass species accounted for < 50% of coverage in a bed, then the bed was considered mixed. Regional divisions and the one nautical-mile squared sampling grid are also shown. Grid cells characterized by low (pink) or high (green) freshwater influence are indicated.

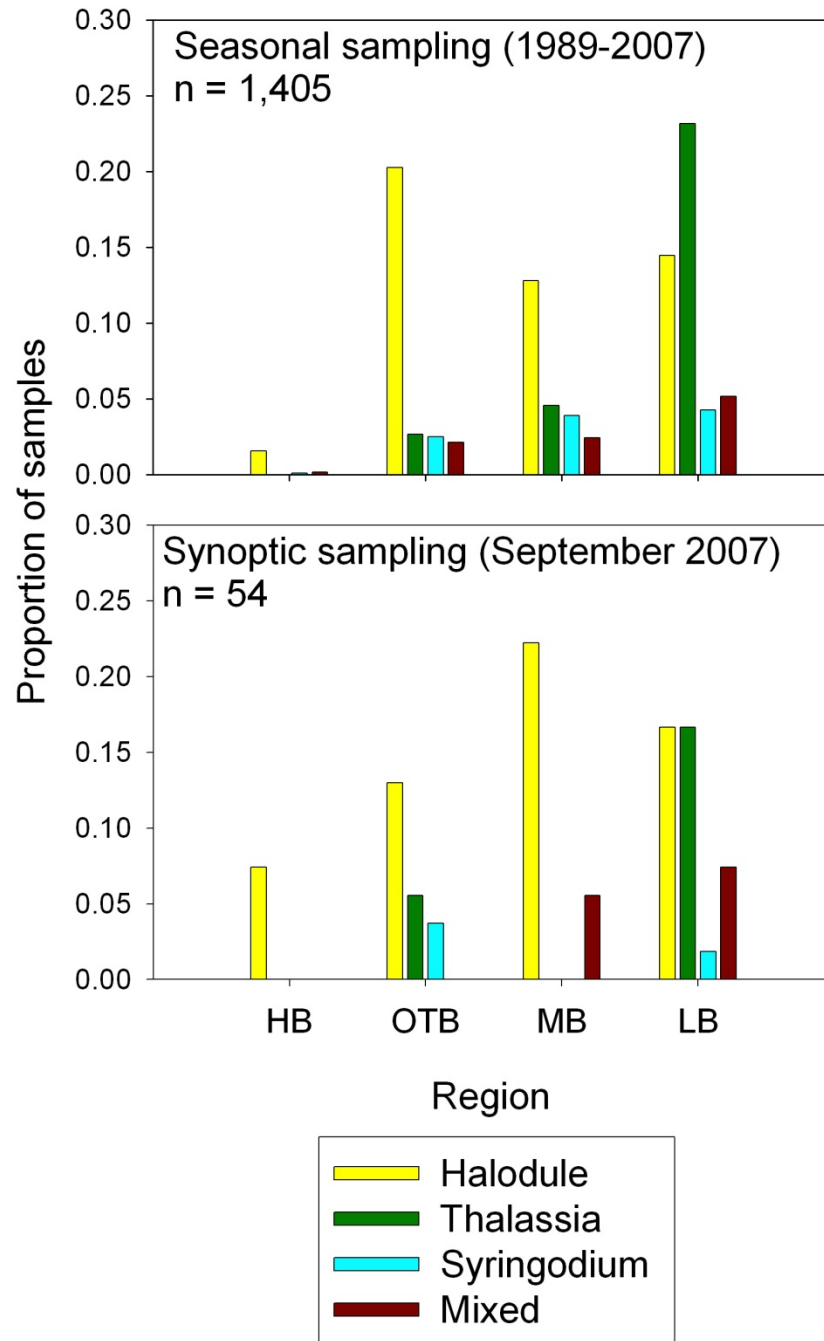


Figure 5. Proportion of nekton samples collected in each region by dominant seagrass composition (e.g., *Halodule*-, *Thalassia*- or *Syringodium*-dominated) for seasonal (spring and fall, 1989-2007) and synoptic (September 2007) sampling. Samples collected in areas dominated by *Ruppia* or other seagrasses are not included due to low frequency.

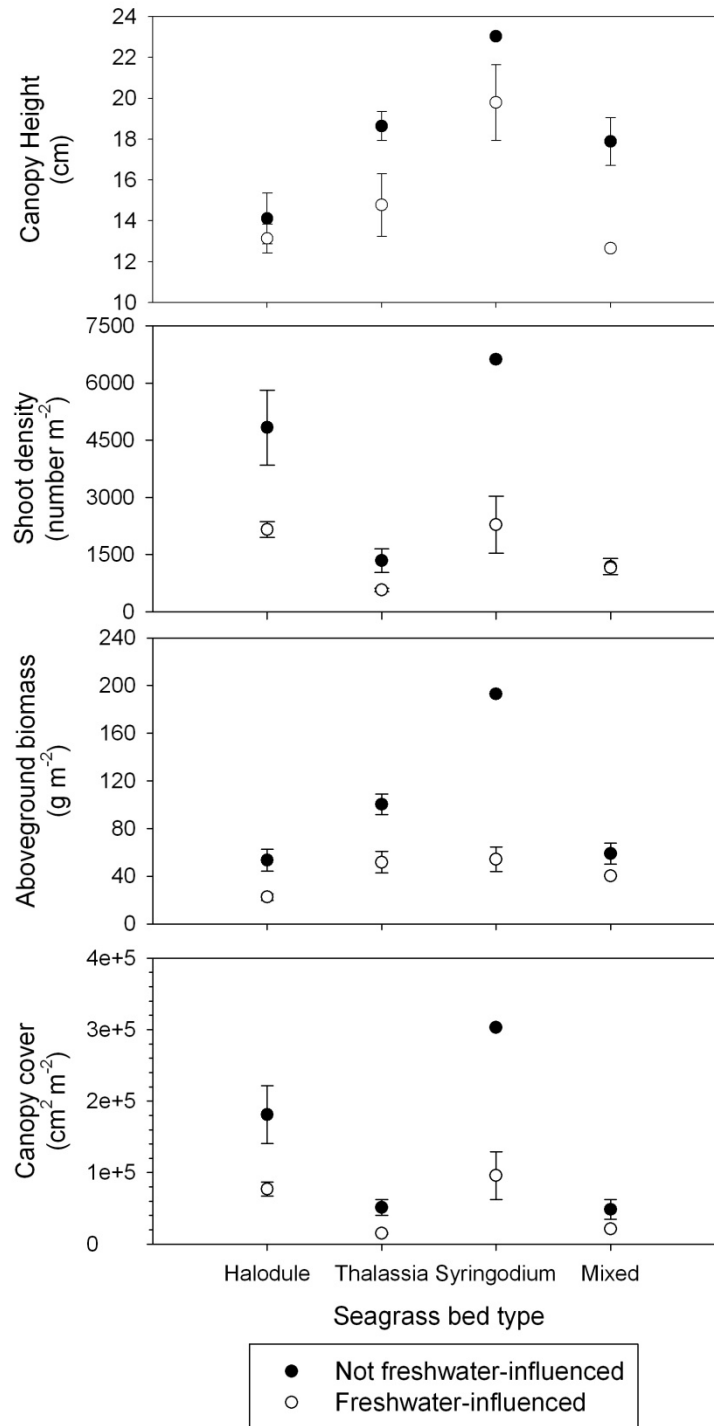


Figure 6. Means and standard errors for canopy height, shoot density, aboveground biomass, and canopy cover measured in seagrass cores collected during synoptic sampling (September 2007), by dominant seagrass and freshwater influence.

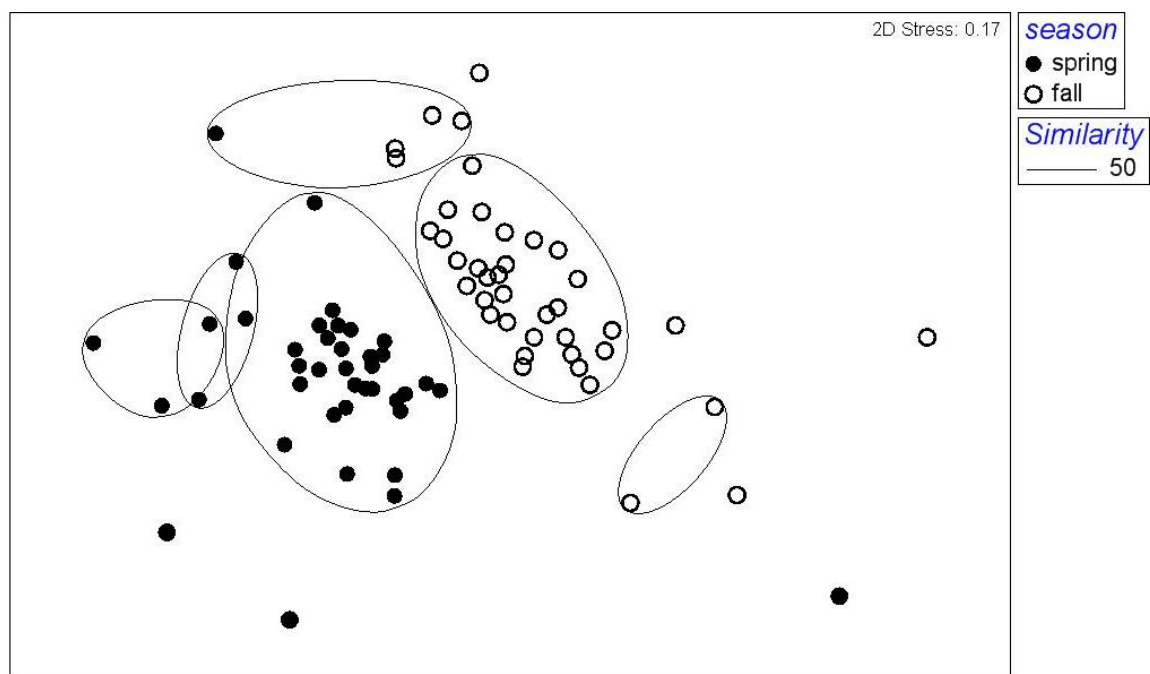


Figure 7. Multivariate ordination plot based on nonmetric MDS demonstrating seasonal differences in assemblages of nekton collected with 21.3-m seines in Tampa Bay during seasonal sampling (1989–2007). Bray-Curtis similarity percentages of 50% from hierarchical agglomerative cluster analysis are included as ellipses. The stress value of the two-dimensional ordination is indicated in the top right hand corner.

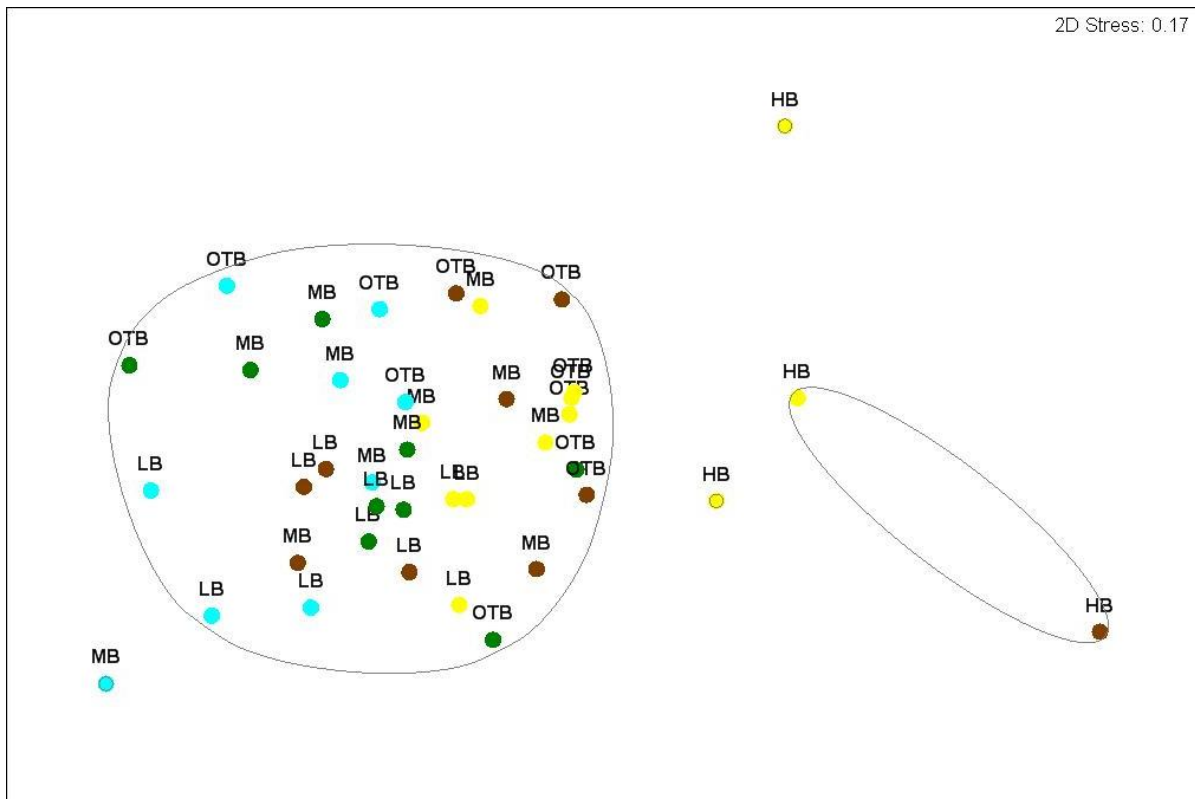


Figure 8. Multivariate ordination plot based on nonmetric MDS of data averaged over seagrass composition, river inflow, and region for assemblages of nekton collected with 21.3-m seines in Tampa Bay during seasonal sampling (fall, 1989–2007). Regions are indicated by labels; colors indicate different seagrass composition (*Halodule*-dominated = yellow; *Thalassia*-dominated = green; *Syringodium*-dominated = blue; mixed seagrass beds = brown). Bray-Curtis similarity percentages of 50% from hierarchical agglomerative cluster analysis are included as ellipses. The stress value of the two-dimensional ordination is indicated in the top right hand corner.

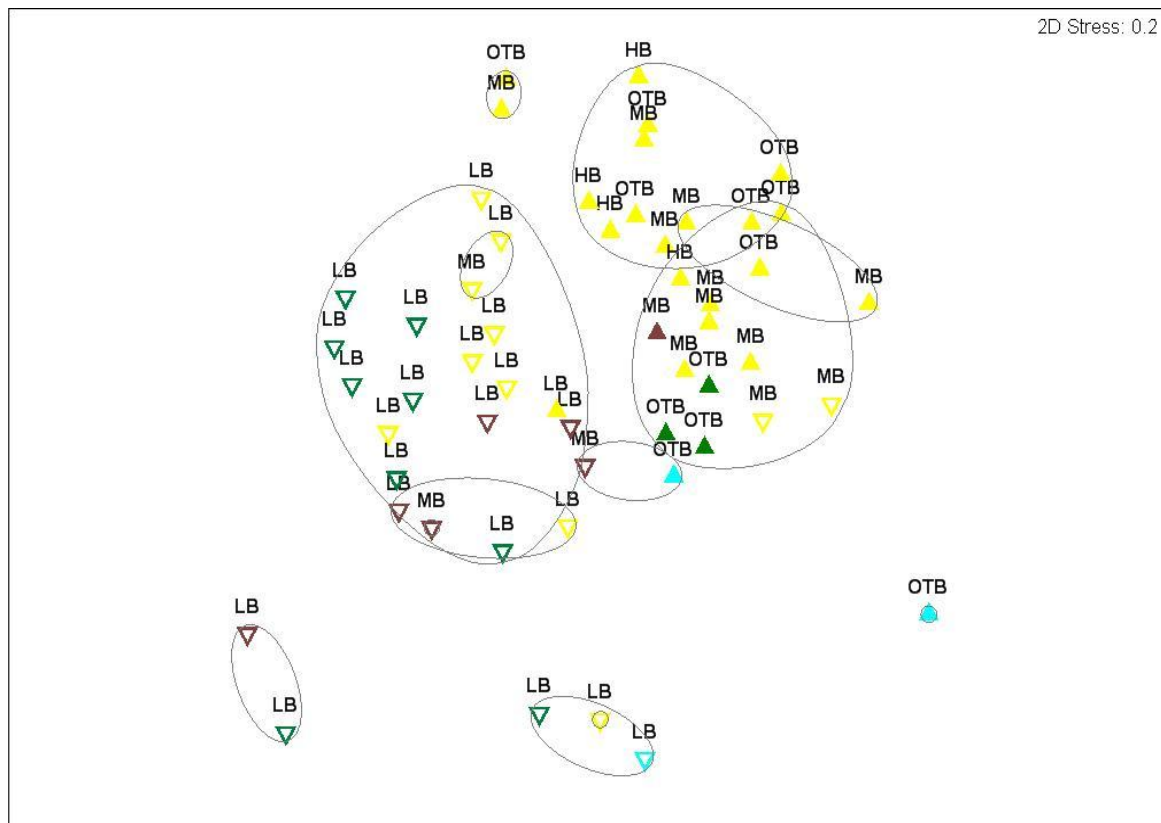


Figure 9. Multivariate ordination plot based on nonmetric MDS of data by seagrass composition, freshwater influence, and region for assemblages of nekton collected with 21.3-m seines in Tampa Bay during synoptic sampling (September 2007). Regions are indicated by labels; colors indicate different seagrass composition (*Halodule*-dominated = yellow; *Thalassia*-dominated = green; *Syringodium*-dominated = blue; mixed seagrass beds = brown). Filled, upward triangles represent areas influenced by freshwater, while empty, downward triangles represent areas not influenced by freshwater. Bray-Curtis similarity percentages of 50% from hierarchical agglomerative cluster analysis are included as ellipses. The stress value of the two-dimensional ordination is indicated in the top right hand corner.

ABSTRACT
Session 7: TAMPA BAY BIOTA
PRESENTATION

An Overview of Recent Research on the Florida Manatee in Tampa Bay: Population, Ecology, and Behavior. C. Deutsch, H. Edwards and P. Carlson (FL Fish & Wildlife Conservation Commission, Fish & Wildlife Research Institute). This overview of recent studies on the Florida manatee in the Tampa Bay region covers long-term population monitoring and short-term ecological/behavioral research. Population monitoring activities include: photo-identification of distinctly scarred individuals to estimate adult survival rate; carcass salvage and necropsy to determine causes of death; and aerial surveys to assess distribution and abundance. The major challenge with using surveys to estimate abundance and infer population trends has been the difficulty in assessing spatio-temporal variation in detection rate. To address this problem FWRI is developing new aerial survey methodologies that should provide statistically unbiased estimates of population size; pilot studies are being conducted in the Tampa Bay area. Behavioral studies have focused on manatee winter use of warm-water aggregation sites, as these provide critical habitat for long-term population viability. We investigated manatee foraging movements and attendance patterns at industrial warm-water refugia in Tampa Bay by tracking 32 manatees over four winters using Argos-linked GPS tags. Manatees behaved as central-place foragers, using a power plant's discharge canal as a focal point from which to make feeding trips to seagrass beds throughout the bay. During winter manatees spent an average of 47% of their time in a thermal refuge. Peak presence at the power plant occurred during mid-day, whereas manatees primarily foraged at night. Access to shallow, nearshore grassbeds was limited to times of higher tidal levels, which typically occurred at night. While traveling across the bay manatees often followed the bottom contour; they reached depths of 16.3 m, corresponding to the bottom of the shipping channel. An FWRI study to assess the effects of manatees on seagrass communities around the aggregation site using exclosures found that biomass and leaf productivity were reduced overwinter and recovered during the growing season, but the impacts of manatee foraging were confounded with those of ray activity. [chip.deutsch@MyFWC.com]

A METHOD FOR ASSESSING TIDAL CREEK STATUS AND TRENDS IN SOUTHWEST FLORIDA USING A CONDITION INDEX BASED ON ECOLOGICAL VARIABLES AND RAPID-SURVEY METHODS

E.D. Estevez, K.L. Meaux, J.K. Culter, J. Sprinkel & R. Janneman

ABSTRACT

The southwest Florida coast is a homogeneous ecological landscape characterized by numerous, usually short tidal creeks located in natural to urban settings. Watershed managers seek rapid-survey methods to accurately describe the ecological condition of tidal creeks in order to document their status and trends. A 4-year project produced a creek index based on oyster, other mollusk, and other invertebrate, fish, and nuisance algae variables used to represent bioassessment attributes. All variables were measured in 16 tidal creeks in Sarasota and Charlotte Counties, Florida, during May-June periods. Creeks are tributaries of Sarasota, Little Sarasota, Blackburn, and Lemon Bays and comprise a homogeneous, natural set of streams and watersheds reflecting gradients of human disturbance. Stream sanitation and habitat alterations affected sampling opportunities and methods. Additional surveys were made in Manatee and Hillsborough counties. Data were compared by transformations to unit-less values and values were combined as geometric means. The index sorted creeks into a large group with scores (intermediate condition) and smaller groups that were higher (better) and lower (worse), but all creek scores were less than 50% of possible maxima that were based on the highest observed values of each variable in any creek. Pearson correlations of index scores to benthic infaunal community data (used as an independent reference standard) were significant ($p < 0.05$) for faunal density and highly significant ($p < 0.01$) for species richness. In 2007 the index was not significantly correlated with a Landscape Development Intensity Index, mean age of watershed development, or other available abiotic measures of creek or watershed condition, but benthic species richness and densities were. Subsequent annual sampling indicated an increase in creek condition after two successive years of drought was reversed by a third dry year. Monthly sampling in one reference creek exhibited changes that may be seasonal, a response to the end of the drought, or both. This project was supported by Sarasota County, the Manasota Basin Board, and the Southwest Florida Water Management District.

INTRODUCTION

Tidal Creeks and Ecological Indices—Small streams are abundant and valued national assets (Paul and Meyer 2001; USEPA 2006) and their assessment for management has been a widespread endeavor (USFWS 1981; Plafkin et al. 1989; Roth et al. 1996), with particular emphasis on bioassessment (USEPA 2006; 2002) including rapid-survey methods (USEPA 2006; Plafkin et al. 1989; Jackson et al. 2000). Tidal creeks are among the commonest of coastal landscape components and are of vital importance for the maintenance of estuarine and marine productivity (Day et al. 1989). Assessment methods have been formulated for various estuarine environments (VanDolah et al. 1999), including Florida waters (McCarron and Frydenborg 2005; Fore et al. 2007; Janicki Environmental 2005), but progress in tidal creek settings has been relatively recent (Holland et al. 2004) and based on traditional field approaches (Lerberg et al. 2000). Recent work in Tampa Bay's tidal creeks has provided new insight to their ecological function and management status (Tampa Bay Tidal Tributary Research Team 2008) but at present, no rapid bioassessment methods are in common use in tidal creeks, despite a call for a new generation of coastal water indicators (Niemi et al. 2004). Efforts in Florida (Barnes and Mazzotti 2005) and other states (Yoder and Rankin 1998) presage the development of new sampling and reporting programs tailored to the tidal creek environment, in order to meet the challenges of urbanization (Holland et al. 1997; Valentine-Rose et al. 2007), declining water quality (Bricker et al. 1999), and climate change (Callaway et al. 2007). Tidal Creeks of Sarasota

County, southwest Florida, offer a favorable venue for advancing the methods of rapid bioassessment.

Sarasota County Creeks—Sixteen tidal creeks enter Sarasota, North Roberts, Little Sarasota, Blackburn, Venice, and Lemon bays, from north to south: Walker (Whitaker) Bayou, Hudson Bayou, and Phillippi, Matheny, Clower, Catfish, North, South, Shakett, Curry, Hatchett, Alligator, Woodmere, Forked, Gottfried, and Ainger Creeks. All but two creeks are entirely within Sarasota County; Gottfried and Ainger Creeks head in Sarasota County but their mouths are in Charlotte County. Phillippi Creek is the largest; Clower Creek is the smallest. Creek-mouth distances to nearest inlet are 0.9-12.0 km and average 5.8 km. Northern creeks are more urbanized than southern creeks.

Purpose of this Paper—In 2005 Sarasota County and Mote Marine Laboratory began a program to develop a tidal creek condition index (TCCI) that was based on biological rather than physical or chemical variables; employed rapid-survey techniques; could be used to assess status and trends in the condition of a creek including responses to pollution events, and allowed for comparisons among creeks. This paper describes the history of preliminary efforts (2005); assessments of creeks, stations, and candidate index variables (2006), and field data, data transformation, and index definition (2007). The 2007 TCCI was also compared to benthic faunal, creek sediment, and watershed data. Index values in 2008-2010, new statistical analyses, and ecological interpretations of the TCCI will be reported in later publications.

METHODS

Preliminary Studies and Interim Results—A literature review established that the county's coastal creeks comprised a homogeneous set of tidal streams with respect to geological, hydrological, oceanographic, and biogeographical features, but that tidal creeks associated with the Myakka River in Sarasota County were not associable with coastal creeks (Estevez 2005b; 2007d). Follow-on studies therefore focused on coastal creeks. Existing watershed data compiled by Sarasota County (Estevez 2005b) were deemed sufficient to characterize each coastal creek's upland influences. To test the hypothesis that creeks' ecological conditions reflected the condition of their respective watersheds, an ecological reconnaissance was made in Whitaker Bayou and Gottfried Creek. Whitaker Bayou is in a long-urbanized, highly-developed basin with high septic tank density, while Gottfried Creek's basin characteristics differed from Whitaker Bayou's more than most of the study creeks (Estevez 2005b). Dramatic differences between the two tidal creeks were documented (Estevez 2005a; 2005c; 2007d;). In May and June 2006, 15 coastal creeks were surveyed to evaluate logistics and test candidate variables for index use (Estevez 2007d). Sarasota County's dry seasons are more reliably and uniformly dry than wet seasons are wet because localized rainfall events cause watershed runoff to vary considerably along the coast: mean salinity in 2006 was 30 ppt with salinity in 2 creeks > 40 ppt (Estevez 2007d). Dry season creeks are characterized by high transparency, rare hypoxia, and high species richness. Logistically, divers are less exposed to electrical storms or septic waste pollution in dry seasons. One station was chosen in each creek. Reasons included the extreme brevity (< 1 km) of some creeks caused by natural conditions or instream barriers; proximity to existing Sarasota County water quality or oyster monitoring stations (Jones 2006); representativeness of common habitats, communities, and bottom types, and considerations of access, safety, and the likelihood of future dredging projects. Twenty-eight rapid-survey variables were measured to describe the location, habitat composition, sediment types, and water quality of each site (stream sanitation limited the ability to sample in four creeks). An additional 31 biological variables were selected to describe the flora and fauna at each site. Measurements were made in 2 strata (intertidal, subtidal) in each creek: for coastal waters of Sarasota County a water depth of at least 37 cm was assumed to overlay subtidal

bottom under usual conditions. Wherever possible, station locations and the location of specific measurements and samples made within stations were biased to represent the most favorable conditions that could be identified during creek reconnaissance and on the day of field effort in each creek.

The 2006 trial found that subtidal settings generally produced more useful data than intertidal settings, and that about half of the biological variables could be measured at half or more of the creeks. No single variable could be measured in both strata of every creek. Different approaches were used to standardize and combine variables into potential index scores, and prototypic indices performed well enough to warrant further development (Estevez 2007d).

Preparatory Tasks for 2007 Tidal Creek Condition Index Implementation—A more comprehensive effort was begun in 2007 to survey all coastal creeks in Sarasota County using a larger list of biological variables and more robust sampling effort. Creeks were revisited, some 2006 stations were moved, and new stations were added to creeks not sampled in 2006. Universal identification numbers were assigned with assistance from the Southwest Florida Water Management District (Estevez 2007a). Data from 2006, and some new data collected early in 2007, were analyzed using rarefaction methods to determine adequate sample sizes and replication, as illustrated in Figure 1 (Estevez 2007a). Results were used to prepare standard operating procedures and field data sheets for field sampling (Estevez 2007a) and a metadata report was developed for the project, using the USGS South Florida Information Access (SOFIA) template (Estevez 2008b).

2007 Field and Analytical Methods—Stations in 16 creeks were sampled from May 20 to June 8, 2007. Up to 49 biological variables were measured intertidally and again subtidally in each creek where the opportunity existed (Estevez 2007a; Estevez 2007c). Only rapid-survey methods were employed. As in 2006, location of specific measurements and samples made within stations were biased to represent the most favorable conditions that could be identified on the day of field effort in each creek. Alternatives were employed to avoid body-contact with polluted waters. Oyster data were provided by Sarasota County (Jones 2006).

Although the purpose of the 2007 effort was to identify a smaller set of variables upon which a creek condition index would be calculated, it was worthwhile to examine the complete data set with respect to creek grouping in terms of their similarity using the original and entire data set. In order to anticipate problems that rare, large values can cause in similarity analyses, and to generalize the comparison of creeks as much as possible, all data were converted to a presence/absence basis. Similarities among creeks were examined by applying a hierarchical cluster analysis (in SPSS) to presence/absence of chosen environmental variables for each creek, using an average linkage clustering with a Euclidean distance matrix. Subsequently, Pearson correlation coefficients were computed between individual variables using a pair-wise approach that employed all original data for every variable. Data from all 16 creeks were used, as available for each variable. One set of analyses was made for intertidal data and another was made for subtidal data sets because of differences in the size, evenness, and coverage of variables in the two settings, as previously described. Then correlations were determined among variables between intertidal and subtidal settings (Estevez 2008a).

RESULTS

Description of All Data—Approximately 4,425 biological data were collected (Table 1). Data were analyzed in a stepwise manner to assess data completeness and distribution by creek, identify correlations among variables, screen variables to achieve index objectives, select

variables for the TCCI, establish a data transformation, and to develop and test methods to aggregate transformed data into the index.

Similarities among creeks were examined by applying a hierarchical cluster analysis to presence/absence of index variables (Figure 2). There was a threefold difference in the number of data available across creeks. Twenty-six variables accounted for 90% of all data; zero values dominated the data space, and subtidal data were twice as numerous as intertidal data. Two groups were evident with one defining an affinity among Whitaker Bayou, and Clower, Matheny, Hatchett, and Woodmere Creeks. The five creeks most distant in similarity from these five were Forked, Alligator, Ainger, North, and Gottfried Creeks.

Pair-wise Pearson correlation coefficients were computed between individual variables using a pair-wise approach that employed all original data for every variable. There were 110 significant or highly significant variables pairs among the 1,317 comparisons (8.3%). The variable most frequently correlated to others with significance was intertidal *Tagelus plebeius* extent, with six pairs. The significant correlation frequency with other variables were, in descending order: measurements of extent, mollusk density, submerged aquatic vegetation, percent cover, oyster attributes, *Tagelus* attributes, net catches, and burrow density.

Selection of Index Variables—Once the size and characteristics of the creek database were known, variables were screened to achieve index objectives using these criteria:

A. Multiple levels of biological organization. A robust index draws from biological and ecological attributes spanning a hierarchy of levels from the condition (health, disease) of single organisms, to ecosystem structure, function, and production. The Florida Department of Environmental Protection seeks to integrate bioassessment across multiple attribute groups (Table 2), where indicator and keystone species, and longevity have been added.

B. Relationship to creek and watershed condition and management. Useful variables are those with known responses to proximate stressors that are in turn related to watershed influences (Barnes and Mazzotti 2005). The longevity of oysters, for example, is regulated by freshets that result either from natural runoff or from the operation of instream control structures. Other factors being equal, a creek with altered hydrology that experiences significantly greater oyster mortality than a comparable creek with natural hydrology may be considered impaired with respect to oyster resources; an adverse salinity regime is the proximate stressor causing the mortality, and the operation of instream structures is the watershed management practice requiring modification.

C. Multiple strata. To the extent possible, variables should be chosen so as to work in each of the strata identified from prior work, study design, or companion studies of resource distribution, water quality, human activities, etc. Where spatial, physical, ecological, or anthropogenic conditions such as urbanization prevent variables from universal use, but multiple strata are more or less evenly represented in creeks, variables should be included to provide some information from each stratum. For example, creeks present both intertidal and subtidal settings and an index that captures unique resource values in both settings may be more representative than otherwise.

D. Scope for change. Index variables should have scope to change as a creek's condition improves or worsens. Scope for change can be provided in terms of how samples and measurements are defined for a variable, as in gear, method, and replication decisions. Another method for providing scope is to consider variables that indicate impairment such as the presence of nuisance accumulations of macroalgae.

E. Logistics. The selection of index variables must necessarily be advised by logistical constraints such as access, time on station, the need to sample multiple creeks in a day, and crew safety.

Variables Selected for the Tidal Creek Condition Index—Nine variables were selected for the 2007 creek index (Table 3).

Periphyton: bluegreen algae and other algae that grow as a flat formation on the bottom.

Cyanobacteria and heterotrophic microbes are often present or dominant components of periphyton; formations may be linear, clumpy, tuft-like, or continuous; short and stubby or a few centimeters in height.

Filamentous algae: stringy masses of living and dead bluegreen (*Lyngbia*) or green (*Cladophora*, *Chaetomorpha*, *Enteromorpha*) algae that begin growing as forms attached to a hard substratum but may become drift forms, masses entangled on snags, or shoreline windrows; thick accumulations may raft or have chlorotic tops and become a nuisance.

Bioturbation: physical signs of sediment being reworked by epifaunal and infaunal animals. Examples include surface trails, excavations made by foraging predators, worm castings, burrows, and other micro-relief on the sediment surface. The index utilizes burrow density.

Bare-bottom dipnet fauna: fishes, crabs, shrimps, and “other” fauna are epibenthic or demersal in nature, and responsive to water and sediment quality (TBTTRT 2008). “Other fauna” includes other crustaceans such as isopods or amphipods, small gastropods, small tunicates, or echinoderms. SAV and especially drift macroalgae harbor higher numbers of such fauna, but the vegetated bottom types do not occur in all creeks.

Tagelus density: the stout or short razor clam, *Tagelus plebeius*, prefers shallow, flowing water and clean sediments, and its presence is often made evident by the holes or depressions made by foraging stingrays. *Tagelus* is easy to recognize in the field. Live *Tagelus* are counted and returned alive after sizing.

Tagelus reproductive success: seasonal reproduction and pulsed growth produce distinctive size classes (cohorts) that indicate the persistence of the species in a creek. One or more cohorts may be missing in a given creek or station.

Other live mollusks: include epifaunal and infaunal forms often more resistant to environmental stress than *Tagelus*. Live mollusks are returned alive after counting.

Crassostrea survivorship: oysters may occur as single individuals, clumps, or reefs. Sarasota County’s oyster monitoring program conducts systematic studies of selected intertidal reefs in most of the creeks. The 2007 effort used county data and contributed data in unstudied creeks: numbers of live and dead oysters were counted and the percentage of live oysters was used for the index.

Crassostrea longevity: characterized by the mean height of the largest oysters at each reef.

FORMULATION OF THE TIDAL CREEK CONDITION INDEX

Variables selected for index use span both intertidal and subtidal settings and several levels of biological organization (Table 2). Variables also have known stressors that in many cases can be mapped to watershed influences amenable to management (Table 4). Three variables were not

significantly correlated with other index variables; *Tagelus* count was significantly correlated to four index variables. Use of the data is described below.

Establishing a Data Transformation Process—The nine variables selected for use present five different unit systems. Before the variables can be combined into an index it is necessary to transform data into unit-less values. Upon reviewing data distributions for the variables the following system was employed. Four bins were used to distribute original data for each variable. Bin 1 was used for data with zero value: this was deemed necessary for aggregation methods described in 4.2. The remaining non-zero data were divided into thirds relative to their maximum values. To account for future values larger than 2007 maxima, the upper limits of Bin 4 were calculated using 2007 maxima plus an increment, usually about 10%, based on the nature of the variable. The final system by which data were distributed is shown in Table 5.

Original field data were used to create binned values for the index rather than transformed data (e.g. number of burrows per m²) so that field crews can compute the creek index while on station, if needed. As another measure to facilitate sampling and measurement, 2007 field results established that dip-net catches >20 total individuals were difficult to enumerate, but numbers >20 could be logged as such with confidence. Because Bin 1 was used for undesirable conditions and Bin 4 was used for desirable ones, the assignment of field data for nuisance algae was reversed so that high percent cover values lowered index scores.

Aggregating Transformed Data into the Index—When a creek index is aggregated from multiple variables three objectives are desirable: that the index scores be comparable among creeks; that one creek's index scores be comparable through time, and that the index is relatively insensitive to missing data. After testing other methods, geometric means were chosen to aggregate binned data. A geometric mean is the n^{th} root of a series of products. For example, if a creek had 8 mean rectified constituent values 1, 3, 2, 1, 4, 4, 3, and 2, the geometric mean will be the 8th root of 576, or 2.213. As presently formulated, the tidal creek index ranges from a low score of 1.0 to a high score of 4.0. Scores are unit-less.

When data sets are uneven the index scores based on geometric means differ from those based on arithmetic means because a reduction in the number of variables does not necessarily reduce all creek scores uniformly. More importantly, the relative rank or order of creeks affected by data loss is conserved. The use of transformed data aggregated by geometric means is the system employed in habitat suitability index (HSI) models. Originally developed by the U.S. Fish and Wildlife Service (1981), the HSI modeling method employs information on a variety of physical, chemical, and biological attributes of a species' habitat. The fitness of the species is defined as unit-less values ranging from 0.0 (completely unfit) to 1.0 (ideally fit) across the range of natural values for each HSI variable, and these are combined as a geometric mean. Data for every variable comprising an HSI index score are not always available so the HSI is similar in this respect to the tidal creek condition index. Even with uneven numbers of variables, the suitability of multiple habitats can be quantified and ranked (USFWS 1981).

2007 Tidal Creek Condition Index Scores—As shown in Figure 3, no index score is presented for Matheny Creek because of unsafe water quality. Creek index scores fell into small groups having relatively low and high values and a larger group with intermediate values. Compared to the cluster analysis dendrogram based on all data for all variables (Figure 2), the sorting of creeks by index scores was conservative. Both approaches identified Whitaker Bayou, and Clower, Matheny, and Hatchett Creeks as a group with similar low resource values, and the two approaches agreed in placing Alligator and Forked Creeks among the group with similar high resource values. This result demonstrates the ability of the index to capture like creek groups and

their relative order as identified through the use of the entire database. Index values ranged from 1.03 to 2.08 or about 35% of the index's theoretical range (1.0 – 4.0). The mean TCCI score was 1.64, or 21% of the index's upper limit. In 2007 at least one creek produced a score of 4.0 for every index variable except *Tagelus* cohort number and maximum oyster height. If perfect scores occurred for most variables in at least one creek then it is reasonable to expect higher creek condition index scores than were seen in 2007.

EVALUATION OF THE TIDAL CREEK CONDITION INDEX

It is generally accepted that the biological condition of a creek is linked in some observable way to the creek's dominant water quality conditions, and to the watershed's dominant landscape and use characteristics. Accordingly, an effort was made to compare the 2007 TCCI to other Sarasota County creek and watershed descriptions: Landscape Development Intensity Index (LDII); Watershed Age of Development (WAD); Hydrological Variability (HYVAR); Antecedent Water Quality (AWQ); Pollution Load Model outputs (PLM), and Benthic Infaunal Community (BENCOM) structure. Descriptions of data sources and methods and how they were compared to the TCCI are described elsewhere (Estevez 2008a; 2007b; Cutler 2007). Results are given in Table 6.

The TCCI was correlated with high significance ($p < 0.01$) to benthic species richness, and significantly ($p < 0.05$) correlated to benthic faunal density, Margalef's Index, and percentage of polychaete taxa. Benthic attributes were in turn correlated with creek sediment and some watershed variables. The significant correlations with species richness included WAD, LDII, % sand, % silt, and PLM for total metal loads. All of the significant correlations between taxa and abundance and WAD, LDII, and PLM metals were negative, suggesting that extent and duration of development in a watershed had a significant negative impact on the benthos of its tidal creek.

That the TCCI had a significant positive correlation with species richness indicates that a high value for the CI is likely to be reflected in a diverse benthic fauna. The TCCI also showed a significant negative correlation to the percentage of polychaetes which, when numerically dominant, signify polluted conditions. The percentage of polychaetes increases when proportions of other major groups such as mollusks and crustaceans decline. Therefore, a higher TCCI value implies that other, pollution-intolerant invertebrate groups have more equitable abundances. Both WAD and LDII had highly significant positive correlations to the presence of PLM total-metal loads, and significant negative correlations to the percentage of polychaete taxa.

DISCUSSION

In the past three decades multiple lines of evidence have shown that freshwater streams are affected by watershed activities and instream habitat alterations (Paul and Meyer 2001; Roth et al. 1996). Work in Charleston Harbor and environs extended that insight to tidal creeks, especially with respect to impervious cover and creek fauna (Holland et al. 2004; Lerberg et al. 2000). During the same period rapid bioassessment of flowing waters has become a mature field and management tool (USEPA 2006; USFWS 1981; Plafkin et al. 1989; Jackson et al. 2000; Van Dolah et al. 1999). Work in Sarasota County represents the first effort to extend rapid bioassessment methods to tidal creek settings. The TCCI associated well with species richness and faunal density of benthic macroinvertebrates, which in turn corresponded significantly with a Landscape Development Intensity Index, Watershed Age of Development Index, and the sum of metals loads computed by a watershed runoff model. In 2007 the TCCI was not significantly correlated with independent variables describing watershed and creek conditions. Drought conditions may have affected comparisons and differences between the response times of infauna and variables employed in the TCCI may have been a factor. Also, the data set was small and imperfect (e.g., by using oyster data collected on different dates, and employing alternative

methods in polluted streams). In addition to wastewater contamination, habitat alteration in some urban streams is extreme, raising the question of whether these systems require treatment as a separate class of tidal creeks.

Sampling and analyses are continuing, with annual surveys in 2008 and 2009 suggesting evidence for changes that reflected the protracted drought. We have observed fidelity in annual creek ranks with respect to the TCCI's ability to identify creek-sets with lowest and highest scores; ranks of creeks intermediate scores vary the most. Also, ongoing monthly sampling in South Creek suggests seasonality in index scores, so work is continuing to determine how synoptic data collected during May and June represent creeks' status and trends. Trends observed in South Creek may reflect the cessation of drought conditions rather than seasonality, so additional data are being collected. Also, closer statistical analyses of existing data are beginning. Other steps that should follow include comparing the TCCI with LDII data proximal to each creek rather than watershed-wide. This approach has been used in Tampa Bay creeks with success (Krebs 2010; Sherwood 2010). Watershed Age of Development data could also be retabulated for a 100 m band of uplands adjoining each creek, and a new generation of pollution load models will provide contemporaneous data to compare to the TCCI. Better empirical descriptions of instream habitat condition should also be sought. Some uniform measures of hydrological variability in the creeks would be extremely interesting to assess but are unlikely in the foreseeable future. Lack of methods to measure or estimate hydrological variability in tidal creeks is a serious hindrance to the detection of creek impairment using the TCCI or any other approach, and should be a priority for future research.

The TCCI brings to bear a set of variables based only in biological attributes. The insight the index provides is, therefore, ecological in nature and, at present, the TCCI is the only ecological reference system available for tidal creeks that is based on rapid-survey methods. There are valid reasons why the biology of a creek is not necessarily tied to its water quality, or the condition of the creek's watershed. This is because numerous factors that are strictly biological in nature, and independent of water quality or watershed condition, are in operation within the creek ecosystem. Examples include recruitment, extirpation, competition, predator-prey cycles, symbioses, and epizootics. Human influences such as harvesting, unintended vessel traffic impacts, and channel maintenance also exert ecological effects independent of creek water quality or watershed condition. Despite potentially confounding effects, benefits of a tidal creek condition index are several. An index provides a basis for generating hypotheses that can be tested to discover the particular causal mechanisms responsible for creek condition. As a rapid-survey method the index offers an ability to respond immediately to a pollution event, and to monitor subsequent conditions at high frequency. In combination with other indicators and metrics, a TCCI can contribute to the production of periodic "report cards" of environmental quality for decision makers, educators, resource enthusiasts, and the general public.

The final session of BASIS 5, "Synthesis of New Knowledge and Management Gaps," identified the value of Tampa Bay's tidal creeks and utility of a bay-wide, synoptic survey to establish baseline creek conditions, perhaps with the assistance of volunteers. The TCCI in conjunction with other rapid bioassessment methods offers a starting point for such efforts.

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Table 1. Number of biological data and cumulative percent of intertidal plus subtidal data by variable for the 2007 Tidal Creek Condition Index. See Estevez (2007a) for definitions of effort for each variable. TAG, *Tagelus*; PERI, periphyton; MOLL, mollusk, SAV, submerged aquatic vegetation; FILAM, filamentous algae; EPI, epiphyton; HT, height.

Variable	Count	Cumulative Percent
1 BURROW COUNT	250	5.65
2 DEAD TAG COHORT NO	238	11.03
3 LIVE TAG COUNT	238	16.41
4 DEAD TAG COUNT	236	21.74
5 PERI PERCENT COVER	220	26.71
6 TAG PERCENT COVER	220	31.68
7 OTHER LIVE MOLL COUNT	208	36.38
8 DRIFT PERCENT COVER	199	40.88
9 BARE NET CRAB	180	44.95
10 BARE NET FISH	180	49.02
11 BARE NET SHRIMP	180	53.08
12 SAV PERCENT COVER	180	57.15
13 BARE NET OTHER	179	61.20
14 OYSTER PERCENT COVER	170	65.04
15 FILAM PERCENT COVER	160	68.66
16 LIVE COHORT NO.	138	71.77
17 DRIFT NET CRAB	97	73.97
18 DRIFT NET FISH	97	76.16
19 DRIFT NET OTHER	97	78.35
20 DRIFT NET SHRIMP	97	80.54
21 LIVE TAG COHORT NO.	90	82.58
22 PERI NET CRAB	70	84.16
23 PERI NET FISH	70	85.74
24 PERI NET OTHER	70	87.32
25 PERI NET SHRIMP	70	88.90
26 SAV EPI COVER	60	90.26
27 SAV EPI LUX	60	91.62
28 SAV NET CRAB	35	92.41
29 SAV NET FISH	35	93.20
30 SAV NET OTHER	35	93.99
31 SAV NET SHRIMP	35	94.78
32 TAG EXTENT	31	95.48
33 PERI EXTENT	22	95.98
34 DRIFT EXTENT	19	96.41
35 SAV EXTENT	18	96.81
36 OYSTER EXTENT	17	97.20
37 FILAM EXTENT	16	97.56
38 SAV SPECIES NUMBER	14	97.88
39 OYSTER SPAT	11	98.12
40 OYSTERS %	11	98.37
41 FILAM NET CRAB	10	98.60
42 FILAM NET FISH	10	98.82
43 FILAM NET OTHER	10	99.05
44 FILAM NET SHRIMP	10	99.28
45 FUNGUS PERCENT COVER	10	99.50
46 OYSTER HT.	10	99.73
47 TUNICATE PERCENT COVER	10	99.95
48 FUNGUS EXTENT	1	99.98
49 TUNICATE EXTENT COVER	1	100.00

Table 2. Florida Department of Environmental Protection organization of biological attributes used in selecting bioassessment variables (McCarron and Frydenborg 2005). Attributes employed by the TCCI are italicized.

INDIVIDUAL CONDITION	TAXONOMIC COMPOSITION	COMMUNITY STRUCTURE	LIFE HISTORY ATTRIBUTES	SYSTEM PROCESSES
Disease	<i>Identity</i>	<i>Taxa Richness</i>	Feeding Groups	Trophic Dynamics
Anomalies	Tolerance	<i>Relative Abundance</i>	Habit	Productivity
Contaminant Levels	Rare or Endangered Taxa	Dominance	<i>Number of Generations</i>	Material Cycles
<i>Longevity/ Death</i>	<i>Indicator Species</i>	<i>Keystone Species</i>		<i>Predation</i>
Metabolic Rate				Recruitment

Table 3. List of 2007 TCCI index variables, gear, and effort (Estevez 2007a). PERI, periphyton; FILA, filamentous algae.

METRIC	GEAR	REPS	SIEVE	STRATA
PERI COVER	1.0 m ² quad	10		Subtidal
FILA COVER	1.0 m ² quad	10		Subtidal
BURROW	.04 m ² quad	10		Subtidal
TAGELUS NUMBER AND COHORTS	10x10 corer*	10	2 mm	Subtidal
OTHER LIVE MOLLUSKS	10x10 corer*	10	2.0 mm and 0.5 mm	Subtidal
BARE BOTTOM FAUNA	Dip net	10	bag mesh	Subtidal
OYSTER LIVE/DEAD	0.25 m ² quad	3		Intertidal
LARGEST LIVE OYSTER		10		Intertidal

Table 4. Relationship of TCCI components to proximate stressors and watershed influences. Other abiotic and biotic stressors (weather; predation) affect these resources but are unrelated to watershed influence.

Level of		Proximate	Watershed
<u>Biological Organization</u>	<u>Index Component</u>	<u>Stressors</u>	<u>Influences</u>
Longevity	Live <i>Tagelus</i> Cohorts	Salinity	Hydrological Modifications
		Chlorophyll a	Nutrients; Residence Time
	Largest Live Oyster	Salinity	Hydrological Modifications
		Dissolved Oxygen	Point and NPS Runoff
Indicator Species	Live <i>Tagelus</i> Density	Chlorophyll a	Nutrients; Residence Time
		Salinity	Hydrological Modifications
	Percent Live Oysters	Chlorophyll a	Nutrients; Residence Time
		Salinity	Hydrological Modifications
		Dissolved Oxygen	Point and NPS Runoff
		Chlorophyll a	Nutrients; Residence Time
Pelagic/Epibenthic Community	Bare Net Catch All	Parasites; Predators	Hydrological Modifications
		Sediment Structure	Sedimentation
		Salinity	Hydrological Modifications
Benthic/Infaunal Community	Burrow Density	Dissolved Oxygen	Point and NPS Runoff
		Sediment Structure	Sedimentation
		Salinity	Hydrological Modifications
	Other Mollusk Density	Dissolved Oxygen	Point and NPS Runoff
		Salinity	Hydrological Modifications
		Dissolved Oxygen	Point and NPS Runoff
Pollution Indicator Species	Periphyton Percent Cover	Chlorophyll a	Nutrients; Residence Time
		Parasites; Predators	Hydrological Modifications
		Light	Sediments, Turbidity, Color
	Filamentous Percent Cover	Nutrients	Point and NPS Runoff
		Salinity	Hydrological Modifications
		Light	Sediments, Turbidity, Color
		Nutrients	Point and NPS Runoff
		Salinity	Hydrological Modifications

Table 5. Bin definitions used to transform 2007 index data, by variable. Note that original field data are used and that the direction of transformed scores for nuisance algae is reversed.

Variable:	Transformation Value			
	1	2	3	4
Burrow Count	0	1-4	5-8	≥ 9
Live <i>Tagelus</i> Density	0	1-2	3-4	≥ 5
Live <i>Tagelus</i> Cohort No.	0	1	2	≥ 3
Other Mollusk Density	0	1-3	4-9	≥ 10
% Live Oyster	0	1-50	51-74	≥ 75
Maximum Oyster Ht. (cm)	<6	6-8	>8-9	≥ 9
Bare Net Catch (All)	0	1-10	11-20	≥ 21
Filamentous Algae % Cover	≥ 66	65-34	33-1	0
Periphyton % Cover	≥ 66	65-34	33-1	0

Table 6. Results of correlation analysis between benthic faunal parameters and select biotic measures (*significant, $P < 0.05$; **highly significant, $P < 0.01$).

Benthic Parameter	Creek Index	Watershed Age	Landscape Development Index	Sediment Parameters				Metals Pb,Cu, Zn, Cd
				Percent Sand	Percent Silt	Percent Moisture	Percent Organic	
No. of Taxa	0.67**	-0.57*	-0.73**	0.50*	-0.52*	0.68**	0.69**	-0.52*
No. of Individuals	0.53*	-0.63**	-0.66**					
Margalef's Index	0.63*		-0.66**					
Metals Pb,Cu,Zn,Cd		0.73**	0.68**					
% Polychaete Taxa	-0.52*	-0.51*	-0.52*					

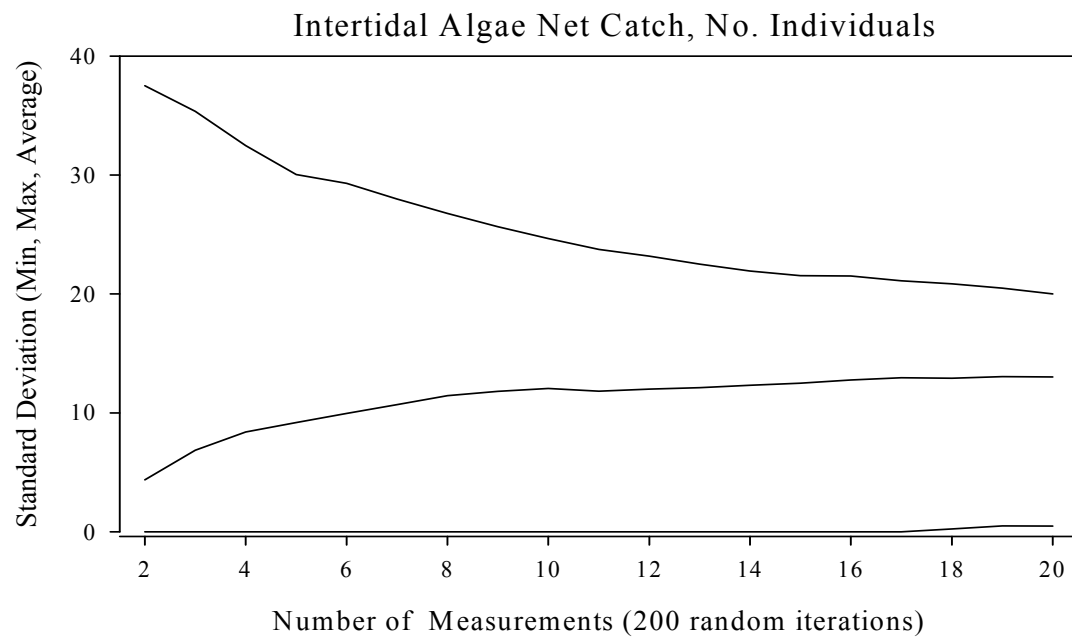
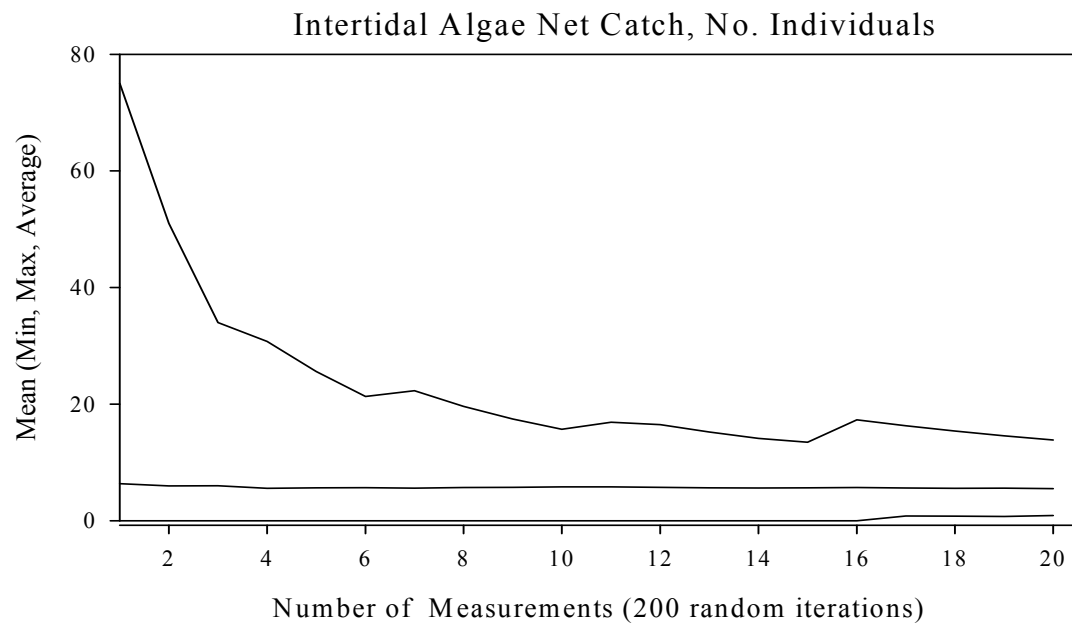


Figure 1. Behavior of minimum, maximum, average mean net catches (upper panel) and minimum, maximum, average standard deviations (lower panel) in relation to sample number (From Estevez 2007a).

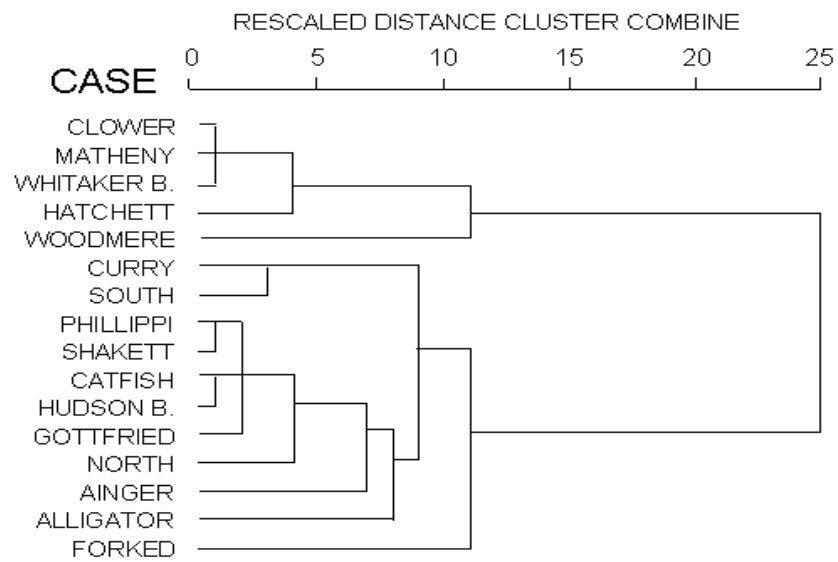


Figure 2. Cluster analysis dendrogram depicting similarity of creeks based on presence-absence transformation of original 2007 data.

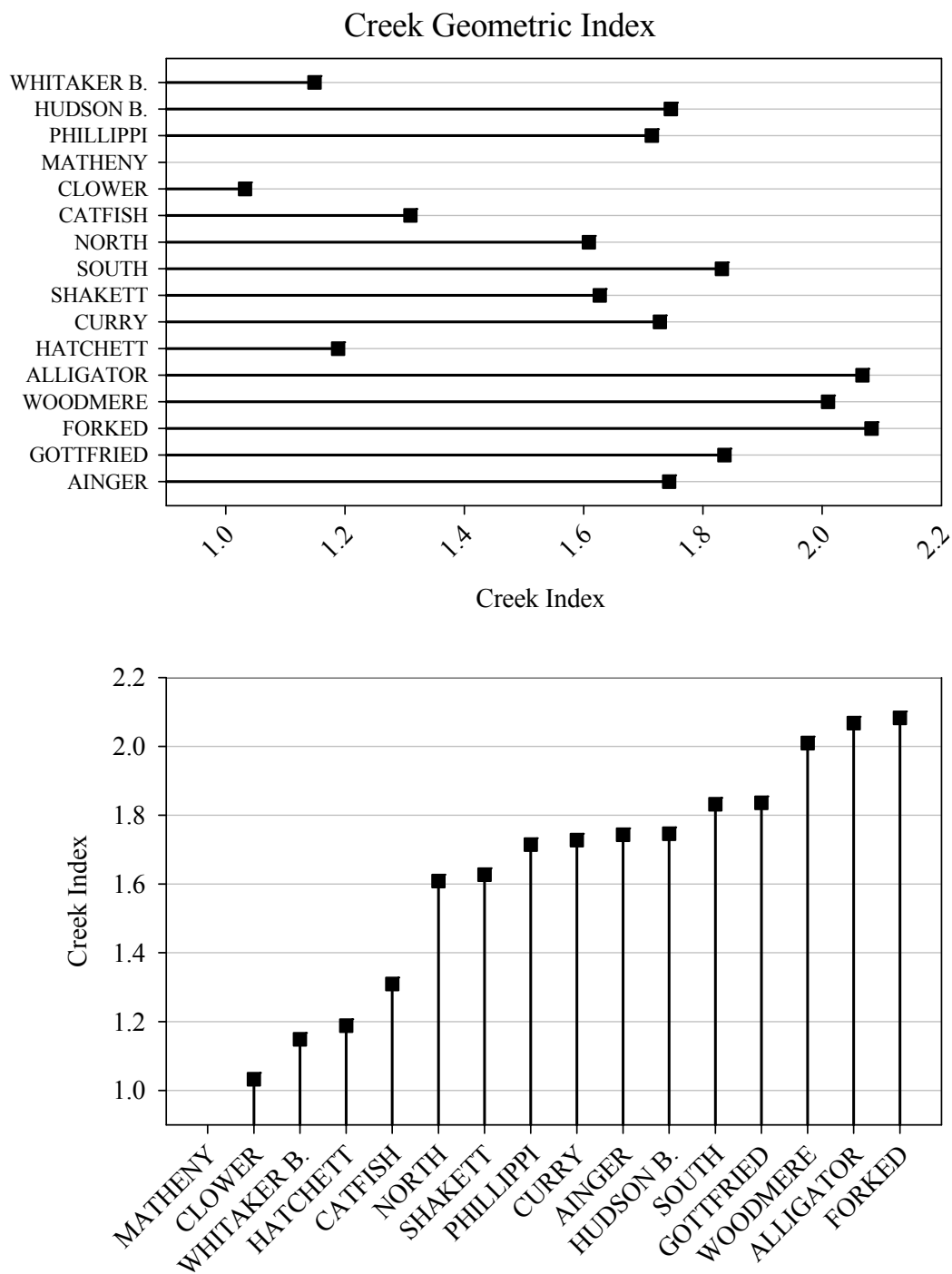


Figure 3. 2007 Tidal Creek Condition Index Scores. Creek scores are shown from north to south in the upper panel and from lowest to highest value in the lower panel.

TIDAL CREEKS AS NEKTON HABITAT IN THE TAMPA BAY ESTUARY

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ABSTRACT

Creeks, streams, canals and other backwater areas (collectively called tidal creeks) that are influenced by the tides and flow into open estuaries or tidal rivers are numerous (>100) in the Tampa Bay estuary, susceptible to change brought about by human activity, and poorly understood in terms of their role in nekton production. Since 2005, the Fisheries-Independent Monitoring program of the Florida Fish and Wildlife Conservation Commission's Fish and Wildlife Research Institute has been involved in several multiagency studies of tidal creeks in the Tampa Bay estuary. During these studies, nekton were sampled monthly in 15 tidal creeks and their receiving water bodies and semiannually (May and September/October) in another 25 tidal creeks. Sampling was conducted with a 9.1-m raft seine. Nekton were identified, counted, and measured; a subsample was returned to the laboratory for analysis of diet and stable isotopes. These samples were collected to 1) assess the importance of tidal creeks to nekton; 2) evaluate the effects of habitat parameters on nekton resources in tidal creeks; 3) determine effects of food and sources of food production on nekton resources in these habitats; 4) identify potential impacts of physical, chemical and biological factors on habitat condition and trophic pathways in tidal creeks; and 5) identify potential impacts of non-native taxa on native nekton taxa. The results indicate that: 1) nekton communities appear to differ as a function of tidal creek morphology and geographic location; 2) the common snook is a prominent species that uses tidal creeks as a nursery habitat; and 3) non-native species such as the pike killifish have the potential to alter native nekton assemblages.

INTRODUCTION

Tidal streams, creeks, canals and other backwater areas form important habitats in estuarine ecosystems but have not been extensively studied and may have physical, chemical, and biological properties that are distinct from other estuarine habitats (Mallin and Lewitus 2004; Tuckey and Dehaven 2006). Anthropogenic alteration of uplands for development has been shown to have negative relationships with nekton abundance in tidal tributaries in South Carolina (Holland et al. 2004). Bilkovic and Roggero (2008) have reported on a negative relationship between habitat alteration and a multimetric index of biological integrity in the James River, Virginia. In the Bahamas, Rypel and Layman (2008) found that gray snapper (*Lutjanus griseus*) collected in mangrove creeks that had reduced hydrological connectivity to the open estuary had more sporadic recruitment, had decreased physical condition, had slower growth rates, and had higher parasite loads than did gray snapper collected in creeks that had better hydrological connectivity to the open estuary. Similarly, secondary production of several grunt and snapper species were significantly reduced in creeks that were not directly connected to the open estuary (Valentine-Rose et al. 2007).

Many studies on nekton communities in Tampa Bay's tidal creeks have concentrated on specific taxa, such as the common snook (*Centropomus undecimalis*) (McMichael et al. 1989; Edwards 1991), red drum (*Sciaenops ocellatus*) (Peters and McMichael 1987), and black drum (*Pogonias cromis*) (Peters and McMichael 1990). Other studies of Tampa Bay have been relatively limited in geographic coverage (TBRPC 1986; Hooper et al. 1990; Haddad et al. 1992; Reynolds, Smith and Hills, Inc. and Lewis Environmental Services 1993) or have documented the effects of habitat restoration efforts (Vose and Bell 1994; Motta 1995; Kurz et al. 1998; Bahk 1999). Recently, researchers at the U.S. Geological Survey, St. Petersburg have used block-net seines to assess

nekton communities in tidal creeks and mosquito ditches at fixed stations within the Tampa Bay estuary (Yeager et al. 2007; Krebs et al. 2007).

Since 2005, the Fisheries-Independent Monitoring (FIM) program at the Florida Fish and Wildlife Conservation Commission's (FWC) Fish and Wildlife Research Institute (FWRI) has been involved in several multi-agency studies of tidally influenced creeks, streams, and canals (collectively called tidal creeks) in the Tampa Bay estuary. Almost 40 tidal creeks and their receiving water bodies were sampled between 2005 and 2007. These collections were made to 1) assess the importance of tidal creeks to nekton; 2) evaluate the effects of habitat parameters on nekton resources in tidal creeks; 3) determine the effects of food and the sources of food production on nekton in these habitats; 4) identify potential impacts of physical, chemical, and biological factors on habitat condition and trophic pathways in tidal creeks; and 5) identify potential impacts of non-native taxa on native nekton taxa.

METHODS

Studies

Manatee River/Gamble Creek (SWFWMD-TT)—Monthly sampling was conducted within Gamble Creek and the Manatee River from November 2005 through July 2006. Nekton sampling was supported as part of a Southwest Florida Water Management District (SWFWMD) study designed to develop data for establishing minimum flows (Florida Statutes 373.042) for the Manatee River and Gamble Creek (Figure 1). Sampling sites were selected based upon a stratified-random sampling design.

Tampa Bay Estuary Program Tidal Tributary Initiative (TBEP-TTI)—Monthly sampling was conducted from January through December 2006 in nine creek systems located in four watersheds (Feather Sound, Alafia River, Little Manatee River, and Terra Ceia Bay) that discharge into Tampa Bay. Feather Sound was represented by one creek (Grassy Creek), Alafia River by four creeks ("Question Mark", "Riverview West", Rice, and "Dog Leg" creeks), Little Manatee River by two creeks (Curiosity and Wildcat creeks), and Terra Ceia Bay by two creeks (Frog and McMullen creeks) (Figure 1). Creek names in quotations indicate that the creek did not have an official name and that the indicate name was assigned by the FIM program for the purposes of the study. The FIM program collected the nekton in this multiagency and multidiscipline study that was funded by a grant from the Pinellas County Environmental Fund (PCEF). Sampling sites in each creek were selected based upon a stratified-random sampling design.

State Wildlife Grant (SWG)—This study had two main components: a monthly sampling of nekton in five creeks and backwaters (SWG-SRS) and a seasonal inventory of nekton in 20 creeks and backwaters (SWG-TS). Monthly samples were collected from January through December 2007 in Andrews Creek, Cockroach Creek, Peanut Lake, "Cute Creek", and Wares Creek (Figure 1). Seasonal samples were conducted during May and September/October 2007 (Figure 1). Creek names in quotations indicate that the creek did not have an official name and that the indicate name was assigned by the FIM program for the purposes of the study. This study was funded by the U.S. Fish and Wildlife Service's State Wildlife Grants Program through the FWC's Wildlife Legacy Initiative. Sites for the monthly sampling were selected based upon a stratified-random sampling design while those for the seasonal sampling were selected based upon accessibility of creeks from land-based access points, typically a road, bridge or culvert, in each system.

FIM program (FIM-Rivers)—The FIM program, which monitors nekton throughout Florida, has monitored Tampa Bay since 1989. The program collects nekton samples monthly to assess abundance trends for resource species such as snook, spotted seatrout (*Cynoscion nebulosus*), and red drum. Data analyzed were those collected in 2006 and 2007 from Tampa Bay's tidal rivers (Hillsborough, Alafia, Little Manatee, Manatee, and Braden rivers). The FIM program uses a stratified-random sampling design to select sampling sites.

Survey methods

Sampling method—During each of the tidal tributary studies (SWFWMD-TT, TBEP-TTI, and SWG) nekton were collected along shorelines with a 9.1-m, 3.2-mm-stretched-mesh, and center-bag seine. To set the seine, one crewmember held an end of the seine against the shoreline while a second crewmember walked in a shallow arc along the shore while deploying the seine from a floating raft. The seine was then hauled ashore from both ends, the lead line kept as close as possible to the bottom, forcing the catch into the bag. Each 9.1-m-seine set sampled an area of approximately 10.1 m² from the shoreline to approximately 1.5 m from shore.

The FIM staff deployed a 21.3-m, 3.2-mm-stretched-mesh, center-bag seine from the back of a boat to collect samples from tidal rivers. Procedures for deploying and recovering the gear were similar to those followed for the 9.1-m seine. Each 21.3-m seine set sampled an area of approximately 68 m² from the shoreline out to approximately 2.5 m away from shore.

Sample processing—Fish and select crustaceans (nekton) collected were identified to the lowest practical taxon, usually species, in each of the studies (SWFWMD-TT, TBEP-TTI, SWG, and FIM-Rivers). Because some species freely hybridize and because it is extremely difficult to identify smaller individuals of some species, members of several abundant species complexes were not identified to species; these included menhaden (*Brevoortia* spp.), silversides (*Menidia* spp.), mojarra (*Eucinostomus* spp.) of < 40 mm standard length (SL), gobies (*Gobiosoma* spp.) of < 20 mm SL, and tilapia (*Oreochromis*, *Sarotherodon*, and *Tilapia* spp.) of < 40mm SL. Grass shrimp (*Palaemonetes* spp.) were identified to species except in the SWG seasonal study, in which they were only identified to genus, and the FIM-Rivers study, in which they were not identified or counted. From each haul, a maximum of 20 individuals per species were measured to the nearest millimeter.

Data Analysis

Abundance comparisons—Abundance data were standardized to 100 m² prior to analysis (# collected/area sampled x 100). Geometric mean abundance $\pm$ 95% confidence limits (Sokal and Rohlf 1981) were calculated for each taxon collected during each study, and taxa were ranked from greatest to least abundant. Monthly length-frequency distributions were plotted for common snook and pike killifish (*Belonesox belizanus*) to identify sizes at which, and months during which, each species was effectively captured by the sampling gear. Analysis of variance (ANOVA, $P < 0.05$; Sokal and Rohlf 1981) was used to determine differences in abundance between water bodies for common snook and pike killifish.

Community analysis—Differences in community structure between systems were investigated using Analysis of Similarities (ANOSIM of Bray-Curtis [1957] similarities; Clarke 1993), and pairwise differences were depicted using Nonmetric Multidimensional Scaling (MDS; Clarke 1993). Data were $\ln(x+1)$ -transformed before analysis, and a dummy species with an abundance of one was added to all collections in order to retain in the analysis samples that did not collect any nekton (Clarke and Gorley 2006). Species characterizing each system were determined with two-way Similarity Percentages Analyses (SIMPER; Clarke 1993).

RESULTS AND DISCUSSION

Overall catch summary—Almost 2200 samples were collected with 9.1-m seines from tidal creeks during the four tidal creek studies (TBEP-TTI, 828 hauls; SWFWMD-TT, 420 hauls; SWG-SRS, 824 hauls; SWG-TS, 125 hauls). During 2006 and 2007, 1099 hauls were collected in tidal rivers (Hillsborough, Alafia, Little Manatee, Manatee, and Braden rivers) during FIM program sampling with 21.3-m seines. Fewer nekton taxa (38, excluding individuals that were not identified to species) were collected during the SWG-TS study than in the other three tidal creek studies (52–56), but this study also had the fewest number of hauls. In contrast, almost twice as many nekton taxa (107) were collected during FIM sampling of tidal rivers between 2006 and 2007 as in the Tampa Bay Estuary Program Tidal Tributary Initiative (56, TBEP-TTI).

The abundant taxa collected in each of the tidal creek studies varied greatly, but a transient species, *Anchoa mitchilli*, ranked among the top three abundant taxa in all tidal creek studies and was the most abundant taxon collected in tidal rivers during FIM program sampling (Table 1). Grass shrimp (TBEP-TTI, SWFWMD-TT, and SWG-SRS) and rainwater killifish (*Lucania parva*) (TBEP-TTI, SWG-SRS, and SWG-TS) were among the top three abundant taxa in three of the four tidal creek studies but were either not worked up (grass shrimp) or were ranked lower (rainwater killifish) during FIM program tidal river sampling. The striped mullet, an economically-important species, was in the top 10 abundant taxa in three of the tidal creek studies and in FIM program sampling (FIM-Rivers); the seasonal inventory (SWG-TS) occurred during months in which striped mullet would not normally have made up a large portion of the catch (May and September/October). Common snook, another economically-important taxon, ranked in the top 20 taxa in three of the tidal creek studies but ranked 35 in both the SWFWMD-TT study and the FIM program sampling in tidal rivers (FIM-Rivers).

Common snook—Tidal rivers and tributaries provide critical nursery habitat for juvenile common snook, but abundance varied significantly between each of the studied creeks and the FIM-River collections. Common snook were significantly more abundant in Wildcat Creek than in any of the other creeks studied in the TBEP-TTI and the FIM-Rivers (Figure 2a; ANOVA, $P < 0.05$; Tukey's post hoc-test). This comparison shows that in 2006 there was 95% confidence that the average abundance of juvenile (<71mm SL) common snook collected between August and December within Wildcat Creek was approximately 10 to 13 times as great as the average abundance in the tidal rivers sampled by the FIM program in 2006 (FIM-Rivers). Similarly, the abundance of juvenile common snook in "Dreadlock" Creek during September and October 2007 was significantly greater (28 to 280 times) than their average abundance in the FIM-Rivers (Figure 2b; ANOVA, $P < 0.05$; Tukey's post hoc-test).

The geographic location of different backwaters may influence the abundance of juvenile snook. Common snook was abundant in Andrews Creek, a system at the inland terminus of Cockroach Bay as well as in the pond like terminus of "Dreadlock" Creek. The greatest abundance observed in the TBEP-TTI study was in Wildcat Creek, a system at the upstream terminus of Hayes Bayou. These patterns could indicate that larval settlement is more pronounced toward regions at the termini of bays. Additionally, Andrews and "Dreadlock" creeks are likely to have minimal freshwater inflow, so that two-layered estuarine circulation may not be as important as flood-tide influx (Peters et al. 1998) for larval movement into these areas. This information has significant implications for those charged with selecting areas for restoration or mitigation.

Common snook require a range of habitat types during their life cycle (Lewis et al. 1985). It is essential that the full range of these habitats be available for sustaining this species; the reduction in any single habitat type in the common snook's life cycle is likely to negatively impact abundance at later life-history stages. Preserving, restoring, and creating juvenile habitat is likely

to be at least as effective as traditional fishery management methods (size limits, bag limits) in increasing the size of the common snook population (Lewis and Robison 1996). Tidal creeks are important juvenile snook habitat that should be preserved and restored. Additionally, monitoring the abundance of common snook within these habitats would generate valuable indices for estimating the health of this recreationally important species and would provide data for understanding the variability in recruitment success between creeks.

Non-native taxa—Non-native taxa made up about 0.05% (average abundance, 0.93 individuals per 100m²; Figure 3) of the total nekton catch in 21.3-m seine collections in the tidal rivers sampled by the FIM program during 2006 and 2007. During this same period, non-native taxa comprised 0.08% (SWFWMD-TT; 0.87 individuals per 100m²) to 0.62% (SWG-TS; 12.44 nekton per 100m²) of the catch in tidal creeks, indicating a tendency of non-native taxa to have greater abundance and represent a larger percentage of the nekton population in the tidal creeks than in the tidal rivers. In three of the tidal creek studies (TBEP-TTI, SWFWMD-TT, and SWG-SRS), as well as in the FIM-Rivers, the most abundant non-native taxa were the tilapias. The pike killifish (*B. belizanus*) was the second most abundant non-native taxon collected during the TBEP-TTI, SWG-SRS, and in the FIM-Rivers.

The pike killifish, like the common snook, is a predator of small fish and could be an important competitor with, and predator of, juvenile common snook in Tampa Bay's tidal creeks. Sampling in Tampa Bay by the FIM program before 2006 indicated that pike killifish occurred in relatively low numbers (< 200 individuals collected since 1989) in the Little Manatee and Alafia rivers. Data from the tidal tributary studies show that the pike killifish exists in several other watersheds that discharge along Tampa Bay's eastern shore and in at least one creek (Salt Creek) on the western shore (Figures 1, 4a and 4b). These same data indicate that pike killifish abundance can be significantly higher in creeks than in the tidal rivers sampled by the FIM program.

Community structure—The nekton community structure varied considerably between creeks during the tidal tributary studies conducted in 2005/2006 (SWFWMD-TT and TBEP-TTI). Although both are located in the Little Manatee River watershed, Wildcat and Curiosity creeks had very different nekton communities (Figure 5a). The assemblage in Wildcat Creek was typified by high catches of hog chokers (*Trinectes maculatus*), clown gobies (*Microgobius gulosus*), and bay anchovies (*Anchoa mitchilli*) and the absence of mosquito fish (*Gambusia holbrooki*). The communities of Curiosity and Gamble creeks (Group C), which grouped at 76% Bray-Curtis similarity, were characterized by large numbers of mosquito fish, sailfin mollies (*Poecilia latipinna*) and rainwater killifish, and low abundance of bay anchovies. Three of the four creeks in the Alafia River watershed grouped together (Group A: Rice, "Dog Leg" and "Riverview West") based upon relatively low overall abundance and high variability. The nekton community in the remaining four creeks (Group B: Grassy, Frog, McMullen, and "Question Mark") had much greater abundance of grass shrimp.

As with the 2005/2006 studies, the nekton assemblages varied between creeks during the 2007 studies (SWG-SRS and SWG-TS). The September/October seasonal survey results are presented here (Figure 5b), but similar between-creek variability was found in the monthly and May seasonal surveys. At the 25% Bray-Curtis similarity level, four creek groupings were identified in the nekton community from the September/October seasonal survey. The largest grouping of creeks (Group A) was dominated by creeks located in Pinellas and western Hillsborough counties. This assemblage was typified by rainwater killifish, sailfin molly, eastern mosquitofish, *Menidia* silverside, grass shrimp, sheepshead minnow (*Cyprinodon variegatus*), and Gulf killifish (*Fundulus grandis*). The second largest cluster (Group B) was characterized by pink shrimp (*Farfantepenaeus duorarum*), bay anchovy, *Eucinostomus* mojarra, grass shrimp, common

snook, *Menidia* silverside, striped mojarra (*Eugerres plumieri*), tidewater mojarra (*Eucinostomus harengulus*), blue crab (*Callinectes sapidus*), pike killifish, rainwater killifish, and sailfin molly. The remaining two groupings (C and D) each comprised two creeks, with Group C characterized by hogchoker and grass shrimp and Group D by tidewater mojarra, common snook, and *Menidia* silverside.

CONCLUSIONS

Tidal creeks within Tampa Bay represent unique and important nekton habitats. Patterns of use by individual species and nekton communities varied greatly between creeks, probably in response to habitat differences and location relative to spawning areas. The importance of these areas as nursery habitat for juvenile snook cannot be overstated. Non-native taxa appear to be more prevalent in tidal creeks than in tidal rivers and may have direct (predation) or indirect (competition for prey) interactions with native taxa such as the common snook. Preservation and restoration of tidal creek habitats is extremely important as is the monitoring of the nekton assemblages in these habitats.

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Figure 1. Tidal streams, creeks, and other backwaters sampled by the Florida Fish and Wildlife Research Institute's Fisheries-Independent Monitoring program from 2005 through 2007. Bold font indicates systems sampled monthly during 2007; italic font indicates systems sampled monthly during 2006 (2005–2006 for Gamble Creek); systems sampled seasonally (May and December) have normal font with the circles indicating sampling locations within the system. Names of three creeks have been abbreviated (QC = "Question Mark" Creek, RW = "Riverview West" Creek, and DC = "Dog Leg" Creek). Creek names in quotations indicate that the creek did not have an official name and that the indicate name was assigned by the FIM program for the purposes of the study.

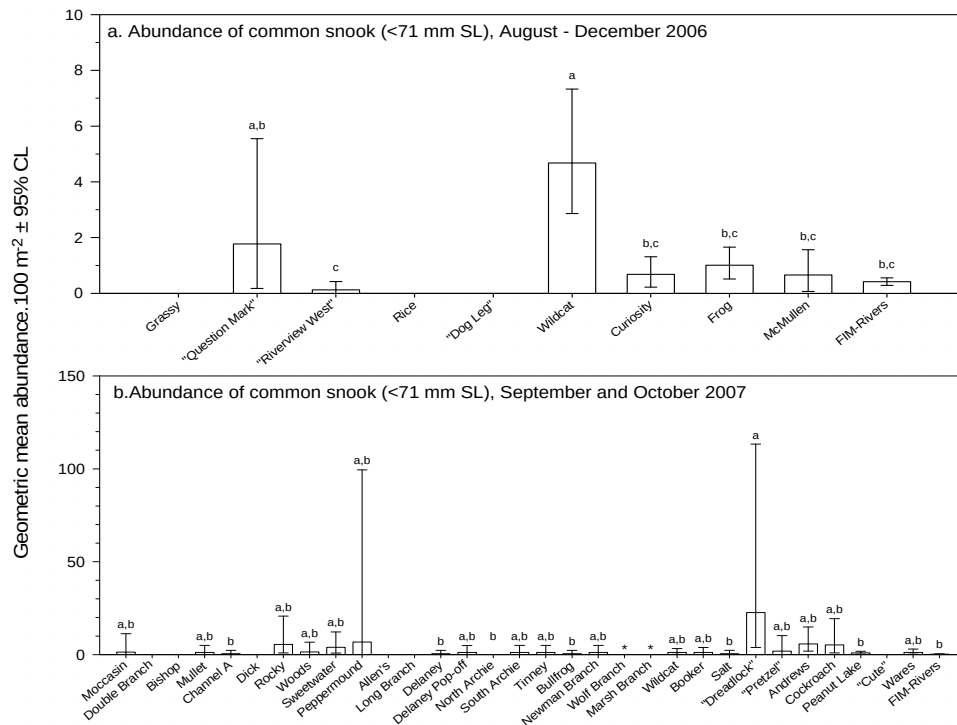


Figure 2. Patterns of abundance for common snook (<71mm standard length) collected (a) from August through December 2006 and (b) during September and October 2007. Samples collected in tidal rivers with 21.3-m seine by the FIM program are represented by FIM-Rivers. All other locations were sampled with 9.1-m seines. Data have been standardized to number of fish per 100m² of bottom area sampled for comparability between studies. Letters above confidence intervals indicate significantly different groupings of creeks (ANOVA, $P < 0.05$, Tukey's post-hoc test: $a > b > c$). Creeks in which common snook were not collected during the analyzed time period were not included in the ANOVA. An asterisk (Figure 2b) indicates that common snook (< 71 mm standard length) were collected from the designated water body but during months other than September and October.

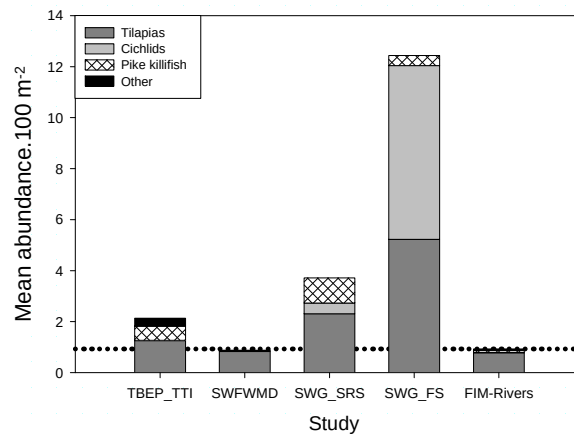


Figure 3. Abundance of non-native taxa in each of the tidal tributary studies and in the FIM-Rivers sampled during 2006 and 2007. Dotted horizontal line represents the abundance of non-native species in FIM river sampling.

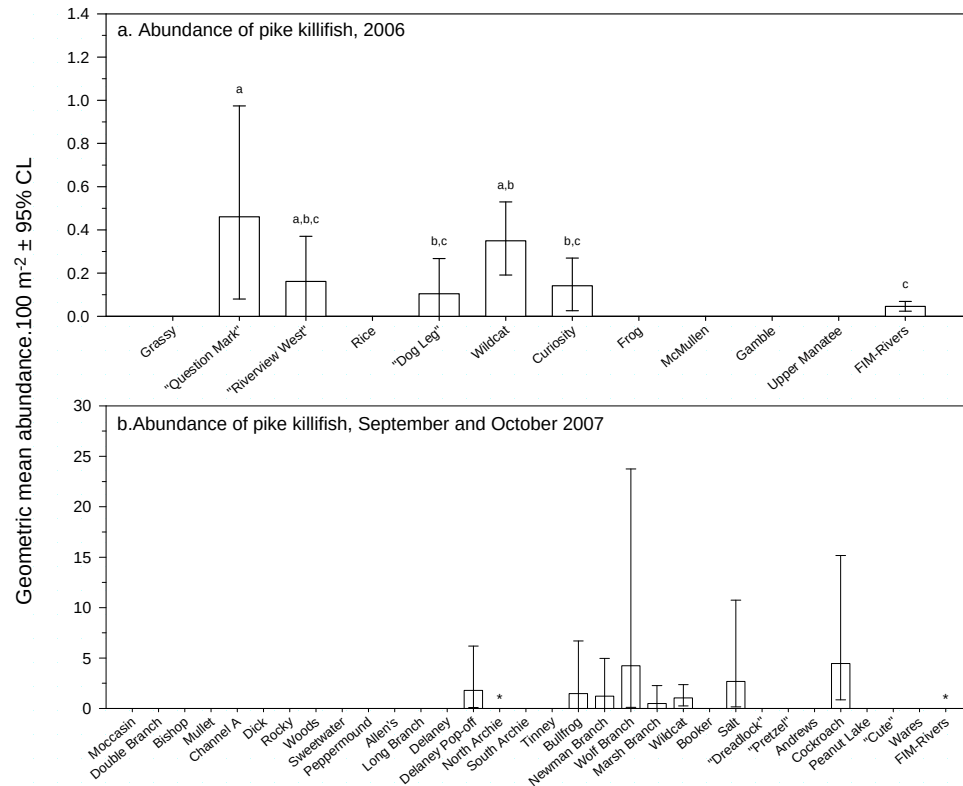


Figure 4. Patterns of abundance for pike killifish collected during (a) 2006 and (b) September and October 2007. Samples collected in tidal rivers with 21.3-m seine by the FIM program are represented by FIM-Rivers. All other locations were sampled with 9.1-m seines. Data have been standardized to number of fish per 100m² of bottom area sampled for comparability between studies. Letters above confidence intervals indicate significantly different groupings of creeks for 2006 (ANOVA, $P < 0.05$, post hoc Tukey test: $a > b > c$); no significant differences were found in 2007. Creeks in which pike killifish were not collected during the indicated time period were not included in the ANOVA. An asterisk (Figure 4b) indicates that pike killifish were collected from the designated water body but during months other than September and October.

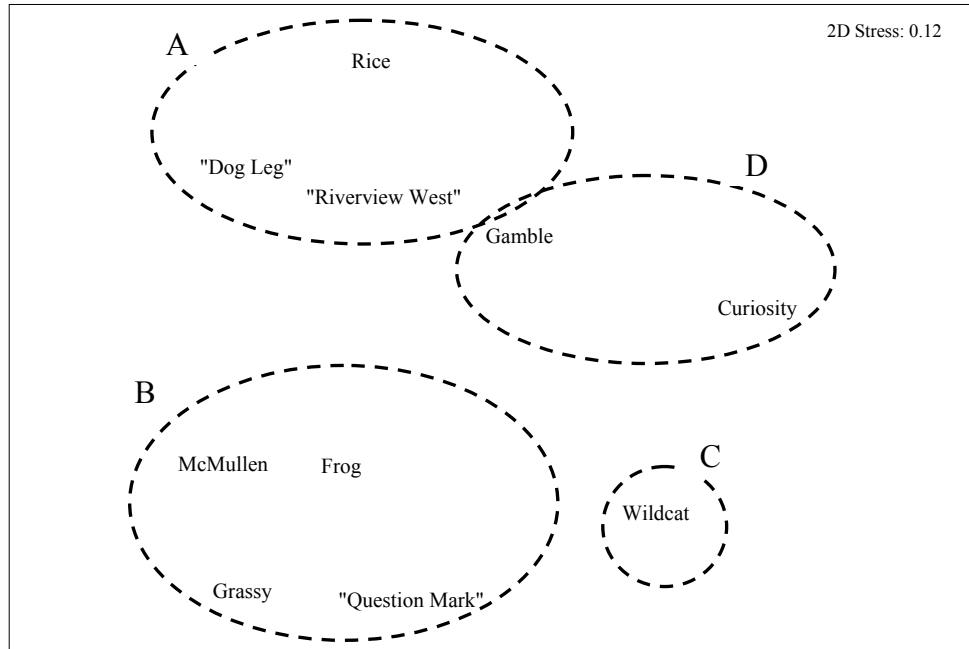


Figure 5a. Nonmetric Multidimensional Scaling plot illustrating differences between the nekton communities in Tampa Bay tidal creeks based on pairwise differences from Analysis of Similarities. Gamble Creek was sampled in 2005 and 2006; all other creeks were sampled in 2006 only. Dashed ovals and letters denote groups of similar systems at 50% Bray-Curtis similarity based on cluster analysis.

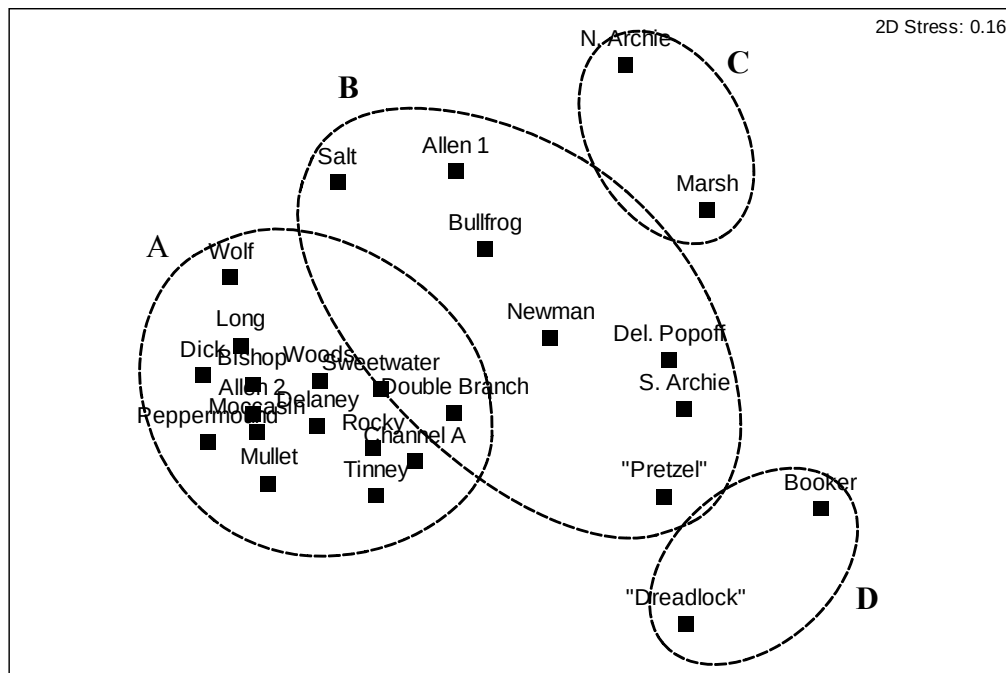


Figure 5b. Nonmetric Multidimensional Scaling plot illustrating differences between the nekton communities of the seasonal survey systems in September/October 2007 (SWG-TS). Dashed ovals and letters indicate groups of similar systems at 25% Bray-Curtis similarity based on cluster analysis.

Table 1. Rank and mean abundance (animals per 100m³) of the abundant taxa collected in the Tampa Bay Estuary Program Tidal Tributary Initiative (TBEP-TTI); Gamble Creek and upper Manatee River study (SWFWMD-TT); State Wildlife Grant monthly (SWG-SRS) and seasonal (SWG-TS) tidal tributary sampling; and FIM program in tidal rivers (FIM-Rivers). Summary is sorted from greatest to least abundance for the PCEF-TTI, and only the top 20 taxa collected in the PCEF-TTI are presented. Data for taxa that ranked in the top 10 of each study are shaded. The eighth ranked taxon in FIM-Rivers, *Harengula jaguana*, was not collected in the TBEP-TTI and is not listed here. Grass shrimp (*Palaemonetes* spp.) were not identified or counted during FIM program sampling in tidal rivers (FIM-Rivers). Taxa are assigned categories (Cat) based upon their use of these habitats (R=resident [found in habitat throughout their lives], T=transient [found in habitat for only part of their lives]). Taxa of economic importance (Eco) are indicated by “X”.

Taxa	Common Name	TBEP-TTI		SWFWMD-TT		SWG-SRS		SWG-TS		FIM-Rivers			
		Rnk	Abund	Rnk	Abund	Rnk	Abund	Rnk	Abund	Rnk	Abund	Cat	Eco
Palaemonetes spp.	Grass shrimp	1	496.2	1	375.1	3	110.9	9	17.8				R
	Rainwater killifish	2	195.9	7	33.4	2	113.7	3	210.5	5	41.9		R
	Bay anchovy	3	176.3	2	285.2	1	187.2	1	1,144.7	1	1,214.0		T
	Menidia spp.	4	141.2	6	44.7	4	63.9	5	89.1	2	286.7		R
Gambusia holbrooki	Eastern mosquitofish	5	123.1	3	154.1	10	19.3	4	102.7	10	11.5		R
Poecilia latipinna	Sailfin molly	6	52.1	5	50.1	5	44.5	2	239.1	14	6.7		R
Mugil cephalus	Striped mullet	7	32.9	8	32.7	6	35.1	32	0.6	4	43.0		T
Trinectes maculatus	Hogchoker	8	22.4	4	87.4	16	3.7	16	4.8	12	8.0		T
Cyprinodon variegatus	Sheepshead minnow	9	21.4	17	3.4	8	26.2	6	39.4	11	9.2		R
Microgobius gulosus	Clown goby	10	20.4	9	21.1	12	5.8	14	5.1	7	16.1		R
Eucinostomus spp.	Eucinostomus mojarra	11	8.7	20	2.6	7	30.5	10	15.4	6	19.7		T
Fundulus grandis	Gulf killifish	12	8.4	14	6.4	9	21.3	12	10.7	17	4.5		R
Floridichthys carpio	Goldspotted killifish	13	6.2	.	.	15	4.0	11	12.0	20	3.5		R
Gobiosoma bosc	Naked goby	14	5.1	10	13.5	25	1.4	24	1.9	19	3.6		R
Fundulus seminolis	Seminole killifish	15	4.2	12	10.1	43	0.2	.	.	23	3.1		R
Eucinostomus harengulus	Tidewater mojarra	16	2.8	24	1.0	11	8.5	8	21.1	9	15.0		R
Centropomus undecimalis	Common snook	17	2.7	35	0.1	14	4.1	19	3.2	35	0.7		T
Lucania goodei	Bluefin killifish	18	2.5	45	0.0	.	.	43	0.1	69	0.0		R
Leiostomus xanthurus	Spot	19	2.4	28	0.6	13	4.2	20	3.2	30	1.1		T
Heterandria formosa	Least killifish	20	2.4	22	1.4	45	0.1	.	.	88	0.0		R

DEFINING FISH NURSERY HABITATS: AN APPLICATION OF OTOLITH ELEMENTAL FINGERPRINTING IN TAMPA BAY, FLORIDA

J.A. Ley, C.C. McIvor, E.B. Peebles & H. Rolls

ABSTRACT

Fishing in Tampa Bay enhances the quality of life of the area's residents and visitors. However, people's desire to settle along the Bay's shorelines and tributaries has been detrimental to the very habitat believed to be crucial to prime target fishery species. Common snook (*Centropomus undecimalis*) and red drum (*Sciaenops ocellatus*) are part of the suite of estuarine fishes that 1) are economically or ecologically prominent, and 2) have complex life cycles involving movement between open coastal waters and estuarine nursery habitats, including nursery habitats that are located within upstream, low-salinity portions of the Bay's tidal tributaries. We are using an emerging analytical technique - elemental fingerprinting of fish otoliths - to determine the degree to which specific estuarine locations contribute to adult fished populations in Tampa Bay. In ongoing monitoring surveys, over 1,000 young-of-the-year common snook and red drum have been collected from selected Tampa Bay tributaries. Using laser ablation-inductively coupled plasma - mass spectrometry (LA-ICP-MS), we are currently processing a subsample of these archived otoliths to identify location-specific fingerprints based on elemental microchemistry. We will then analyze older fish from the local fishery in order to match them to their probable nursery areas, as defined by young-of-the-year otoliths. We expect to find that some particularly favorable nursery locations contribute disproportionately to the fished population. In contrast, other nursery areas may be degraded, or act as "sinks", thereby decreasing their contribution to the fish population. Habitat managers can direct strategic efforts to protect any nursery locations that are found to be of prime importance in contributing to adult stocks.

INTRODUCTION

The overall aim of this paper is to introduce an ongoing study (McIvor et al. unpublished) focused on identifying specific nursery habitats that perform well as sources of recruits to red drum (*Sciaenops ocellatus*) and common snook (*Centropomus undecimalis*) populations of Tampa Bay, based on an analysis of otolith microchemistry. The life history of these species involves shifts in habitat as individuals grow and develop. While both species are highly valued as fishery resources, conservation of stocks involves management measures that both regulate the fishery itself and protect fish habitat - especially for the younger components of the population. Bay-wide evaluation of nursery quality and performance can be used to strategically direct habitat protection efforts. Otolith microchemistry is an emerging analytical technique applicable to the study of variation among nursery habitats and connectivity to adult populations (Elsdon et al. 2008). This technique has been applied in only a few studies of red drum and common snook populations in Florida to date (Patterson et al. 2004; Patterson et al. 2005). However, together with results of our own preliminary analyses, successful evaluation of red drum and common snook nursery performance through application of otolith microchemical techniques seems highly likely. The current paper provides an overview of background information relevant to the ongoing project and how the results may be used in decision-making by resource and land use managers.

Red drum and common snook: Life history and status in Tampa Bay—Red drum and common snook are species that have a great deal in common. Both share a similar life history in that they depend on a variety of estuarine habitats during different phases of their lives. Fishery management concerns are also similar for these species in Florida. In the following paragraphs,

life history and fishery issues of importance are outlined, focusing on the role of nursery conditions and habitat connectivity.

Red drum *Sciaenops ocellatus*—The species *Sciaenops ocellatus* (red drum), one of 270 species in the large fish family Sciaenidae occur in the Atlantic Ocean, Gulf of Mexico and associated estuaries, from Massachusetts, around Florida, and westward to northern Mexico. Intensively studied in Texas, Louisiana, Florida, and North Carolina, red drum populations are known to vary in their basic biology (e.g. growth rates, size at maturity) and in their responses to environmental conditions (e.g. temperature, salinity preferences) throughout their range. Thus, conditions and parameters relevant to the Tampa Bay area population may differ from red drum in other portions of their range. Wherever possible, studies specific to Tampa Bay red drum have been used to summarize life history information in the current paper (Peters and McMichael 1987; Murphy and Taylor 1990; Seyoum et al. 2000; Murphy and Crabtree 2001).

The life cycle of a cohort (or generation) of red drum is illustrated in Figure 1, with components of the diagram referred to in the paragraphs below in bold type. Red drum are broadcast spawners, with sound-producing males aggregating among females adjacent to coastal inlets and passes around the mouth of Tampa Bay during a 2 to 3 month spawning season that peaks in October (**Spawners**). Since eggs sink at salinity levels less than 25 parts per thousand (ppt), conditions for successful egg fertilization and hatching include near full strength sea water salinity (35 ppt) as well as temperatures near 25° C (**Eggs**). The billions of fertilized eggs produced during a single evening are transported by currents through inlets and passes where they hatch in the water column. Pelagic larvae remain in the bay water column consuming copepods and growing rapidly, until they are about 15 days old (**Pelagic Larvae**). By the time red drum settle to the substrate and transform into juvenile fishes, they have travelled through the interior of the Bay to peripheral habitats primarily within low salinity, tidal tributaries (**Post-settlement Juveniles**). For the next 3 months, they occupy quiet backwaters growing rapidly as they consume mysids, amphipods and insects. Red drum occupy these habitats during the coldest months of the year (November, December, January), when water temperatures may drop (typically to as low as 12° C). In the next phase (**Larger Juveniles**), growth is most rapid, approximating 1 mm of new growth per day until a standard length of 300 mm is reached. These larger juveniles form aggregations around oyster bars and over seagrass beds. They are capable of surviving in a range of environmental conditions and of consuming a variety of foods. The larger juveniles are targeted by the fishery (see below), which is primarily active within the Bay itself. Although individuals vary in age-at-maturity, generally, **Adult Males** mature at 2 years old (460 mm SL – standard length), whereas **Adult Females** mature at age 5 (730 mm SL). As they approach maturity, red drum move out of the Bay, forming aggregations in nearshore waters of the Gulf of Mexico. They remain in the Gulf, growing more slowly than in the bay, until the end of their lives which can be up to 35 years.

The number of young-of-the-year (YOY) or age-0 red drum recruiting to nursery habitats tends to vary dramatically from year to year (Figure 2a). For example, considering the cohort recruiting during the 1995 season, Florida Fish and Wildlife Conservation Commission's Fish & Wildlife Research Institute (FWRI) monitoring teams found that catch rates of YOY red drum were very high in estuaries along the Gulf coast of Florida relative to other years (Murphy and Munyandorero 2009). However, perhaps surprisingly, this 1995 cohort was not particularly abundant in subsequent surveys (e.g. 2 years later, see Figure 2b). Thus, while some years may have strong recruitment levels as reflected in catch-rates of YOY, red drum may not survive in adequate numbers to produce strong subsequent year classes. The abundance of fish available to the fishery in Tampa Bay appears to depend not only on the number of recruits that arrive in the nursery habitat, but also on their survival and growth there (i.e. how well they were able to find

food, shelter, and benign conditions in their nursery habitats). Thus, the consistent availability of high quality nursery habitat is critical to recruitment in Bay populations.

Since 1988, the commercial fishery has been closed and only recreational fishing has been permitted for red drum in Florida. The number of recreational anglers has doubled over this time period, reaching 2 million by 2007 (Murphy and Munyandorero 2009). Of the 3 million fish caught along the Gulf Coast in 2007, 85% were released alive, but an estimated 5% of these released fish were believed to have died after they had been returned to the water. Thus, about one-half million red drum were killed in this fishery in 2007. About half of the harvested fish would have been age 2, one-third age 3, and about 10% age 4. Thus, the red drum fishery along Florida's Gulf Coast is chiefly based on juvenile males (up to age 4) caught in estuaries. Regulations in Florida consist of slot size limits, which protect both smaller fish (minimum legal size, 378 mm SL, or about 1.5 years old), and older spawners (maximum legal size, 571 mm SL, or about 3 years old). Although no closed seasons currently exist, anglers are limited to one fish per day.

The criteria established to evaluate the status of Gulf Coast red drum populations calls for maintaining a level of 40% escapement from all sources of death (i.e. due to fishing and natural causes). The most recent stock assessment calculated that escapement is currently at 37%, and is predicted to decline over the next 5 years due to increased fishing pressure. Thus, despite the establishment of stringent regulations, the stock remains overfished (Murphy and Munyandorero 2009).

Common snook (*Centropomus undecimalis*)—Common snook (*Centropomus undecimalis*), one of only 22 species in the fish family Centropomidae have been known to occur from Pamlico Sound, North Carolina southward to Rio de Janeiro, Brazil. However, in Florida, common snook are usually found from Cape Canaveral southward around the Florida Keys to north of Tampa Bay. Although they are rare off islands such as the Bahamas which lack permanent freshwater streams connecting with the sea, common snook do occur off islands with high topography, including Hispaniola, Tobago, Cuba, Grand Cayman and Puerto Rico (Rivas 1986).

Nine studies of common snook were used to develop a life history diagram specific for the Tampa Bay area (McMichael et al. 1989; Howells et al. 1990; Peters et al. 1998; Taylor et al. 1998, 2000; Blewett et al. 2006, 2009; Stevens et al. 2007; Winner et al. 2010). Spawning sites are known to include passes from the Gulf (e.g. John's Pass, Hurricane Pass), channels in the Bay (e.g. Mermaid Pass at Weedon Island), channels around nearshore islands (e.g. Rattlesnake Key), and along beaches (e.g. Honeymoon Island) in the Tampa Bay area (Taylor et al. 1998) (**Spawners** in Figure 3). Adult common snook migrate to spawning sites in the summer from more dispersed habitats occupied during the remainder of the year. Around Tampa Bay, peak spawning activity is during June and July. Within large spawning aggregations, individual females release eggs multiple times throughout this period (**Eggs**). Eggs hatch within one day of fertilization, with larvae developing in the water column for about 15 days while moving with currents from the spawning sites to estuarine nursery grounds (**Pelagic larvae**). Larvae metamorphose into juveniles within tidal tributary systems, in ponds and waterways bordered by red mangroves. For the next month they live and feed in the water column along these fringing mangrove shorelines (**Pelagic juveniles**). Growth rates increase as the juveniles develop the morphological features required to consume larger demersal prey such as shrimp and killifish (**Small demersal juveniles**). Individuals in this phase usually occupy upstream portions of tidal tributaries. From six months to 2 years old, the **Larger demersal juveniles** occupy a broad range of habitats, including tidal tributary ponds, open bay shorelines and seagrass beds. Common snook begin to mature at age 2, but since they are protandrous hermaphrodites, individuals first

develop as males (**Adult males**). By age 5, an estimated 50% of individuals have developed into females (**Adult females**), but there is considerable variation in this pattern. As they become larger, common snook occupy a broad range of habitats, including tidal tributary systems, freshwater rivers and creeks, and offshore reefs. However, even large adults live in close proximity to sizeable structure with submersed features, primarily mangroves.

Like red drum, the abundance of YOY common snook in nursery habitats of Tampa Bay varies dramatically from year to year. For example, in 1999, high abundances were captured in small mesh nets by FWRI monitoring teams (Figure 4a). However, three years later, this cohort was not distinctive relative to other cohorts in the monitoring surveys (Figure 4b). As with red drum, the abundance of common snook available to the fishery depends not only on the number of recruits to the nursery ground, but also on their ability to survive there. Thus, consistently available high-quality nursery habitats are critical to sustaining populations in the Bay.

Since 1957, the commercial snook fishery in Florida has been closed, and only recreational fishing has been permitted. The number of Gulf Coast recreational anglers has doubled since the 1980s, reaching 1.5 million by 2004 (Muller and Taylor 2006). Of the 2 million fish caught along the Gulf Coast in 2004, 97% were released alive, but an estimated 2% of these released fish were believed to have died after they had been returned to the water. Thus, an estimated 115,000 snook were killed in the Gulf Coast fishery during 2004. About one-quarter of the harvested fish would have been age 5, one-third age 6, and about 20% age 7. Regulations for this fishery consist of slot size limits, which protect both smaller fish (minimum legal size, 587 mm SL or about 5.5 years old), and older spawners (maximum legal size, 696 mm SL, or about 8.6 years old). The harvest of snook is only permitted during five months of the year (March, April, September, October and November). Anglers may keep one fish per day but must purchase a special snook stamp in addition to the cost of their saltwater fishing license.

An assessment of the status of the Gulf Coast snook fishery has most recently calculated that at 25% spawning potential ratio (SPR), the stocks are overfished. Fisheries management officials continuously institute measures in an attempt to increase SPR to a more sustainable level of 40%, primarily by regulating the fishery. However, like red drum, despite the establishment of stringent regulations, the populations along the Gulf coast are not at levels considered to be advantageous for management purposes (Muller and Taylor 2006).

Summary of similar concerns: red drum and common snook—Both red drum and common snook are valuable assets to the Tampa Bay community, supporting recreational fisheries that enhance the quality of life of residents and visitors. The number of anglers participating in these fisheries is increasing, and despite the fact that strong fishery management regulations have been established, the populations of both species on Florida's Gulf Coast are currently overfished. Habitat degradation is also of concern for both species, since many of the tidal tributaries they use as nurseries have been altered. Evidence from fishery-independent monitoring surveys in nursery habitats indicates that while recruitment of YOY is high in some years, juveniles may not survive in sufficient numbers to produce strong year-classes that would contribute to sustaining the fished populations in the Bay. Thus, the picture in Tampa Bay, as elsewhere is complex – the importance of nursery grounds is not simply a function of the density of newly-settled fish, but also the survival and connectivity of the nursery to adult habitats (Chittaro et al. 2009).

Evaluation of nursery performance—Clearly, processes that contribute to juvenile survival in their nurseries are critical to maintaining the stocks of red drum and common snook. While the ecology of tidal tributaries around Tampa Bay is now better understood due to intensive investigations in recent years (e.g. Greenwood et al. 2008a; b), there is still a critical need to

assess how and why they vary in the production of species of fishery interest such as red drum and common snook. Addressing this need involves integrating aspects of multiple disciplines in problem-focused efforts, employing a variety of scientific tools and analyses. In our approach under the current study, we ask the question: *Can the relative contributions of individual nursery areas to the fished components in the Bay be determined for a given year class?* In other words, *do some nurseries perform better than others in particular years?*

Nursery performance can be defined as the demonstration that a specific nursery area produces a certain relative quantity of juveniles, and the degree to which those fish recruit to adult populations (Dahlgren et al. 2006). In the case of red drum and common snook and their use of tidal tributary habitats, nursery performance encompasses two critical questions: (1) does a particular tidal tributary provide adequate features for juveniles to grow and survive; and, (2) do many of the young produced there eventually move into Tampa Bay?

To address these questions, it would be useful to have a tool that can track where an individual fish captured in the Bay had lived during juvenile stages of its life. The current study employs otolith microchemistry as such a tool, attempting to establish the specific nursery habitats used by individual fish during their lives. By analyzing a number of individuals from around the Bay, trends in the spatial distribution of specific nursery habitats are expected to emerge. Some tributaries may contribute proportionally large numbers of recruits, acting as effective sources of common snook and red drum to the Bay. Other tributaries may be too degraded or transformed to perform well as nursery habitats, and therefore act as sinks relative to the young that may settle there. These distribution patterns may change from year to year, as environmental conditions and other factors vary. This information can provide insights into the habitat features and conditions necessary to support successful nurseries for red drum and common snook in Tampa Bay and into how essential processes must be managed to ensure continued productivity.

An overview: applied otolith microchemistry—An otolith is a very hard calcium carbonate (primarily aragonite) structure that is a key component of the hearing and equilibrium system of bony fishes (i.e. not elasmobranchs). Three pairs of otoliths occur in the head of a fish, each contained within a fluid-filled otic capsule. The largest otoliths, the sagittae, are often used to determine the age of an individual by examining the pattern of rings laid down as the fish grew. Some of the elements that comprise an individual growth ring of the otolith are derived from the water mass the fish was living in when the otolith was laid down (Campana 1999). Elements in the water mass are permanently stored in the growth rings, and can be used to identify certain characteristics of the water mass history (Elsdon et al. 2008). Since individual aquatic systems often have water mass characteristics that are distinctive, a particular series of otolith rings may reflect these distinctive features. Thus, the microchemical composition of the otolith (i.e. the “fingerprint”) may be determined using inductively-coupled mass spectrometry (ICP-MS) of laser-ablated samples from otolith locations representing known ages, revealing the fish’s history occupying different water masses (Milton et al. 2008). Data derived from the microchemical analysis of otoliths can be applied at fine time scales (e.g. 15 days) to deduce movements among water bodies by individual fish at particular times during their lives. Thus, an individual common snook or red drum caught in Tampa Bay as an adult may be traceable to a particular water body that it occupied as a juvenile.

Three recent applications of otolith microchemistry are presented here to illustrate how this technique has been used to learn more about spatial distribution patterns of fish populations in Florida. Patterson et al. (2004) applied otolith microchemistry to determine whether red drum juveniles could be distinguished between Gulf and Atlantic coast estuaries. Distinctive juvenile fingerprints were detected in the otoliths of age-0 and 1 year-old red drum for each coast. When

standardized to calcium, two elements (Mn, Mg) had higher concentrations in the Gulf of Mexico, while Sr and Ba were more highly concentrated in the Atlantic estuaries. The only area represented by samples from more than one year class was Tampa Bay. In the analysis, otoliths from Tampa Bay juveniles that had recruited in 1999 (an El Nino year) had chemical signatures distinctly different from samples taken from the two other year classes analyzed (i.e. 1982, 1998). Since characteristics of the water mass reflected in the otolith varied from year to year in a particular system, patterns in nursery habitat contribution may change from year to year as well. Thus, it is essential that projects aimed at identifying nursery contribution analyze recruits from several annual cohorts in a system.

Patterson et al. (2005) also applied otolith microchemistry to determining if common snook juveniles could be distinguished between Gulf Coast (Charlotte Harbor) and Atlantic Coast (Indian River Lagoon at Tequesta) estuaries in Florida. As with red drum, each coast had a distinctive fingerprint, reflected in the elemental composition of otoliths from age-0 and 1 year-old common snook. Three elements were significantly different between the coasts (Mn, Ba and Sr). Furthermore, they investigated the question: do adult common snook in Florida Bay come from nurseries located along the coasts of the Atlantic or Gulf? Although adult common snook are common in Florida Bay and the Keys, young-of-the-year common snook are rarely observed there. To answer this research question, they analyzed the microchemistry of adult otolith cores, i.e. the group of rings laid down when the individual was a young juvenile. Their results showed that some of the Florida Bay common snook did indeed come from either a Gulf or Atlantic nursery area; however, a substantial number of individuals could not be clearly assigned to either coast, and their nursery origins were unknown. Thus, otolith microchemistry studies may reveal that previously undiscovered nursery grounds exist in an area, leading to further investigations to locate sources of recruits.

These two studies (Patterson et al. 2004, 2005) demonstrate that differences in otolith concentrations of three elements (Mn, Ba and Sr) were detectable between estuaries located along the east and west coasts of Florida for both red drum and common snook. Differences in concentrations of these elements often reflect differences in salinity of the water mass the fish was living in when the relevant otolith segment formed (Campana 1999). Regardless of the cause of the variation, these studies demonstrate that otolith microchemistry can be used to identify distinctions between nursery habitats for red drum and common snook at a large scale (i.e. hundreds of miles apart).

More recently, a study investigating nursery habitat distinctness at a finer spatial scale was conducted for gray snapper (*Lutjanus griseus*), a species having a life history that is similar to red drum and common snook. Lara et al. (2008) analyzed otoliths from juvenile gray snapper distributed in several water bodies around the Florida Everglades, including Florida Bay, Biscayne Bay and the Ten Thousand Islands. High resolution ICP-MS and meticulous sample processing led to the detection of 32 trace and rare earth elements. Eight of these elements (Mn, Rb, P, Li, Y, Dy, Er, Sc) comprised the optimal subset that contributed substantially to separating otolith samples among the nursery areas. This study revealed that gray snapper juveniles displayed distinctly different otolith microchemistries among areas that were as close as 10 km apart. Thus, otolith microchemistry analysis may detect separate distinctive nursery areas existing in close proximity to each other, each likely to contribute relatively different quantities of recruits to Bay stocks.

The level of specificity achievable in detecting habitat use by a fish through analysis of otolith microchemistry depends on (1) movement of the individual within its habitat, and (2) characteristics of the water mass within this habitat. For example, the Small demersal juvenile

stage in the life history of common snook is about four months in duration (Figure 3). The habitat, or home range, occupied during this phase is located along a shoreline lined with mangroves, shrubs, or salt marsh plants. The home range used by juvenile snook consists of a particular segment of a tidal tributary throughout the demersal phase (Gilmore et al. 1983; Brennan et al. 2008). Salinity tolerance of small demersal snook is broad, and the range within its habitat may vary widely (e.g. from 0 to 40 ppt). Thus, an individual juvenile that grows sufficiently and survives to the next life phase, presumably settled within a suitable tidal tributary segment and occupied a home range area with sufficient food resources, cover and suitable environmental conditions. The water mass within this home range generates the elemental composition of the fish's otoliths over the 4 months. Among the areas of the Tampa Bay system that might be suitable as juvenile snook habitat, differences in concentrations of elements in fish otoliths reflect anthropogenic, natural factors, or a combination of both encountered in their home range. Strontium-to-calcium ratios (Sr:Ca) are often used to distinguish salinity influences since freshwater usually has lower Sr concentrations compared to seawater (Campana 1999; Limburg and Siegel 2006). Other elements (Ba, Mn, Li, P) occurring in surface and ground water that may become incorporated into fish otoliths derive from dissolved elements due to weathering of rocks and sediments in their watersheds (Limburg and Siegel 2006). Otolith elements may also be derived from industrial, atmospheric, agricultural or urban runoff (e.g. Mg, Mn, Ba, Pb, Zn, Ba) (Dove and Kingsford 1998; Thorrold et al. 2007).

A proof-of-concept test—The current study team conducted a test using otolith microchemistry to determine if patterns in spatial variation among red drum nursery habitats could be distinguished at the relatively smaller scale of Tampa Bay (Figure 5). A collection of 24 red drum otoliths from 2005 was available for analysis. Otoliths that had been extracted and stored in vials were cleansed and dissolved whole in 0.5N (~2%) nitric acid by analytical staff at the USF College of Marine Science laboratory. Using solution-based ICP-MS, 19 elements were identified and were expressed as ratios to calcium (an internal standard). Discriminant function analysis was used to examine patterns among areas of the Bay based on the chemical composition of each otolith. The analysis demonstrated that, with one exception, each fish could be accurately assigned to the area of the Bay from which it had been collected (Figure 6). Nine elements contributed to the classification: Li, Na, Mg, P, Mn, Ge, Rb, Rh and Ba. Despite the small sample size, we determined that fish captured from different regions of the bay and two rivers had chemically distinctive elemental ratios in the otoliths that corresponded to the region of capture.

National Fish Habitat Action Plan Grant: 2009-2012—The results of the proof-of-concept test were very encouraging and led to development of a grant proposal, which was accepted for funding under the National Fish Habitat Action Plan (NFHAP) (McIvor et al., unpublished). This project focuses on red drum and common snook around Tampa Bay, addressing the overall question: *to what degree do specific estuarine locations contribute to adult fished populations in Tampa Bay?* Specifically, the study objectives are to:

- identify distinctive nursery areas by detecting spatial patterns in otolith microchemical fingerprints for young-of-the-year juveniles;
- determine if adults collected in Tampa Bay can be assigned to distinctive nursery areas based on otolith core microchemistry;
- determine if elemental fingerprints for fish from specific locations remain constant in successive years.

Samples of young-of-the-year (YOY) juvenile common snook and red drum have been collected using beach seines in over 36 tidal tributary, riverine and estuarine systems under FWRI fishery-independent monitoring and supplementary programs (Figure 5). In addition, USGS has archived samples from a variety of studies (McIvor et al., unpublished). As a result of these

efforts, and of ongoing sampling trips, over 1,000 individual YOY red drum and common snook will be available. Subsamples will be selected representing a broad spatial distribution of juveniles from the 2006 through 2009 cohorts. They will be used to identify distinctive nursery areas by detecting spatial patterns in otolith microchemical fingerprints, and to determine if these patterns change among years.

Samples of adults and sub-adults are being collected through routine FWRI monitoring programs. These samples derive from each of the four main Bay subareas (Figure 5). They will be used to determine if adults collected in Tampa Bay can be assigned to distinctive nursery areas based on microchemistry of the otolith core.

Our research team will also examine evidence and generate hypotheses about what environmental factors may help explain differences in nursery contributions to fisheries identified in this study. Comprehensive ecological investigations have been undertaken in most of the nursery areas and considerable background information has been generated (e.g. Greenwood et al. 2008a; b; Sherwood and the Tampa Bay Tidal Tributary Research Team 2008). We will analyze ancillary data collected with each fish sample, including water temperature, salinity, dissolved oxygen, water depth and habitat type. In addition, our team will generate two synoptic sets of water samples to characterize the elemental signatures of the locations from which the archived otoliths originated.

Implications for land use and resource managers—Year-to-year variation in elemental fingerprints will be examined over the 3-year study period. However, the full range of potential conditions in hydrology, rainfall, temperature, and other sources of variation may not be present during this period. Studies elsewhere have concluded that if the nursery sources of the adults are to be determined, a library of elemental fingerprints needs to be built up over time for contributing systems (Gillanders 2002). Thus, our study should be considered a first step in evaluating the potential use of otolith microchemistry as a tool for identifying specific contributing nursery habitats.

For fishery resource managers, these results will enhance the base of knowledge on essential fish habitat for common snook and red drum. For land use and watershed managers, the information produced under this ongoing project builds upon previous and other current efforts in tidal tributaries (Krebs et al. 2007; Greenwood et al. 2008a; b; Sherwood and the Tampa Bay Tidal Tributary Research Team 2008). Specifically, recent research in Tampa Bay has demonstrated that small tidal tributaries flowing directly or indirectly into the bay often contain higher abundances of juveniles of certain economically-important fish and crustacean species than other habitat types within the bay ecosystem. This distinction is true for common snook and red drum. For common snook, seasonal and interannual patterns of recruitment to – and growth within – tidal tributaries, microhabitat use within those tributaries, and details concerning their diet and trophic relationships have been documented. Although we have established a general understanding of the geographic distribution and ecological function of these nursery habitats, we are now in critical need of an ability to assess the relative contributions of individual habitat locations, as they relate to adult stocks. Since limited financial resources require careful and strategic application of habitat protection efforts, the results of the current investigation should provide a basis for prioritizing conservation, protection and restoration efforts that influence nursery habitats around Tampa Bay.

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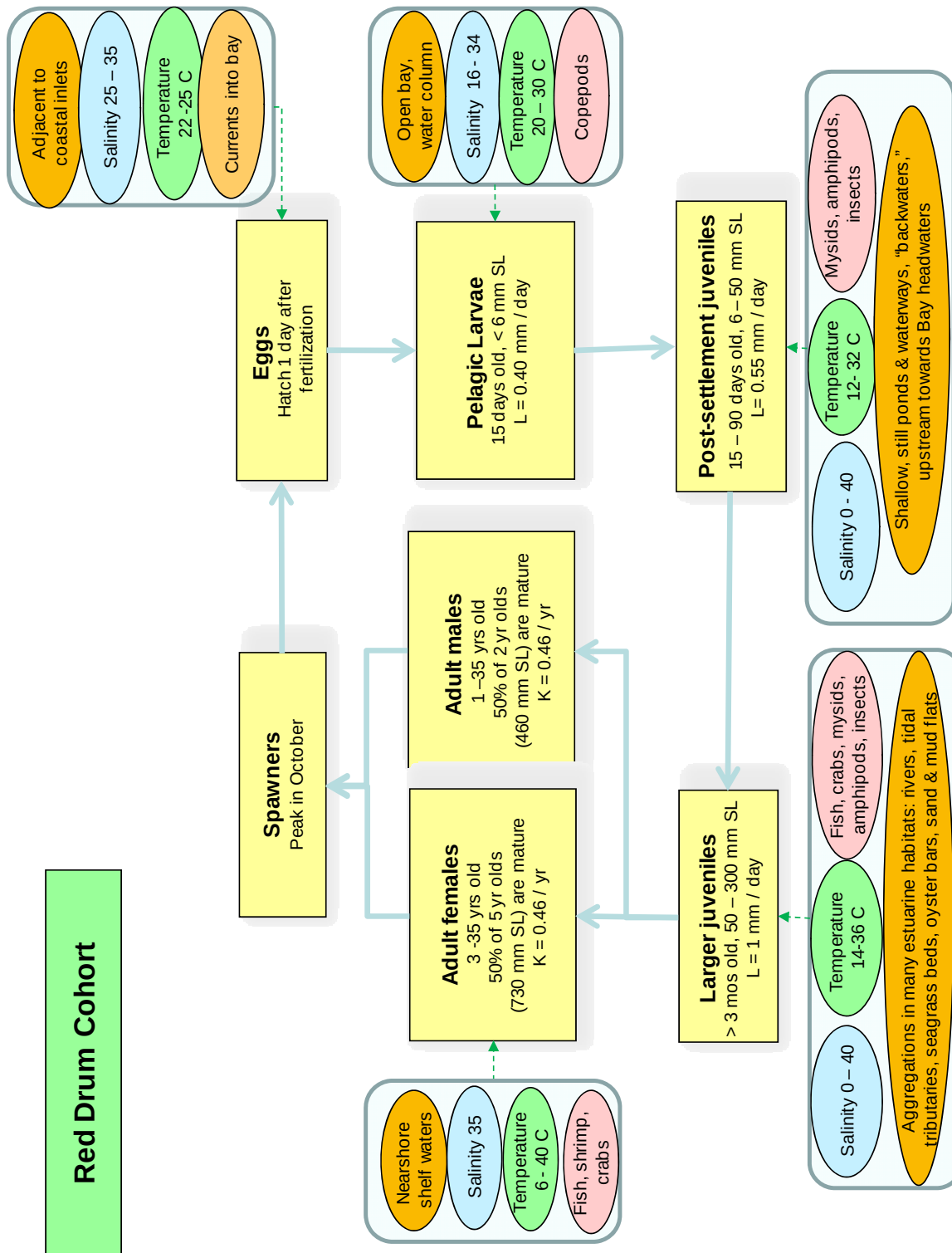


Figure 1. Life history diagram red drum (*Sciaenops ocellatus*) representing the phases in the life of a single cohort. Boxes = phases including duration, standard length (SL), and growth rate (L = linear growth, K = von Bertalanffy growth); ovals = environmental influences and foods consumed.

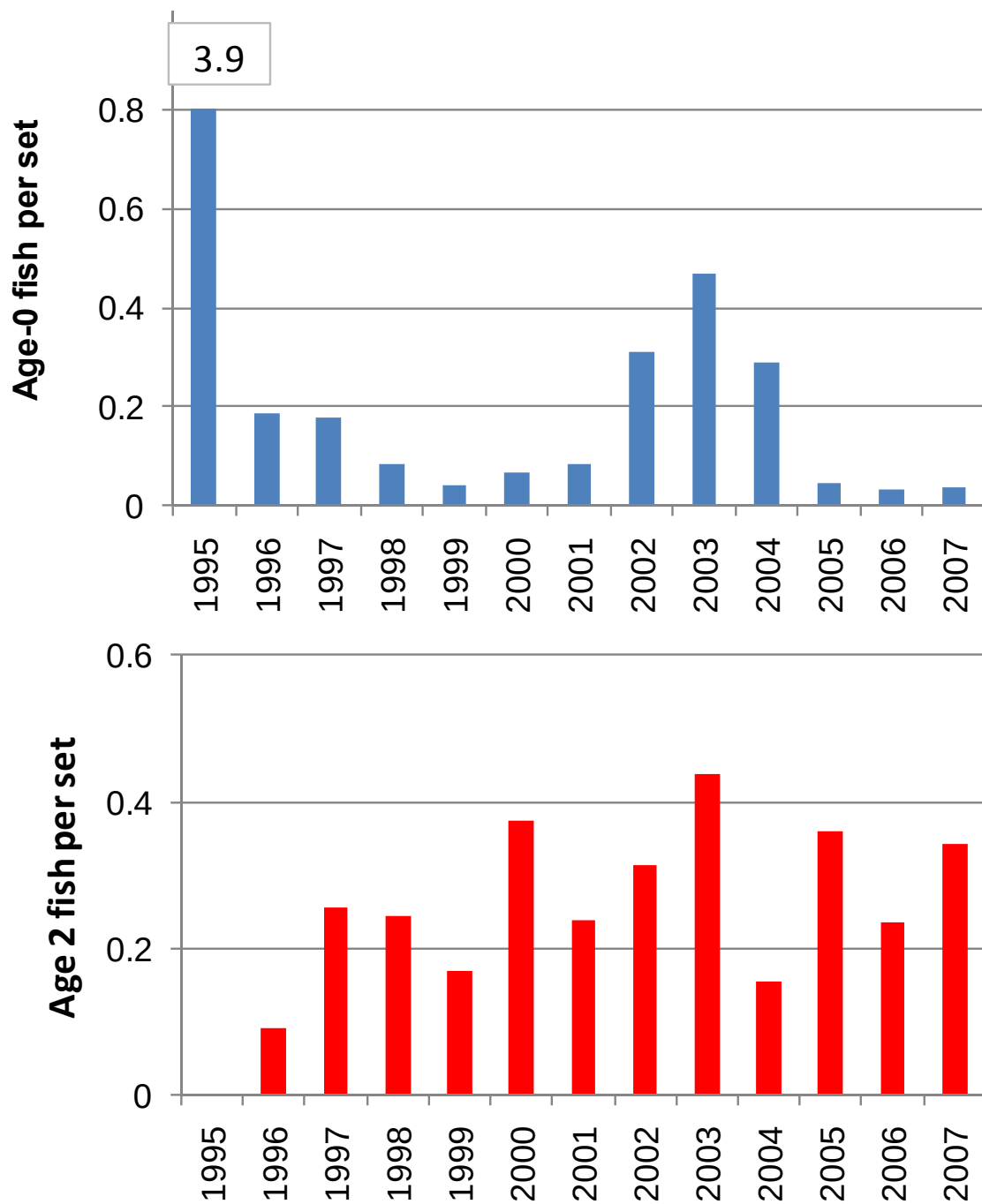


Figure 2. Fishery-independent monitoring catch-rates for juvenile red drum from monthly stratified random sampling by FWRI monitoring teams deploying 21.3-m and 183-m seines in estuaries along Florida's Gulf coast, 1995-2007. a. young-of-the-year (40 mm SL or less, 21.3-m seines); b. age-2 (183-m haul seine). (SOURCE: Derived from Murphy and Munyandorero 2009, pages 78 and 79)

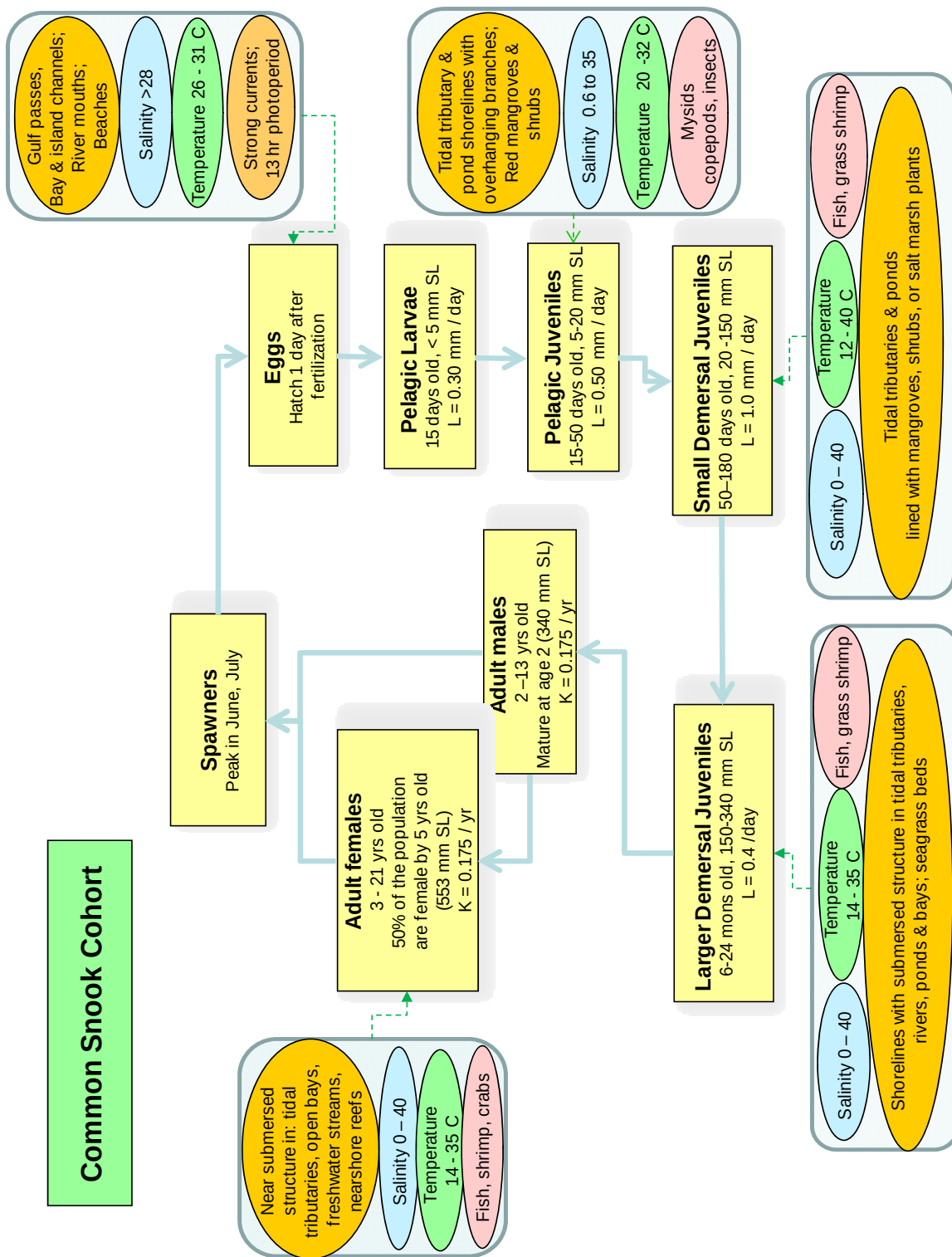


Figure 3. Life history of common snook (*Centropomus undecimalis*) representing the phases in the life of a single cohort. Symbols and abbreviations as listed in Figure 1.

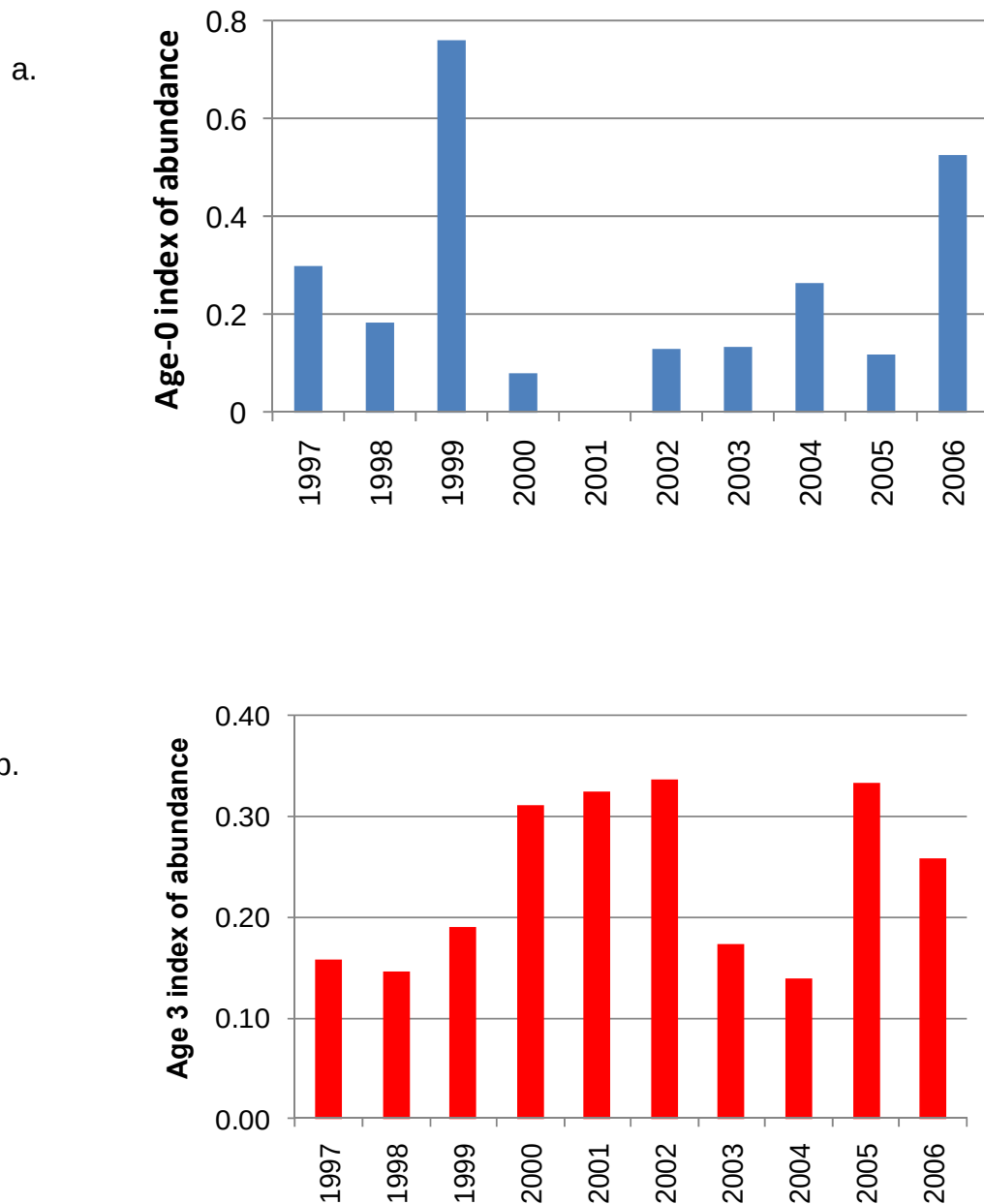


Figure 4. Fishery-independent monitoring catch-rates for juvenile common snook from monthly stratified random sampling by FWRI monitoring teams deploying 21.3-m and 183-m seines throughout Tampa Bay, 1997-2006 a. Young-of-the-year (abundance index based on fish 126 mm or 5 inches SL or less); b. Age-3 (abundance index based on otolith aged subsamples proportionally applied to overall abundance of fish caught). (SOURCE: Derived from Muller and Taylor 2006, unpublished background data)

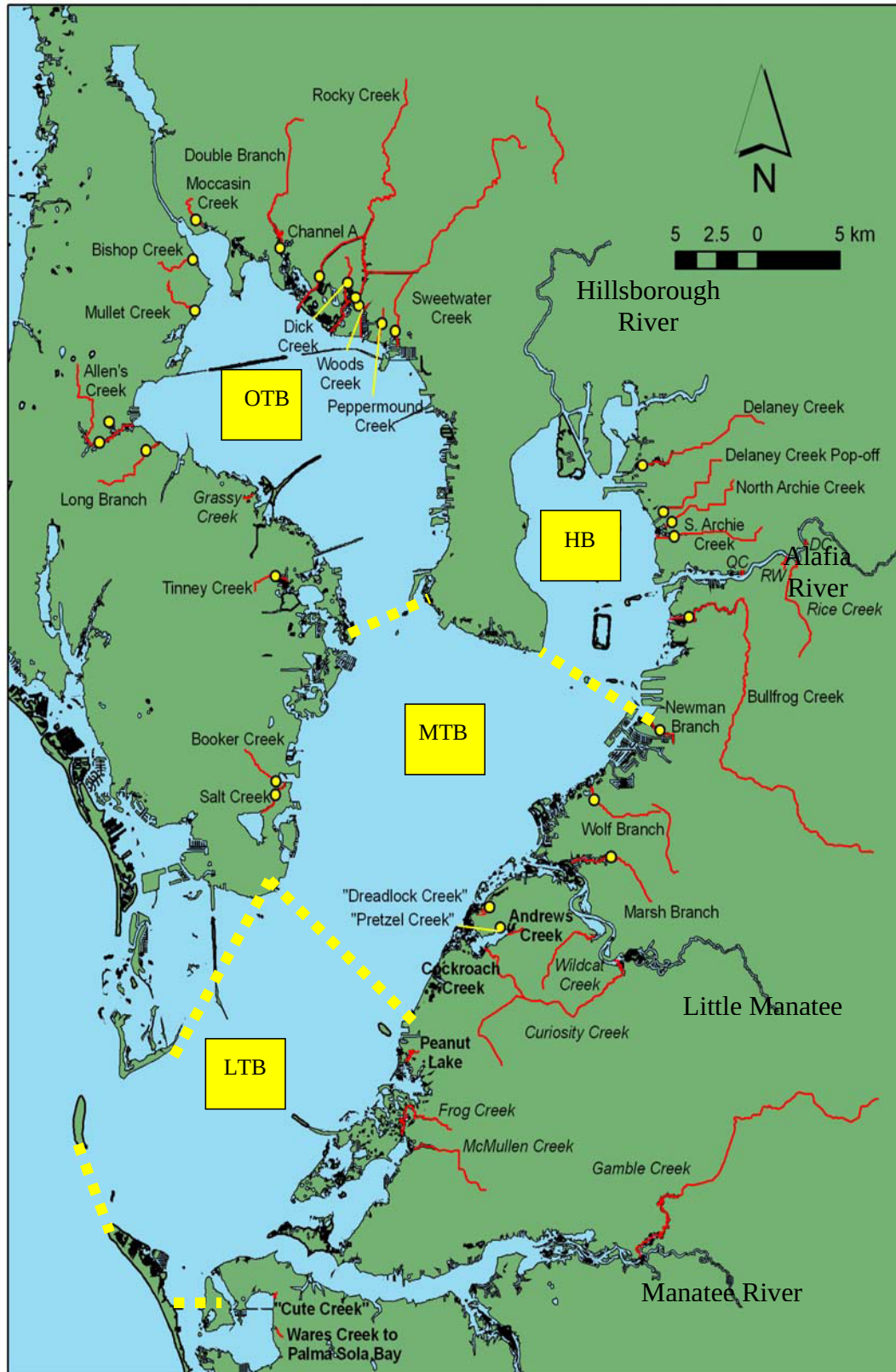


Figure 5. Tidal tributaries and subareas of Tampa Bay where sampling has been conducted for young-of-the-year juvenile common snook and red drum, 2005-2010. Dashed lines indicate boundaries of subareas of Tampa Bay, OTB = Old Tampa Bay, HB = Hillsborough Bay, MTB = Middle Tampa Bay, LTB = Lower Tampa Bay.

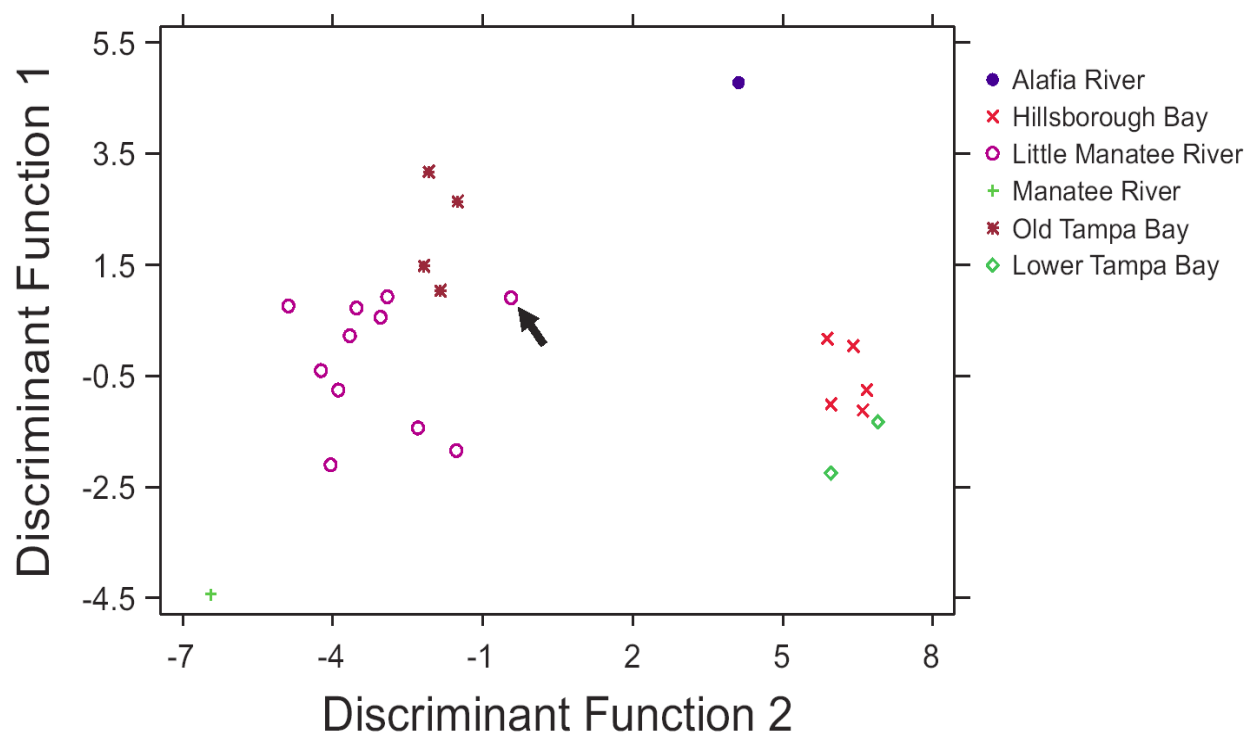


Figure 6. Discriminant function analysis of 24 red drum juveniles based on 19 elements identified in microchemical analysis. Fish samples were collected by FWRI fishery-independent monitoring teams in 2005. Each symbol represents an individual fish; each symbol-type represents a subarea of Tampa Bay (Figure 5). Arrow indicates the single fish that was not assigned to the location in which it was caught based on the analysis.

USING ESTUARINE FISH STABLE NITROGEN ISOTOPES TO PINPOINT LAND COVER NUTRIENT ORIGINS

E. Malkin, D. Hollander & E. Peebles

ABSTRACT

A novel, stable-isotope-based methodology is described that will allow comparison of the nitrogen connectivity of different land-cover types in estuarine watersheds to nitrogen in estuarine organism tissues. This approach: (1) identifies the specific land use types in the watershed that can be targeted for effective nitrogen reductions; and (2) provides a criterion for evaluating the effect of new best management practices for land-cover or hydrologic alterations to the watershed. The method is based on using ordinary least-squares multiple linear regression with forward selection to fit variation in local tributary proportional land-cover area to variation in local tributary fish nitrogen isotopes. In a mechanistic sense, empirical coefficients fitted in this process represent the product of the processes delivering nitrogen from land-cover to receiving waters (nitrogen connectivity) and mean land-cover nitrogen isotope ratios. These products (fitted coefficients) are repeatedly ($n = 10,000$) estimated by bootstrapping proportional area and fish-isotope data from different combinations of local tributaries. The bootstrapped product distributions can then be divided by measured mean land-cover isotope ratios (as $\delta^{15}\text{N}$ in dissolved inorganic nitrogen) to produce de facto, bootstrapped land-cover nitrogen connectivity coefficients even if different land covers have similar isotope ratios. This methodology has undergone successful initial testing in 14 west-central Florida estuarine tributaries, which indicated row crops and citrus have much higher nitrogen connectivities to estuarine organisms than other non-urban land-covers, despite their low relative areas. Wetlands, on the other hand, had negative nitrogen connectivity, as might be expected.

INTRODUCTION

Agriculture is Florida's most important economic sector, second only to tourism, which is driven by the region's billion dollar recreational fishing industry. Sub-estuary receiving waters that drain agricultural watersheds support a variety of economically important fishes and crustaceans because during their juvenile stages, species such as snook, red drum, striped mullet, tarpon and blue crab populations utilize these habitats as feeding nurseries (Killam et al. 1992; Peebles et al. 1991; Peebles et al. 2007) often deriving their biomass via hypoxia-susceptible benthic trophic pathways (Sherwood and the Tampa Bay Tidal Tributary Research Team 2008). Potentially, nutrients, a byproduct of the second most important sector, may either support or degrade fish nursery habitats which are essential to Florida's most important economic sector. Before implementing policy to preserve both economic sectors, it is crucial to understand the relationship between anthropogenic nitrogen and fish. This is because excessive nutrient export from agriculture may destroy the economically- important benthic-dependent nursery fish habitats as seen in many polluted systems, while according to the agricultural model (Caddy 1993), completely shutting off nutrient support or completely removing water supply would result in no fish biomass. To prevent these two economic sectors from collapsing, appropriate nutrient policy must be established, but first, the pathway that links nutrients derived from specific agricultural land use to fish must be defined.

Water Management & Hydrology—Another key issue affecting both economic sectors, especially in west-central Florida, is water management, and there is a growing emphasis on using rivers for water supply for agricultural applications rather than groundwater. In general, agricultural operations withdraw large amounts of groundwater for irrigation, which is eventually transformed into surficial flow which can carry nutrients into receiving water. This process has recently been shown to cause salt intrusion of ground water causing drinking water contamination.

On the other hand, groundwater irrigation processes provide a mechanism of water and nutrient delivery, especially during Southwest Florida's dry season (March through May). Since rainfall and surface runoff are the major processes that deliver land use nutrients to fish habitats, and these processes peak during this region's wet season (June through September), it is essential to take into account seasonal hydrological connectivity when defining the nitrogen pathways between land use and fish during the wet season.

Stable isotopes identify nutrient pathways—Stable nitrogen isotopes have been widely used to define nitrogen pathways in a variety of sub-estuaries. These tracers are appropriate for determining the effects of nutrients on fish biomass because the isotopic composition of the actual organisms and the dissolved species present in aquatic ecosystems inherently provide information on coupling between watershed nutrients and receiving waters. Dissolved inorganic nitrogen (DIN) supports primary production downstream, but also carries with it information on its land-use origin in the form of stable isotopes of nitrogen and oxygen (Kendall 1998). For example, land use types such as agriculture often exports nitrate with very heavy $\delta^{15}\text{N}$ values due to large amounts of $^{14}\text{NH}_3$ and $^{14}\text{N}_2$ released during ammonia volatilization and denitrification that occur after fertilizer application. DIN derived from treated sewage and septic systems undergo similar biogeochemical processing. DIN derived from urbanized soils often have a wide variety of isotopic values because of the various forms of nitrogen processing that occurs in urban watersheds. If developed land use does not play a role in delivering nitrogen to ecosystems, then the isotopic composition of nitrate would likely indicate a non-land-use specific source of nitrogen such as the light values of NO_x (Kendall 1988) present in atmospheric deposition.

Objectives—In this study we use nitrogen stable isotopes in fish muscle in 14 west-central Florida watersheds to identify the pathways between different land-use types and the resident fish species occupying receiving waters during this region's wet season. We aim to 1) define the specific anthropogenic and natural land-use types that influence the isotopic composition of nitrogen in fish muscle, 2) determine the specific land-use percent contribution to the isotopic composition of nitrogen in fish muscle 3) assess the actual values of the isotopic signatures that are associated with these land-use types. To clarify the affects of hydrology on these linkages, this study will be conducted during west-central Florida's wet and dry seasons. We also discuss how results of this study relate to isotopic foodweb studies in the area as well as nutrient loading.

METHODS

Project scopes and study sites—Isotope data reported here originate from three investigations. The first, Hollander and Peebles (2004), identified the primary producers that support upper-trophic-level biomass in four west-central Florida tidal river sub-estuaries (Figure 1, left panel). The second and third investigations, the "tidal tributaries study" (Sherwood and the Tampa Bay Tidal Tributary Research Team 2008) and Greenwood et al. (2009) targeted smaller tidal stream sub-estuaries (Figure 1, right panel). Isotopic fish data from all three studies are considered here.

Sample collection and processing—This study aims to define the nitrogen linkages between agriculture and fish in diverse west-central Florida tidal fish nursery habitats by relating land-use to stable N isotopes in fish muscle in 14 different sub-estuaries, instead of focusing within individual systems. Resident fish species, rainwater killifish (*L. parva*) and silversides (*Menidia spp.*), were collected from most sub-estuaries. In cases where these species were unavailable, alternative estuarine resident species including tidewater mojarra (*E. harengulus*), hogchoker (*T. maculatus*), and clown gobies (*M. gulosus*) were collected. These species were determined to occupy similar trophic positions in all systems (Greenwood et al. 2009) and are supported by benthic trophic pathways (Hollander and Peebles 2004). These sessile resident species are

appropriate for defining nutrient source because they incorporate nitrogen across a wider spatial and temporal scale within each sub-estuary compared to the instantaneous concentration and isotope ratio measurements of dissolved species such as nitrogen and ammonium.

In Hollander and Peebles (2004), fish samples were collected at two to four different sites along sub-estuary reaches ranging from 5 km to 25 km. Each sampling site was only visited once for sample collection during the wet season. Wet season sampling dates ranged from September to October of 2003. Greenwood et al. (2008) collected fish from approximately one to two sites along sub-estuary reaches ranging from 1 km to 3 km. Each sampling site was only visited once for sample collection during the wet season. Wet season sampling occurred in September 2007. In the tidal tributaries project, samples were collected at four to five sites per sub-estuary with individual sub-estuary reaches ranging approximately from 1 to 4 km. Each sampling site was visited monthly during the wet season for primary producer collection during both seasons. Fish were collected monthly from August through November for wet season data.

Samples of fish tissues were obtained using mid-channel trawls (Hollander and Peebles 2004 study) and shoreline seines (all studies). All samples were limited to tissues (primarily muscle) located posterior to the peritoneal cavity. All solid samples were dried at 50°C, powdered, and frozen until isotopic analysis.

$\delta^{15}\text{N}$ values were measured using a Carlo Erba 2500 Series I elemental analyzer equipped with a zero-nitrogen blank auto-analyzer that was coupled to a Finnigan Mat Delta Plus XL stable-isotope mass spectrometer. All samples were run in duplicate using Spinach Leaves (NIST 1570a) as an internal reference material, and N isotope values are reported in conventional per mil notation (‰) relative to the air international standard. Instrumental standard deviation remained under 0.3‰.

Land-use geographic information system data and watershed boundary geographic information system data for the entire west-central Florida region were downloaded from the official SWFWMD website. According to the metadata files, the 2004 and 2006 land-use were categorized according to the Florida Land Use and Cover Classification System (FLUCCS), and the features were photointerpreted at 1:12,000 using 2004 or 2006 1-meter color infrared digital aerial photographs. The watershed boundary data were interpreted and digitized from the USGS 1:24,000 quadrangles, and the boundaries were categorized according to the USGS Hydrologic Unit Classification coding system. Using these two datasets percent area coverage for each land-use type was calculated using the spatial analyst function in ArcGIS version 9.

Because agriculturally-dominated watersheds are rapidly being urbanized, this study utilized a forward stepwise ordinary least squares multiple regression (FSOLSMR) and a bootstrapping approach to assess the contribution of each land-use type to fish biomass. This technique is generally used in situations where data is expensive, information is sparse, assumptions of normality are unclear, or where further data are difficult to acquire (Henderson 2005).

Pinpoint the Land-Use Types that influence fish $\delta^{15}\text{N}$ —The relative percent of 27 land-use classifications were calculated in each watershed. To remove effects of collinearity, correlated land-use types were grouped resulting in 12 land-use variables. For each of the 14 watersheds ($n=14$), the relative percent land cover of 12 land-use groups were considered as possible independent variables that explain the dependent variable of each watershed's average fish $\delta^{15}\text{N}$ values.

A bootstrap program using R was then executed to randomize the observations and to conduct a FSOLSMR to select 3 land-use variables that explain fish $\delta^{15}\text{N}$ variability. The selected land-use groups were then automatically recorded in a matrix. This process was repeated 10,000 times to ensure data robustness; no single observation could dominate the selection process. The significance ($p < 0.05$) of selected land-use variables were checked using an ordinary least square multiple regression (OLSMR) on the non-bootstrapped data. Only significant land use-types were further investigated.

Defining land-use specific coefficients—The magnitudes of the coefficients assigned to land-use variables are likely a function of the isotopic composition associated with land-use as well as the amount of nitrogen exported from that particular land-use type. Fish $\delta^{15}\text{N}$ from local tributaries ($\delta^{15}\text{N}_{\text{TISSUE\_TRIB}}$) can be described as the combination of nitrogen connectivity (C), land-cover specific $\delta^{15}\text{N}$ ($\delta^{15}\text{N}_{\text{DIN}}$), and land-cover-specific proportional area (A) across n land-cover types in the model:

Equation 1.

$$\delta^{15}\text{N}_{\text{TISSUE\_TRIB}} = (C_1)(\delta^{15}\text{N}_{\text{DIN}_1})(A_1) + (C_2)(\delta^{15}\text{N}_{\text{DIN}_2})(A_2) + (C_3)(\delta^{15}\text{N}_{\text{DIN}_3})(A_3) \dots \\ + (C_n)(\delta^{15}\text{N}_{\text{DIN}_n})(A_n) + k$$

The constant k is a trophic constant that accommodates isotopic enrichment associated with trophic level ($\sim 3\%$ per trophic level above primary producer).

Because the ranges of these coefficients provide important information about ecosystems, the confidence intervals around the coefficients were more appropriately determined using a bootstrap OLS multiple regression model using Statistics 101 Version 1.4 with an algorithm modified from (Simon 1997), similar to those described by Draper and Smith (1998).

The significant land use variables and fish isotope data were entered. Observations were randomized, and values for the coefficients of the land-use variables were recorded. This process was repeated 10,000 times to construct distributions of each coefficient values.

Defining land-use specific contributions to fish biomass—To determine land-use specific contributions to the nitrogen isotopic composition of fish muscles, R^2 distributions were created to describe how much fish $\delta^{15}\text{N}$ variability was explained by each land-use type. This was achieved using a bootstrap methodology similar to that described above. In every bootstrap sample, predicted fish isotopes were calculated using a single land-use type's variables and coefficients calculated from the FSOLSMR. Predicted fish isotopes were correlated against observed values and the resulting squared correlation coefficient was recorded. This process was repeated for each land use variable in every bootstrap sample. This process was repeated 10,000 times to construct R^2 distributions. Concurrently, total variability explained by the model was also determined using a bootstrap method with all variables and coefficients to calculate predicted fish isotope values. These predicted fish isotope values were regressed against observed values and the resulting adjusted R^2 value was recorded. It should be noted that the sum of the R^2 values for the individual land-use type will not necessarily add up to the total adjusted R^2 distribution, since individual land-use R^2 were not calculated using the residuals of the FSOLSMR, rather they were the direct squared correlation of the predicted values to the observed values.

RESULTS

Pinpoint the Land-Use Types that influence fish $\delta^{15}\text{N}$ —Row crops, tree crops, and vegetated wetlands were all selected by the bootstrapping procedure and all had significant p values calculated from OLSMR. Land-use types which covered a relatively high percentage of the watershed, such as fallow cropland and urbanized areas, were not selected by the model.

Defining land-use specific coefficients—The wet season coefficient distributions for row crops, tree crops, and wetlands are depicted in Figure 2. Row crops and tree crops had coefficient magnitudes as expected assuming that agriculture areas export nitrogen with high $\delta^{15}\text{N}$ values. Wetlands on the other hand had a coefficient distribution centered on negative values, either implying nitrogen removal from the system or the export of low $\delta^{15}\text{N}$ nitrogen values.

Defining land-use specific contributions to fish biomass—The wet season R^2 distributions for row crops, tree crops, wetlands, and overall adjusted R^2 are depicted in Figure 3. Row crops and tree crops explained the majority of the variability of fish $\delta^{15}\text{N}$ while wetlands explained less than 20% of the variability. Overall adjusted R^2 values were high; 92% using these three land use groupings.

DISCUSSION

The three major land cover types shown to influence the isotopic composition of fish muscle were shown to be row crops, tree crops, and vegetated wetlands. Results imply that these specific land use types export more nitrogen to nearby aquatic ecosystems than their relative areas would suggest. Figure 4 compares the mean percent relative area covered to the mean percent fish $\delta^{15}\text{N}$ variability explained.

This methodology is relatively cheap and could potentially be used to support hydrological models that predict nutrient exports based on land-use cover. The relationship between an area of a specific land-use and its influence on nitrogen loading can be obtained by simply dividing the percent variability explained by the relative land area covered to obtain a given land-use's specific nitrogen influence per given unit area of land cover (Figure 4).

This may be more effective than calculating nutrient concentrations in water bodies adjacent to specific land-use types since the nature of nutrient concentrations are often fickle.

Furthermore, the connectivity coefficients in the mechanistic model described in Equation 1 can be used to directly assess the amount of nitrogen released from an individual land-use type and whether these land-use types are a net sink or source of nitrogen. For example, the isotopic composition of nitrogen derived from agricultural land use is likely enriched in ^{15}N (Kendall 1997). According to bootstrapped coefficient results, row crops and tree crops would therefore have a positive connectivity coefficient assigned, indicating that these land-use types are a major source of nitrogen during the wet season. Nitrogen derived from wetlands on the other hand are likely less enriched ^{15}N , but have been shown to have positive $\delta^{15}\text{N}$ values (Hollander and Peebles 2004). The negative bootstrapped coefficient therefore implies that wetlands are a net sink of nitrogen during the wet season.

Nutrient transport in this region is a dynamic process in which agricultural land-use exports heavy nitrogen via runoff with the transport mechanism being daily thunderstorms. As runoff is filtered via wetlands, some heavy $\delta^{15}\text{N}$ is removed from the system on its path to reaching aquatic ecosystems. Once exported nitrogen reaches aquatic ecosystems, it is incorporated into algal biomass. This algal biomass is transferred to resident estuarine fish species via invertebrate trophic

intermediates, either through a Benthic microalgal-based foodweb or through water column phytoplankton that have been deposited to the benthos forming a phytoplankton-deposit based foodweb (Sherwood and the Tampa Bay Tidal Tributary Research Team 2008). In the watersheds studied here, the ultimate source of nitrogen to these systems is fertilizer applied to tree crops and row crops during the wet season. Nitrogen connects Florida's most important economic sector to its second most important economic sector; stable isotopes of nitrogen have the potential to monitor the strength of this connection.

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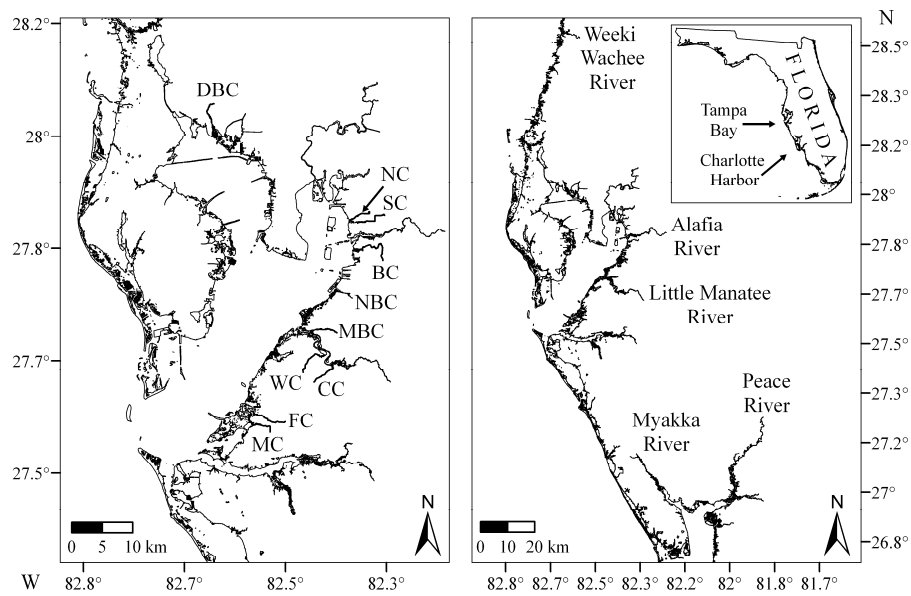


Figure 1. Study sites. DBC = Double Branch Creek, NC = North Archie Creek, SC = South Archie Creek, BC = Bullfrog Creek, NBC = Newman Branch Creek, MBC = March Branch Creek, WC = Wildcat Creek, CC = Curiosity Creek, FC = Frog Creek, MC = McMullen Creek.

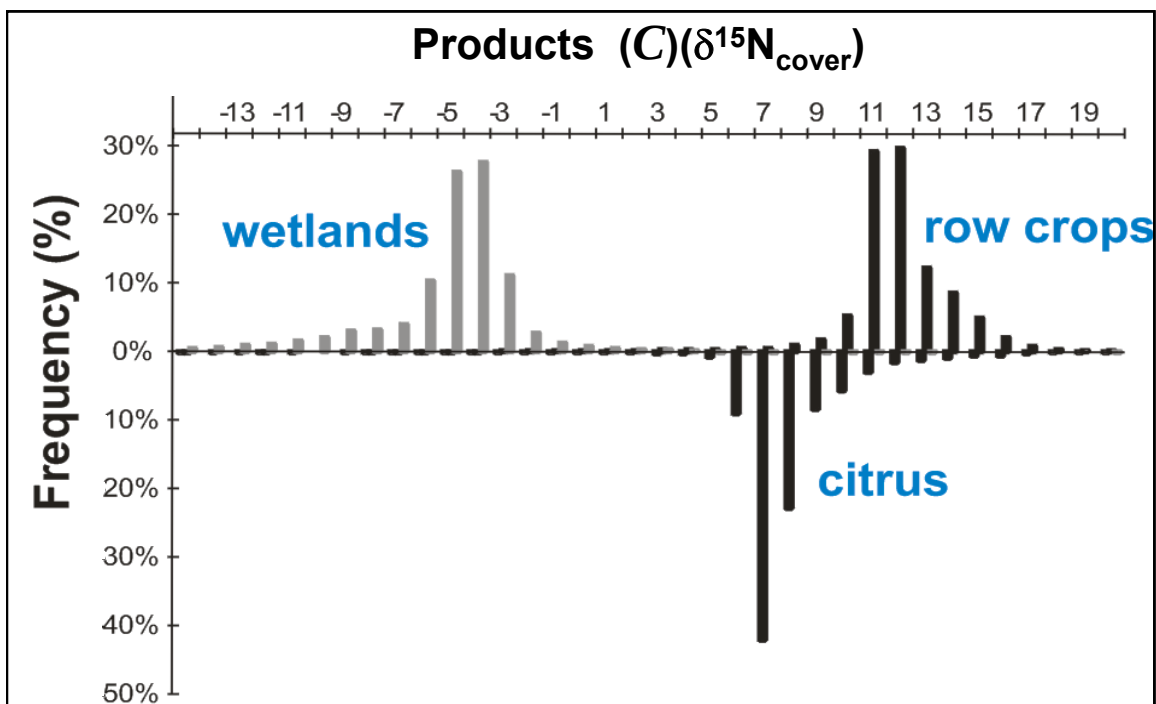


Figure 2. Bootstrap distributions of the coefficients (Products on the x-axis) assigned to independent variables of land use relative area using OLSMR. Frequency on the y-axis is the number of bootstrap sample observations divided by the total number of bootstrap observations.

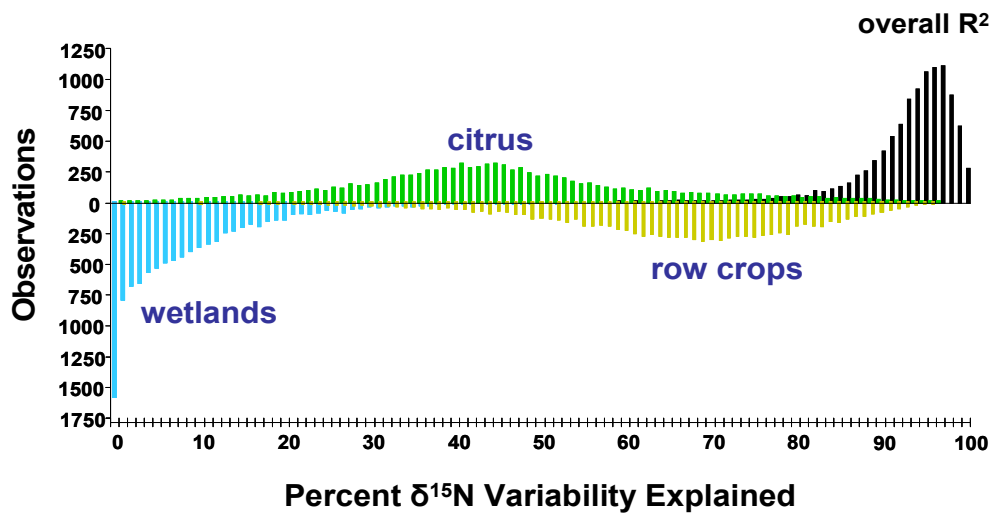


Figure 3. Bootstrap distributions of the R^2 of individual land use types and total adjusted R^2 (Percent $\delta^{15}\text{N}$ Variability Explained on the x-axis) describing the amount of fish ^{15}N variability explained by land use. Observations on the y-axis are the total number of bootstrap observations.

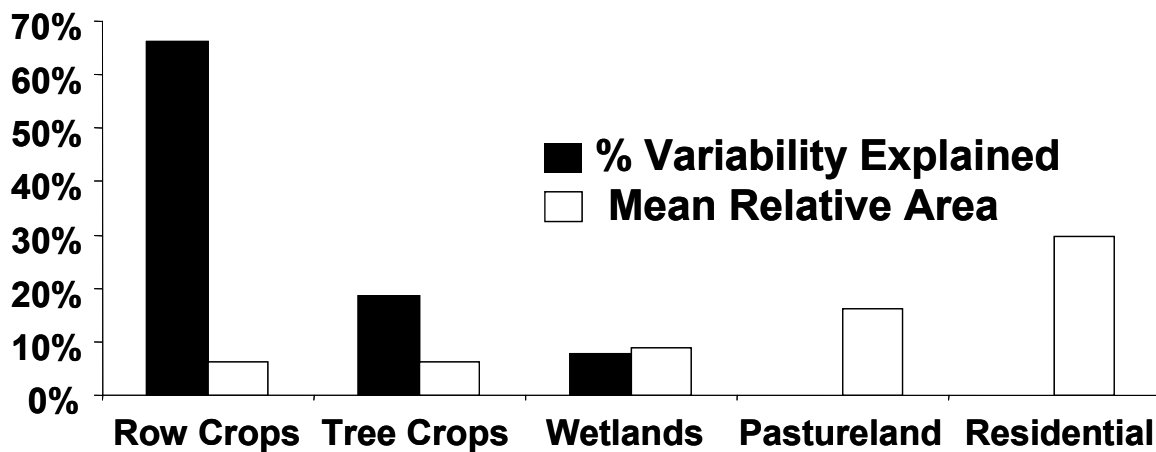


Figure 4. Comparison of the mean percent $\delta^{15}\text{N}$ variability fish variability explained by individual land use types and the mean amount of each land use's area.

RESULTS AND MANAGEMENT RECOMMENDATIONS FROM THE 2006 TAMPA BAY TIDAL TRIBUTARIES HABITAT INITIATIVE

E.T. Sherwood

ABSTRACT

Tidal tributaries within the Tampa Bay estuary encompass a collection of system types including coastal and riverine creeks with and without direct freshwater input, dredged inlets, and other “backwaters.” Relative to other, larger riverine systems, the ecological condition and function of the >100 small tidal tributaries flowing into the bay is not well understood. In 2006, the Tampa Bay Estuary Program embarked on a collaborative research project to assess the importance of these systems to estuarine processes. Water and sediment quality, benthos, and nekton species were found to be variable across the nine small tidal tributaries studied. Landscape development intensity along the tributary corridors was associated with water and sediment quality degradation. Position of the tributaries along the estuarine gradient appeared to influence the variability in observed ecology, and seasonal shifts in benthic microalgae production played an important role in nitrogen pathways to nekton. Common snook (*Centropomus undecimalis*), an important fisheries species, were significantly more abundant in these tributaries than adjacent habitats.

Preliminary management actions developed from the results included: 1) maintaining connectivity between open bay waters, tidal rivers, and smaller, tidal tributaries to allow fish movement, water flow and nutrient flux between systems; 2) reducing “flashiness” of water flow to tidal tributaries to promote natural flow patterns and foster the productivity of fish food sources within these systems; 3) tracking the condition of additional tidal tributaries to further assess their uniqueness; and 4) improving public education and stewardship of tidal tributaries by promoting the importance of these systems as key habitats to important estuarine fish species. Based on these recommendations, local partners are currently investigating the feasibility and implications of salinity barrier removal as a restoration option for Tampa Bay tidal tributaries.

INTRODUCTION

Tidal creeks have a major influence on the productivity and diversity of natural resources in many estuarine systems (Holland et al. 2004). Based on preliminary work in Tampa Bay, minor, tidally-influenced systems which include coastal and riverine creeks with and without direct freshwater input, dredged inlets, and other “backwaters” - collectively termed “tidal tributaries” - appear to be subject to a range of anthropogenic impacts and serve as important nursery habitat for many species of fishery value. However, current conditions within these minor systems and their associated faunal communities have only been limitedly explored. This information is vital in developing an effective management strategy for these systems. In December 2003, the Tampa Bay Estuary Program’s (TBEP) Technical Advisory Committee identified assessing the importance of tidal tributaries as a top-ranked research need in reaching Tampa Bay’s Comprehensive Conservation Management Plan goals. As a result, TBEP’s Policy Board approved the development of restoration and protection goals for tidal tributaries as a management priority.

Tidal tributaries are important habitats relative to their small-scale within the larger Tampa Bay watershed and are typically amongst the first coastal habitats altered by human land development. Studies from other areas in the southeast U.S. describe tidal creeks as “sentinel” habitats because their relatively small watershed footprint is often superseded by their importance to estuarine value and function (Holland et al. 2004). The Florida Fish and Wildlife Conservation Commission lists the status of Florida’s coastal tidal streams as “poor and declining” (FWCC 2005). A 1986

study of thirty representative tidal tributaries around Tampa Bay suggested that 60% were natural or restorable (TBRPC 1986), indicating that these important habitats are not yet beyond salvation in this area and are still providing important functions to estuarine processes. Yet until recently, very little formal study of the ecology of small Tampa Bay tidal tributaries has taken place.

Initial studies by researchers from the US Geological Survey have noted the importance of tidal tributaries and associated habitats for many highly-valued, estuarine species including blue crab, common snook, and red drum (Krebs et al. 2007; Yeager et al. 2007). In addition, work initiated in Sarasota County on coastal creek systems has focused on rapidly cataloging their relative health using a host of physical and biological metrics (Estevez, these proceedings). The TBEP 2006 Tampa Bay Tidal Tributaries Habitat Initiative aimed to advance the science of “tidal tributaries” by comparing their ecology with surrounding areas such as open-bay and river shorelines and describing links to watershed processes that may have an effect on their overall ecosystem value and current condition in comparison to other, more studied systems in the Bay. More than 100 named and un-named creeks and tributaries have been identified in the Tampa Bay watershed, including approximately 150 which are considered tidally-influenced. Tidal tributaries occur in all areas of the bay (Figure 1) and include urban, residential, agricultural, mosquito ditches, and relatively unaltered drainage areas that occur at or below sea level.

The objectives of the TBEP study were to improve protection and management of these minor, tidally-influenced systems in the Tampa Bay estuary by: 1) characterizing the fisheries resources of Tampa Bay tidal tributaries; 2) determining the effects of various habitat parameters (e.g., watershed condition, water quality, structural habitat, etc.) on fisheries resources in tidal tributaries; 3) developing measurable goals and management recommendations from these characterizations and analyses that will lead to the development of a Tidal Tributary Management Strategy, and 4) ultimately communicating the results and recommendations to managers and the public to support informed decision-making regarding the preservation or restoration of tidal tributary habitats.

METHODS

Assessment of minor, tidally-influenced systems and their adjacent river/bay habitats was conducted in two altered and two unaltered watersheds (Table 1; Figure 1). We hypothesized that tidal tributaries are critical habitats within the Tampa Bay ecosystem and aimed to assess the extent of their importance in relation to adjacent riverine and bay habitats which have been studied in greater detail in the past. As a further working hypothesis, we postulated that water and sediment quality and biological resources in tidal tributary systems whose watersheds have been modified by human activities are degraded relative to those in tidal tributaries with less altered watersheds. A multi-disciplinary sampling regime began during January 2006 to assess the water, sediment, benthos, and fisheries resources within select tidal tributaries of Tampa Bay and the resulting trophic ecology of these sites based on the flow of nitrogen sources from their watersheds to the fish utilizing these tidal tributary systems. Resulting analyses from each of these sampling efforts were focused on the hypotheses listed above.

Tidal Tributary Site Selection, Classification & Watershed Characteristics

Reconnaissance of four selected basins (Feather Sound, Terra Ceia Bay, Alafia River, and Little Manatee River) was conducted in 2005 to assess the habitat conditions (e.g., water depth, inundation of shoreline vegetation, substrate type) of tributaries draining to these areas. Several tidally-influenced systems were selected for study in each basin based upon this initial reconnaissance. Sites within each of the four basins included at least one small headwater “inside” tributary and a downstream contiguous “outside” sample area (larger river or estuary embayment).

The tributaries were further classified into the following general category types (Malkin et al. 2007; Figure 2):

- Minor tidal tributary – Systems whose stream channel originates in a freshwater watershed above sea level and descends toward sea level until it reaches a drowned river valley, an embayment or coastal sea. These systems tend to be strongly affected by variations in freshwater inflow.
- Tidal creek – Systems created by tidal flooding and drainage that do not drain substantial areas of land above sea level. These systems tend to import dissolved nutrients from the seaward direction and export dissolved and particulate organic material seaward (Rivera-Monroy et al. 1995).
- Dredged inlet – Systems that do not drain substantial areas of land above sea level and that have been artificially deepened through dredging activities. These systems tend to share the characteristics of adjacent waters with the degree of similarity being determined by the relative influence of adjacent mixing and the dredged area's local watershed.

A geographic information systems (GIS) analysis was performed to assess watershed landscape development intensity (LDI) (Brown and Vivas 2005) and imperviousness in each tidal tributary subbasin. Tidal tributary flowlines and waterbody attributes with direct hydrologic connection to the priority tidal tributary watersheds were extracted from the National Hydrography Dataset (NHD) obtained from the U.S. Geological Society (USGS 2007). One-hundred meter corridors around the NHD flow line and connected waterbody attributes were created using ArcGIS 9.2's (ESRI, Inc. 2006) buffer tool. Land use/land cover GIS data (SWFWMD 2004) and USGS % impervious surface area estimates (USGS 2001) were clipped along the 100-m stream corridors for further processing. Area-weighted average imperviousness and LDI (derived from land use/land cover) along the stream corridors and within the subbasin were then determined for each priority tidal tributary (Table 2).

Water & Sediment Quality

Monthly water quality samples were collected by the Environmental Protection Commission of Hillsborough County and Manatee County Environmental Management Division. Total annual sampling effort in each system is listed in Table 1. Samples included instantaneous measurements of physical water quality parameters (temperature (°C), dissolved oxygen (mg/L), conductivity (µmhos/cm), salinity (PSU), and pH) from the bottom, mid-depth, and surface of the water-column, if water depths were greater than 0.5m in the tributaries. Concurrent samples were collected with a Horizontal Alpha Bottler sampler at ≤0.5-m depth for laboratory analysis. The core group of laboratory analytes included: color (PCU), turbidity (NTU), total suspended solids (TSS, mg/L), chlorophyll-a (µg/L), total Kjeldahl nitrogen (TKN, mg/L), dissolved ammonia nitrogen (mg/L), dissolved organic nitrogen (DON, mg/L), dissolved nitrate + nitrite nitrogen (mg/L), dissolved orthophosphate (soluble reactive phosphorus, SRP, mg/L), total phosphorus (mg/L), reactive silica (mg/L), fecal coliform bacteria (# of col./100mL), and enterococci bacteria (# of col./100mL).

Seasonal benthic biota and sediment samples were collected within the tidal tributaries using a petite ponar grab (0.023m²) and in the adjacent rivers or embayments using a Young grab sampler (0.04 m²) during Spring (May 2006) and Fall (September 2006) periods following the field protocols outlined in Courtney et al. (1995) and Greening (2006). Total annual sampling effort in each system is listed in Table 1. Benthic biota were identified to the lowest practical taxon and enumerated for each sample. Species counts were standardized to the approximate area sampled. For sediment quality analyses (laboratory analysis for metals and organic contaminants), the upper 2-cm of sediment was removed from a separate grab sample (taken proximal to the biota sample) following the methodologies outlined in Greening (2006). Chemical contaminants and

characteristics analyzed included selected EPA priority pollutant metals, selected EPA priority pollutant organic compounds, % silt-clay content, and benthic microalgae (BMA) community biomass (as indicated by chlorophyll-a concentrations in the upper 1-cm of substrate).

Nekton Resources

Nekton were collected monthly (see Table 1 for total annual effort) along the tidal tributary and adjacent riverine or embayment shorelines using a 9.1-m, 3.2-mm stretched mesh, center-bag seine with leads spaced every 150 mm. Water depths were limited to ≤ 1.0 m at the bag and ≤ 0.5 m at the shore (≤ 1.0 m for seawall shorelines). The seine was set from a small, floating raft ('boogie board') in a shallow arc by a walking crewmember, while a second crewmember held one end of the seine against the shoreline. The seine was then hauled ashore from both ends, keeping the lead line as close to the bottom as possible and so forcing the catch into the center bag. Each 9.1-m seine set sampled an area of ~ 10.1 m² from the shoreline to approximately 1.5 m away from shore and covered ~ 7.4 m of shoreline. Instantaneous measurements of physical water quality parameters and qualitative habitat assessments were recorded during the time of sampling and related to nekton metrics. A subset of fish species were retained for additional gut analyses. All species collected and prey contained in sub-sampled gut contents were identified to the lowest practical taxon.

Nitrogen Sources & Related Trophic Ecology

Fish were collected from all sites for stable isotope analysis (see Table 1 for total annual effort). These samples consisted of trunk sections that did not contain internal organs. Additional materials were collected to investigate nitrogen linkages at the Frog Creek, Grassy Creek, and Curiosity Creek sites. These included samples of particulate organic matter (POM, a proxy for phytoplankton), benthic microalgae (BMA), mangroves (MAN), emergent vegetation (EM), upland trees (UP), sedimentary organic matter (SOM) and sedimentary porewater ammonium (SedNH₄⁺) directly from the tributary systems themselves, the vegetation associated with their shorelines, or in their adjacent watersheds. The isotopic values for $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ were determined using a Carlo Erba 2500 Series I elemental analyzer equipped with a zero-nitrogen blank autoanalyzer and coupled to a Finnigan Mat Delta Plus XL stable-isotope mass spectrometer. All samples were run in duplicate. Instrumental standard deviation remained under 0.3‰ for nitrogen and under 0.5‰ for carbon. To explore the interactions between ecosystem nitrogen and watershed land use, the average nitrogen isotopic compositions of fish were related to Florida Land Use and Cover Classification System (FLUCCS) data for each site using a land-use mixing model.

MAJOR RESULTS

Sherwood (2008) summarized the major findings stemming from the multi-disciplinary research conducted for this project and as described in detail in Malkin et al. (2007), Sherwood et al. (2007), and Greenwood et al. (2008). Listed below are the integrated results from those studies categorized according to the major themes developed by the research team.

Water and Habitat Quality

- Dissolved oxygen and salinity conditions within most of the tributaries were adequate to support a variety of estuarine biota for much of the sampling period; however, instances of hypoxia and supersaturated oxygen conditions were observed in more altered systems, particularly those tributaries draining to the Alafia River.
- Small tidal tributaries generally had higher nutrient concentrations than adjacent larger tidal rivers and nearshore bay waters to which they drained (Figure 3), but levels of sediment contaminants were generally lower within the tidal tributaries.

- Isotopic results indicated that benthic microalgae (BMA) provide the basis of food webs in small tidal tributaries and that there were seasonal shifts in the production of BMA versus the overlying water-column within the majority of the small tidal tributaries sampled during 2006 (Figure 4).
- Sediment organic material was dominated by vascular plant material, but BMA recycle ammonium much more quickly and is the major source of ammonia within the tidal tributary systems, particularly in the dry season (Figure 5).
- Tidal tributaries provided BMA a low-energy environment with organic accumulation for sustained production.

Benthos and Nekton Use of Tidal Tributaries

- A tributary's proximity to the bay was more important than season or whether a fish was collected in a small tributary or an adjacent larger waterbody in describing nekton diet and community structure (Figure 6).
- For fish, the system-scale (i.e., the bay, river, minor tributaries and backwater systems) was determined to be the appropriate management scale for defining essential fish habitat. Both tidal tributaries and larger adjacent waterbodies were needed to support the suite of species living in these estuarine areas.
- The overall nekton community was dominated by resident species within the Tampa Bay tidal tributaries studied in 2006 (Table 3).
- Minor tidal tributaries appeared to be critical as "food factories" (especially with BMA-based sources) as well as the more traditional habitat nursery areas for fish. These areas may also serve as nekton feeding grounds for "keystone prey species" (amphipod and mysid species) that were not as readily available in mainstem rivers and nearshore bay areas during certain seasons (Figure 7).
- Differences in fish diet were very small between tributaries and adjacent habitats, much smaller than differences between seasons or different tributaries. Diets reflected location of feeding in relation to geographic and physico-chemical features of the environment. Fish feeding in the upstream, low-salinity tributaries tended to have diets composed of prey typical of more freshwater habitats (e.g., insects).
- Isotopic signatures from fish guts can indicate the degree of residency of the fish. Diets of fish collected in smaller tributaries upstream are based on watershed sources of nitrogen, while diets of fish collected in larger tributaries downstream are based on nitrogen originating in Tampa Bay. Resident fish obtained their food biomass (primarily BMA) from small tidal tributaries (Figure 8).
- Common snook abundance was 2 to 36 times greater in small tributaries than in adjacent larger rivers or nearshore bay waters (Figure 9). This was the most economically important species of nekton that consistently showed this pattern.
- Nitrogen, the limiting nutrient within the studied systems, appeared to follow this trophic pathway within tidal tributaries: from sedimentary ammonia internally recycled from phytoplankton production in the "wet season" to BMA production in the "dry season" to benthic intermediates to fish (Figure 8).

Watershed Connectivity to Tidal Tributaries

- Location of the tidal tributaries both on a micro- (i.e. along a riverine or embayment gradient) and on a macro- (i.e. within the Tampa Bay estuary) geographic scale influenced the overarching variation in abiotic and biotic conditions observed in the tidal tributaries during 2006.
- Most biota collected used both minor tidal tributaries and larger adjacent waterbodies, so managing both connected systems is vital for estuary-dependent resources important to society.

- Land use and watershed metrics within 100-m corridors along the tidal tributaries studied were more closely associated with tidal tributary abiotic conditions than watershed metrics encompassing the tidal tributaries' full drainage catchment (Figures 10 and 11).
- Landscape development intensity along the tributaries' immediate corridors was more closely associated with in-situ water (Figure 10) and sediment quality (Figure 11) conditions than with the tributaries' biotic resources (i.e., benthos/nekton diversity and abundance). However, there was some indication that exotic nekton species may prefer more altered tributaries (Figure 12).
- Landscape development within the watersheds and corridors of the tidal tributaries studied did not adequately represent end-member systems (i.e. minimally and highly altered systems) and may explain the lack of more significant association with the biotic resources of the creek systems.
- Watershed physical characteristics and tidal tributary position along the estuarine continuum appear to have influence on tidal tributary biota largely through changes in tributary 'flashiness' (or freshwater inflow variability) and salinity conditions within the systems.
- Better methods need to be developed to quantify and characterize the stream morphology of tidal tributaries with low hydraulic gradients.

PRELIMINARY RESTORATION AND PROTECTION TARGETS

Based on the results from this project, the research team evaluated seven indicators as potential measurable targets to monitor and assess the ecosystem function of Tampa Bay tidal tributaries. Preliminary restoration and protection targets were identified for two indicators:

- Presence of common snook in tidal tributaries was an indicator of habitat connectivity among Tampa Bay open bay waters, riverine habitats, and minor, tidally-influenced systems. Achieving a high density or encouraging the recruitment and survival of juvenile common snook in restored or protected tidal tributary areas could serve as a positive indicator of ecosystem functionality for these systems.
- Maintain current State of Florida dissolved oxygen standards as measurable, protective water quality targets for Tampa Bay tidal tributaries to protect fish and benthic production. Until more site-specific research is conducted within Tampa Bay tidal tributaries, dissolved oxygen levels should not average less than 5 mg/l over a 24-hour period and should never fall below 4 mg/l in restored or protected tidal tributaries.

Five other indicators were identified as strong candidates for the development of measurable targets; however, the research team concluded that these indicators would require further evaluation in more Tampa Bay tidal tributaries prior to identifying specific targets. The potential targets and the necessary actions needed to refine their use are described as follows:

- Maintaining a balanced ratio of benthic algal production to water-column algal production according to natural and/or seasonal fluctuations. Skewed ratios could serve as a potential indicator of tidal tributary eutrophication and/or disruption of hydrology.
- Identifying key tidal tributary sediment characteristics as a proxy for potential benthic degradation associated with adjacent land use impacts. Landscape development intensity along the 100-m corridors of the tidal tributaries was positively correlated with sediment silt-clay percentage and sediment contaminants which may indicate a tendency for altered systems to become muddier and more prone to the deposition of contaminants over time.
- Refining the observed relationships between the Landscape Development Intensity Index for 100-m stream corridors and key indicators describing degraded water and sediment quality in tidal tributaries.

- Monitoring exotic species presence or abundance in tidal tributaries could serve as a proxy for the loss of ecosystem function. Nekton results suggested that exotic species may provide an indicator for tidal tributaries whose watersheds have been altered; however, other confounding physical factors precluded the direct association of exotic species with watershed alteration. Exotic benthic species were also not associated with tidal tributaries whose watersheds were determined to be more altered.
- Determining whether other watershed metrics are associated with benthic biota and nekton condition. Although general declines in biotic metrics were observed for altered watersheds with greater LDIs or imperviousness, no statistical links were determined. Performing similar analyses in tidal tributaries that represented greater extremes, in terms of perceived watershed alteration, could improve the associations and lead to the development of biotic indicators that are associated with watershed condition.

RESULTING MANAGEMENT RECOMMENDATIONS

In 2006, the Tidal Tributary Habitat Initiative was initially incorporated into Action Plan FW-6 (Preserve the Diversity and Abundance of Bay Wildlife) of the Comprehensive Conservation and Management Plan (CCMP) as a priority research need. As a next step stemming from the results of this project, the research team developed recommended management actions that could be incorporated into future updates of the TBEP's CCMP. The four preliminary management actions developed as a result of the research conducted during the 2006 Tampa Bay Tidal Tributary Habitat Initiative include:

- Maintaining connectivity between open bay waters, tidal rivers and smaller, tidal tributaries to allow fish movement, water flow and nutrient flux. Fish require a mosaic of habitats throughout their life cycle, and management should be based on a system-wide scale. This concept is important in terms of the landscape's connectivity to the surface waters, as well as for processes internal to the hydrologic landscape that promote desirable conditions within the tributaries for biotic resources.
- Reducing "flashiness" of water flow to tidal tributaries, to promote natural hydrologic regimes and foster productivity of BMA and trophic intermediates. Maintaining and restoring natural wetlands, marshes and riparian corridors in tidal tributary watersheds, as well as considering alternative best management practices for retaining and gradually releasing stormwater from the connected watershed to small tidal tributary systems, are primary recommendations to reduce flashiness in these systems.
- Continuing to track the nursery function and habitat condition of Tampa Bay tidal tributaries by monitoring freshwater inflow, watershed development, water quality indicators and nekton habitat use. This recommendation would allow for broader-based studies and cataloging of minor tidal tributary systems whose functions are not yet defined.
- Improving public education and stewardship of tidal tributaries by promoting an Adopt-a-Creek program. Tidal tributaries are habitually overlooked in management scenarios and promoting their importance as sentinel habitats in the public perception can lead to an improved management response.

The TBEP has been working with partners to further refine these management actions and determine whether the actions are relevant to the majority of tidal tributaries in the Tampa Bay watershed. Further, the research team concluded that a major monitoring effort would be needed to catalog the existing habitat and fisheries value of tidal tributaries that have not yet been studied. In response and planned for initiation in 2010, the TBEP will conduct a research survey that is focused on: 1) cataloging the location and extent of salinity barrier structures within Tampa Bay tidal tributaries; and 2) determining the feasibility, implications, and habitat enhancement benefits

of their removal. This research project is intended to identify and prioritize tidal tributaries where habitat restoration would improve connectivity with the watershed and the downstream estuary while reducing the impacts of “flashy” hydrology on the estuarine continuum.

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Table 1. Classification and sampling effort for tidal tributary systems sampled during the 2006 Tampa Bay Tributary Habitat Initiative.

Major Watershed	Tributary	Tidal System Type	Water Quality Samples	Benthic Samples	Nekton Samples	Isotopic Samples
Alafia River (a priori more altered)	"Dog Leg Creek"	Dredged Inlet	12	2	48	.
	"Question Mark Creek"	Dredged Inlet	12	3	48	36
	Rice Creek	Minor Tributary	16	4	48	26
	"Riverview Park West Creek"	Dredged Inlet	12	3	48	38
	Mainstem Alafia River	Major River	160	18	168	.
Little Manatee River (a priori less altered)	Curiosity Creek	Minor Tributary	28	6	120	71
	Wildcat Creek	Minor Tributary	28	6	120	121
	Other Monitored Tributaries	Minor Tributary	7	.	.	.
	Mainstem Little Manatee River	Major River	12	12	144	.
Feather Sound (a priori more altered)	Grassy Creek	Tidal Creek	23	12	96	92
	Other Monitored Tributaries	Tidal Creek	10	.	.	.
	Adjacent Old Tampa Bay	Embayment	137	14	52	.
Terra Ceia Bay (a priori less altered)	Frog Creek	Minor Tributary	24	6	217	84
	McMullen Creek	Minor Tributary	12	6	82	83
	Adjacent Terra Ceia Bay	Embayment	12	11	102	.

Table 2. Summary of watershed metrics for the tidal tributaries studied.

Major Watershed	Tidal Tributary	Basin Size (ha)	% Impervious Cover (USGS 2001)		Landscape Development Intensity Indices (SWFWMD 2004)	
			Basin (%)	100-m Corridor (%)	Basin (Indices)	100-m Corridor (Indices)
Alafia River	"Dog Leg Creek"	35.5	2.55	2.13	6.04	3.05
	"Question Mark Creek"	23.2	7.04	2.35	5.68	3.96
	Rice Creek	1315.4	9.81	6.9	4.91	3.72
	"Riverview Park West Creek"	68.2	16.57	6.93	6.22	6.43
	Average	360.6	9.0	4.6	5.7	4.3
Little Manatee River	Curiosity Creek	5725.9	0.9	0.58	3.19	3.01
	Wildcat Creek	926.3	1.89	0.83	3.2	2.48
	Average	3326.1	1.4	0.7	3.2	2.7
Feather Sound	Grassy Creek	159.9	4.24	1.18	1.62	1.08
Terra Ceia Bay	Frog Creek	2357.2	6.65	3.12	4.11	3.31
	McMullen Creek	757.5	13.89	9.12	5.16	4.79
	Average	1557.4	10.3	6.1	4.6	4.1

Table 3. The 25 most abundant fish and invertebrate species collected from 827 9.1-m seine samples in nine Tampa Bay tidal creeks. R=resident species (found in tidal creeks throughout their lives), T=transient species (found in tidal creeks for only part of their lives), REC=species targeted by recreational fishers, COM=species targeted by commercial fishers. Main predators are birds (B), fishes (F), or marine mammals (M).

Common name	Scientific name	Number collected	Cate-gory	Known Predators	Fisheries
Daggerblade grass shrimp	<i>Palaemonetes pugio</i>	40,620	R	B, F	-
Rainwater killifish	<i>Lucania parva</i>	16,373	R	B, F	-
Bay anchovy	<i>Anchoa mitchilli</i>	14,744	T	B, F	-
Menidia silversides	<i>Menidia</i> spp.	11,812	R	B, F	-
Eastern mosquitofish	<i>Gambusia holbrooki</i>	10,297	R	B, F	-
Sailfin molly	<i>Poecilia latipinna</i>	4,355	R	B, F	-
Striped mullet	<i>Mugil cephalus</i>	2,735	T	B, F, M	COM/REC
Hogchoker	<i>Trinectes maculatus</i>	1,871	T	B, F	-
Sheepshead minnow	<i>Cyprinodon variegatus</i>	1,791	R	B, F	-
Clown goby	<i>Microgobius gulosus</i>	1,702	R	B, F	-
Eucinostomus mojarra	<i>Eucinostomus</i> spp.	724	T	B, F	-
Brackish grass shrimp	<i>Palaemonetes intermedius</i>	715	R	B, F	-
Gulf killifish	<i>Fundulus grandis</i>	699	R	B, F	-
Goldspotted killifish	<i>Floridichthys carpio</i>	518	R	B, F	-
Naked goby	<i>Gobiosoma bosc</i>	428	R	B, F	-
Seminole killifish	<i>Fundulus seminolis</i>	355	R	B, F	-
Tidewater mojarra	<i>Eucinostomus harengulus</i>	234	T	B, F	-
Common snook	<i>Centropomus undecimalis</i>	225	T	B, F	REC
Bluefin killifish	<i>Lucania goodei</i>	211	R	B, F	-
Spot	<i>Leiostomus xanthurus</i>	202	T	B, F, M	REC
Least killifish	<i>Heterandria formosa</i>	199	R	B, F	-
Coastal shiner	<i>Notropis petersoni</i>	193	R	B, F	-
Blue crab	<i>Callinectes sapidus</i>	185	T	B, F	COM/REC
Diamond killifish	<i>Adinia xenica</i>	143	R	B, F	-
Marsh killifish	<i>Fundulus confluentus</i>	141	R	B, F	-

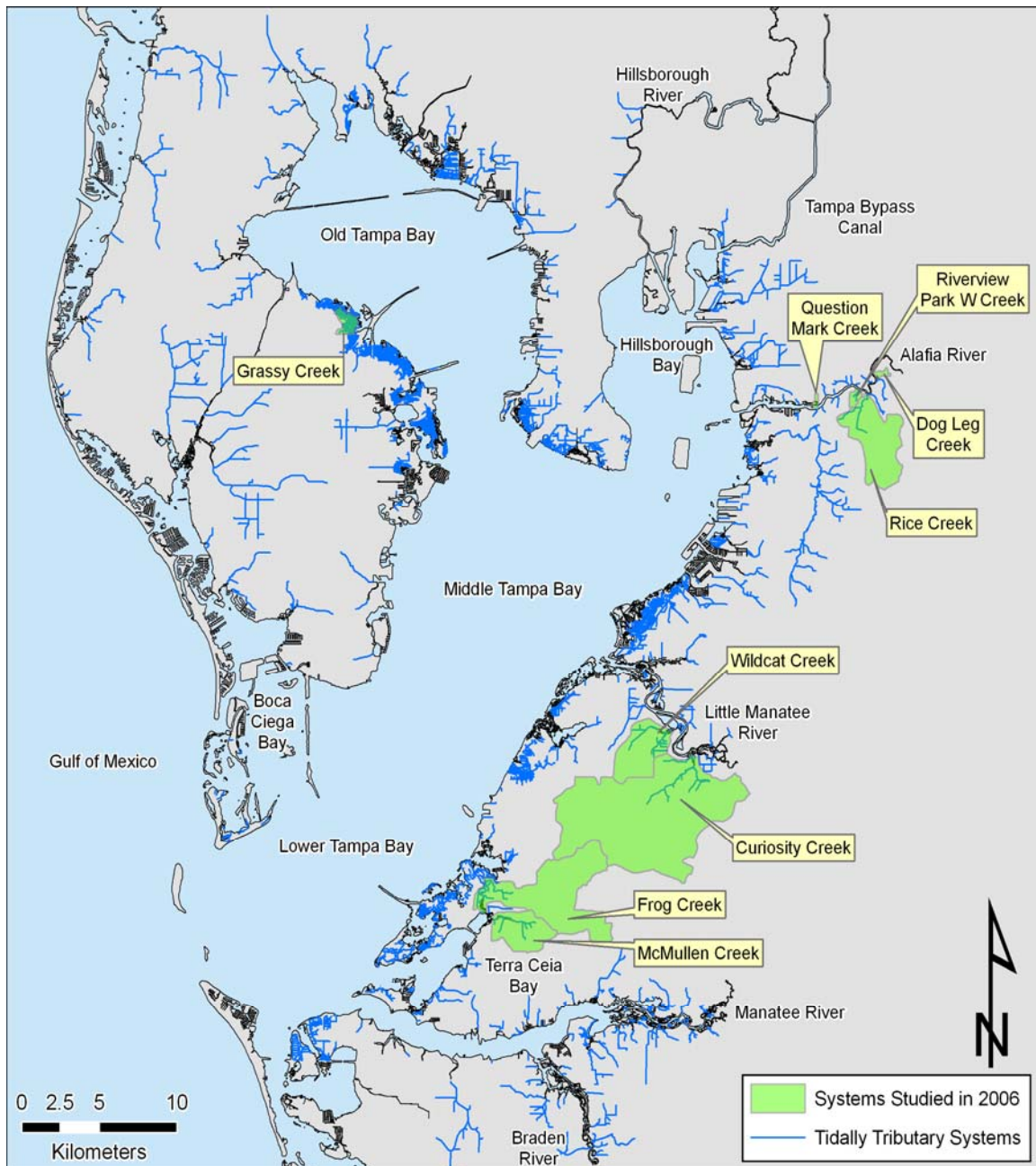


Figure 1. Schematic of minor tidally-influenced tributaries, creeks, ditches, and canal systems in the Tampa Bay area as identified from the USGS National Hydrography Dataset (adapted from Lewis 1996). The basins of the tidal tributary systems studied in 2006 are highlighted.

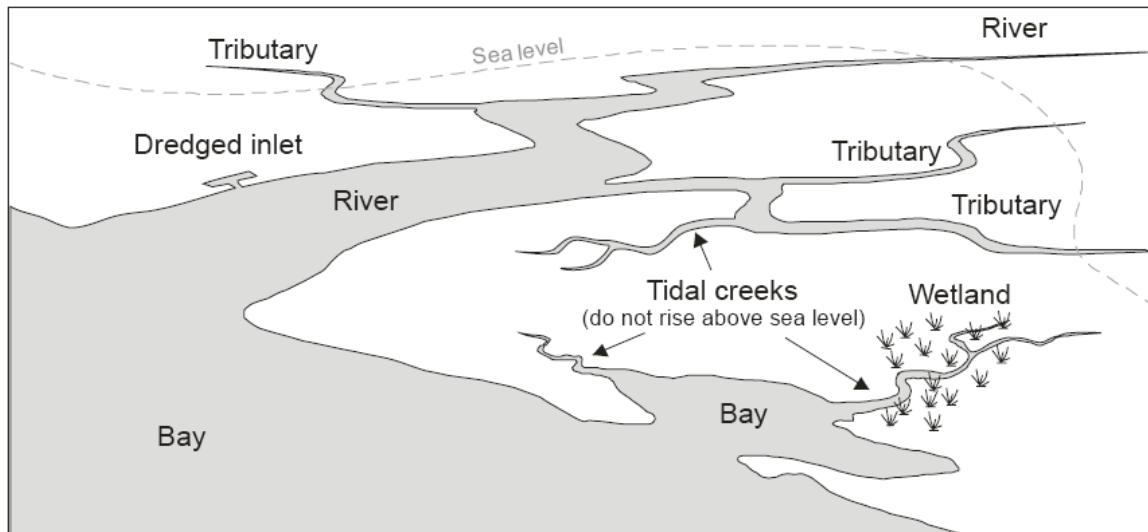


Figure 2. Schematic diagram of various “tidal tributary” systems typically encountered in the Tampa Bay estuary. Well-studied areas in Tampa Bay include the major bays and rivers.

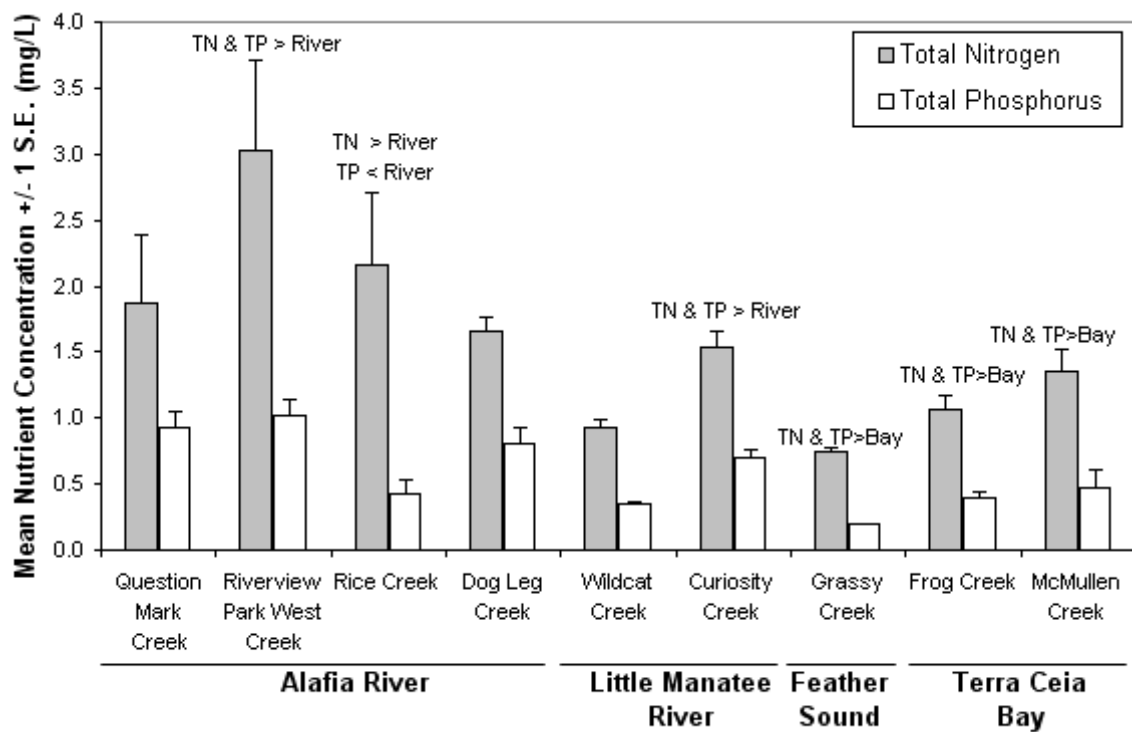


Figure 3. Mean nutrient concentrations within the tidal tributaries during 2006. Significant differences ($P < 0.05$) in total nitrogen (TN) and total phosphorus (TP) concentrations between the tidal tributaries and the areas to which they drained are designated above the respective bars (Sherwood et al. 2007).

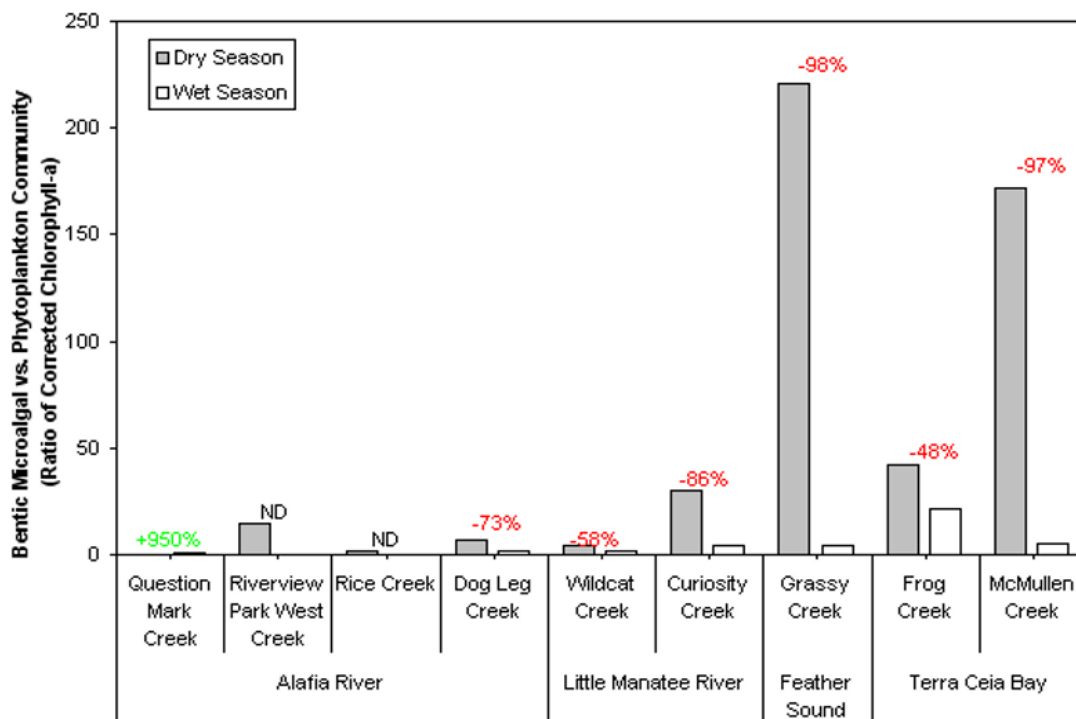


Figure 4. Relative change (%) in seasonal ratios of benthic:phytoplankton algal communities in the tidal tributaries during 2006 (Sherwood et al. 2008).

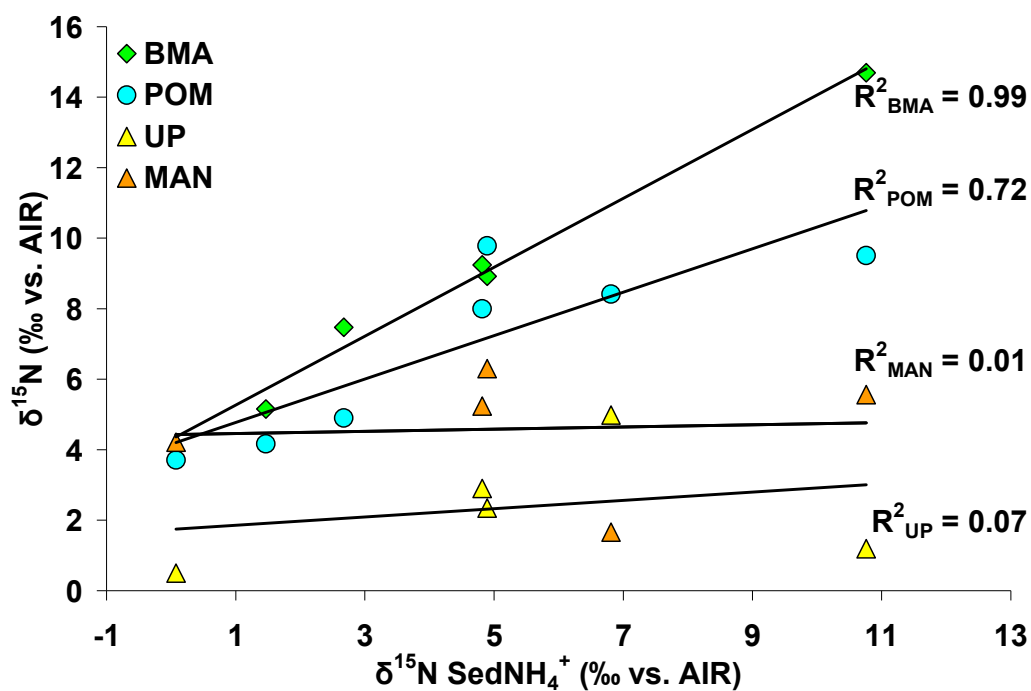


Figure 5. Plot of site-averaged $\delta^{15}\text{N}$ values of benthic microalgae (BMA), particulate organic matter (POM), mangroves (MAN), and upland plants (UP) against $\delta^{15}\text{N}$ values of sedimentary pore water ammonium (SedNH_4^+) during the dry season. The strongest relationship is observed for BMA. (Malkin et al. 2007)

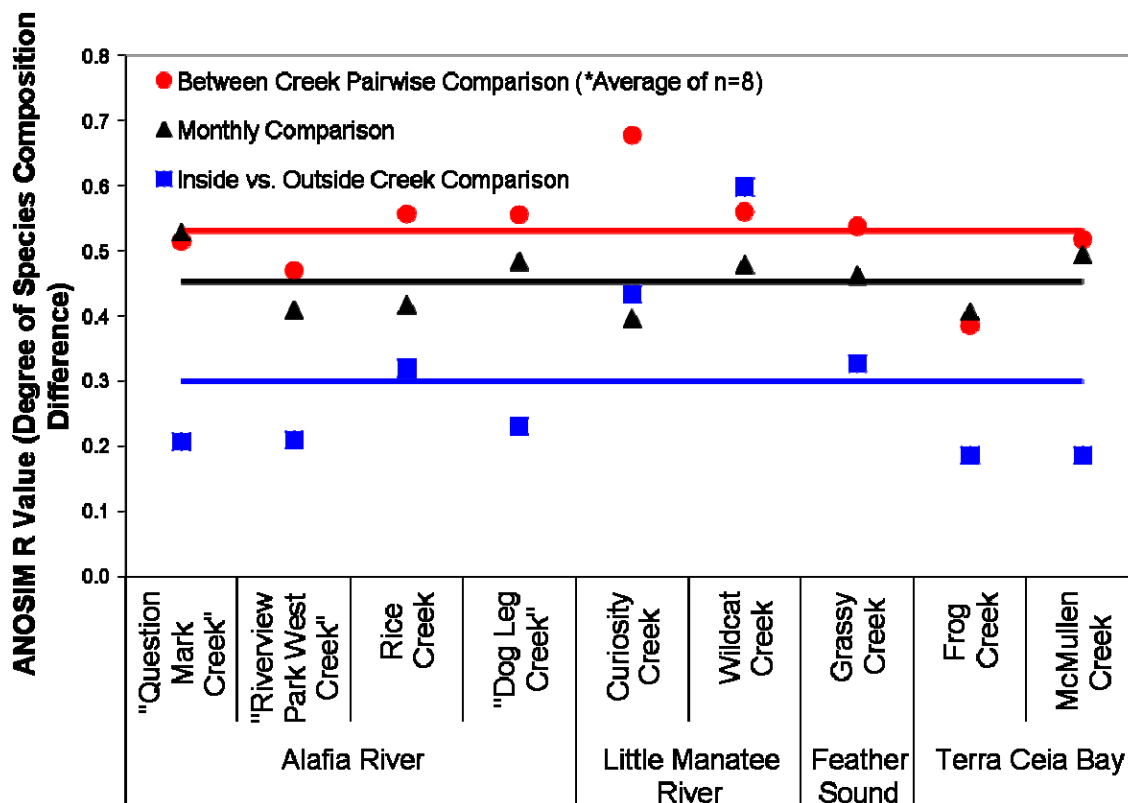


Figure 6. Comparison of nekton species composition between the different tidal tributaries sampled (red circles and line), among months sampled (black triangles and line), and between the tidal tributaries and the adjacent riverine or embayments to which they drained (blue squares and line). The red line indicates the average species composition difference among all the creeks sampled, while the black line indicates the average monthly difference and the blue line indicates the average creek vs. drainage area difference (i.e. inside creek vs. outside creek species composition differences). Higher Analysis of Similarity (ANOSIM) R-values indicate greater differences in species composition (Greenwood et al. 2008).

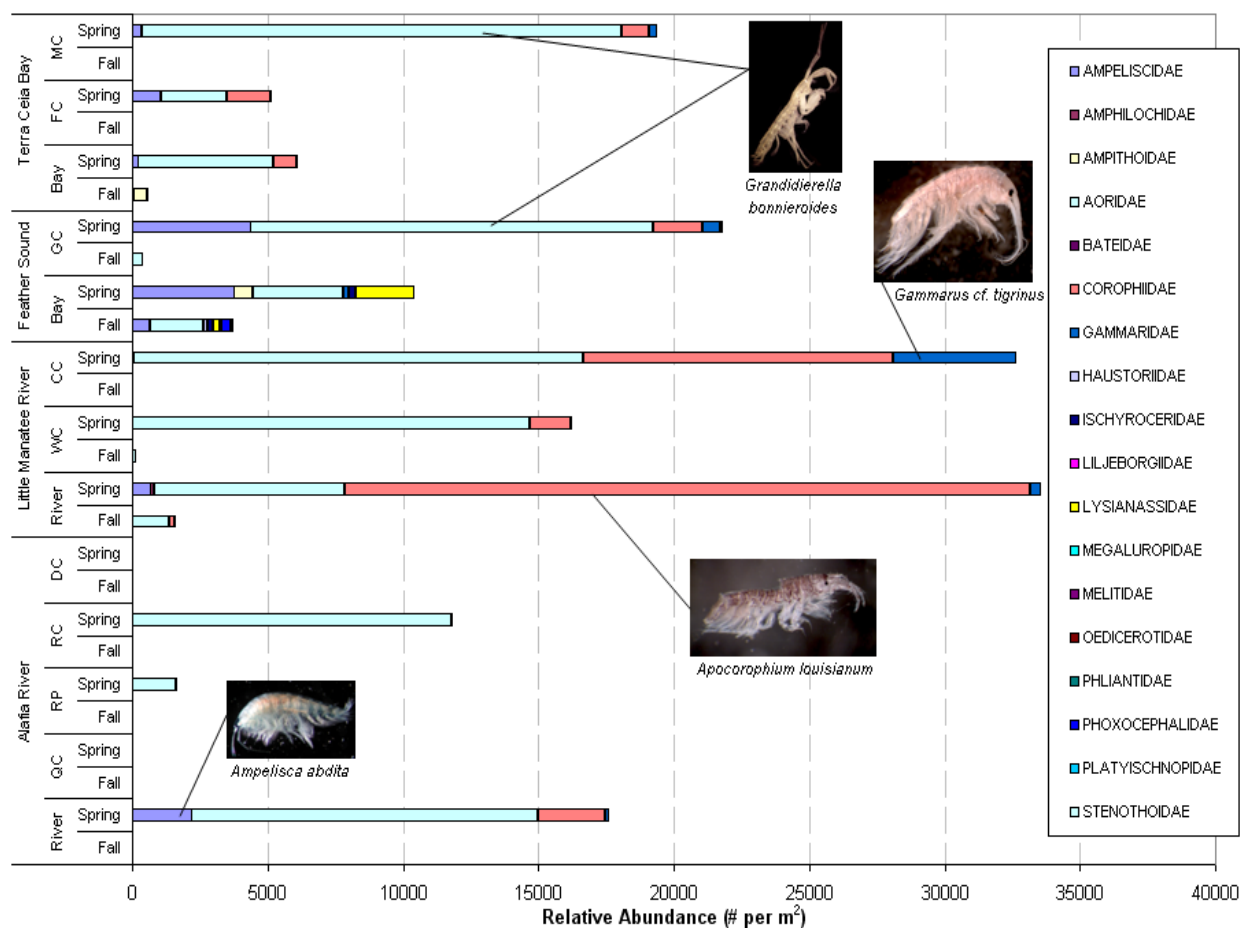


Figure 7. Seasonal amphipod abundance in tidal tributaries studied during 2006. MC=McMullen Creek, FC=Frog Creek, GC=Grassy Creek, CC=Curiosity Creek, WC=Wildcat Creek, DC= "Dog Leg Creek," RC=Rice Creek, RP= "Riverview Park West Creek," QC= "Question Mark Creek" (Sherwood et al. 2008).

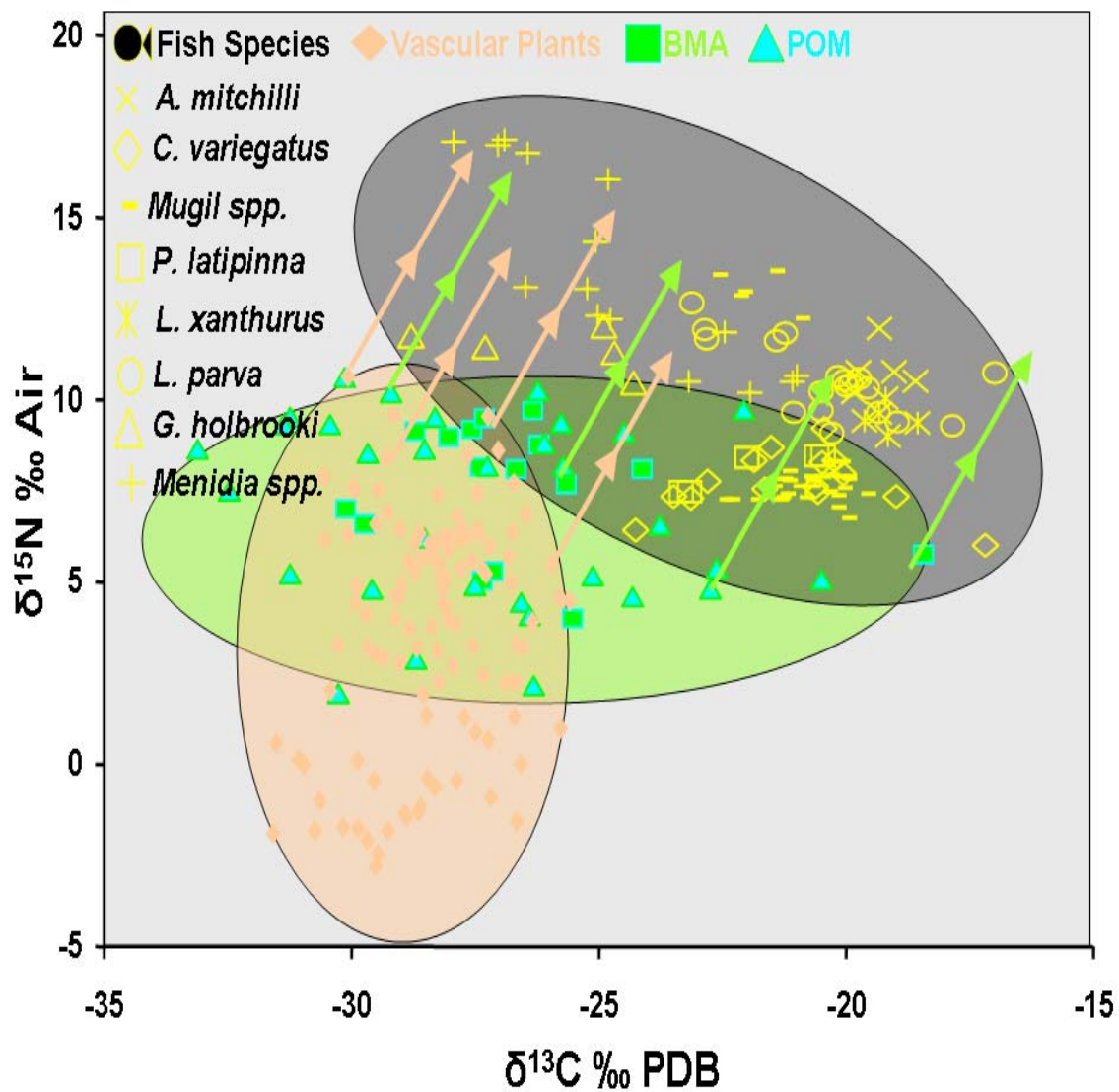


Figure 8. Isotope comparisons between common nekton species found in tidal tributaries and primary producers. Most species appear to be supported by benthic microalgae (BMA) and water-column phytoplankton (POM, either direct or recycled through sedimentary ammonium sources) (adapted from Malkin et al. 2007).

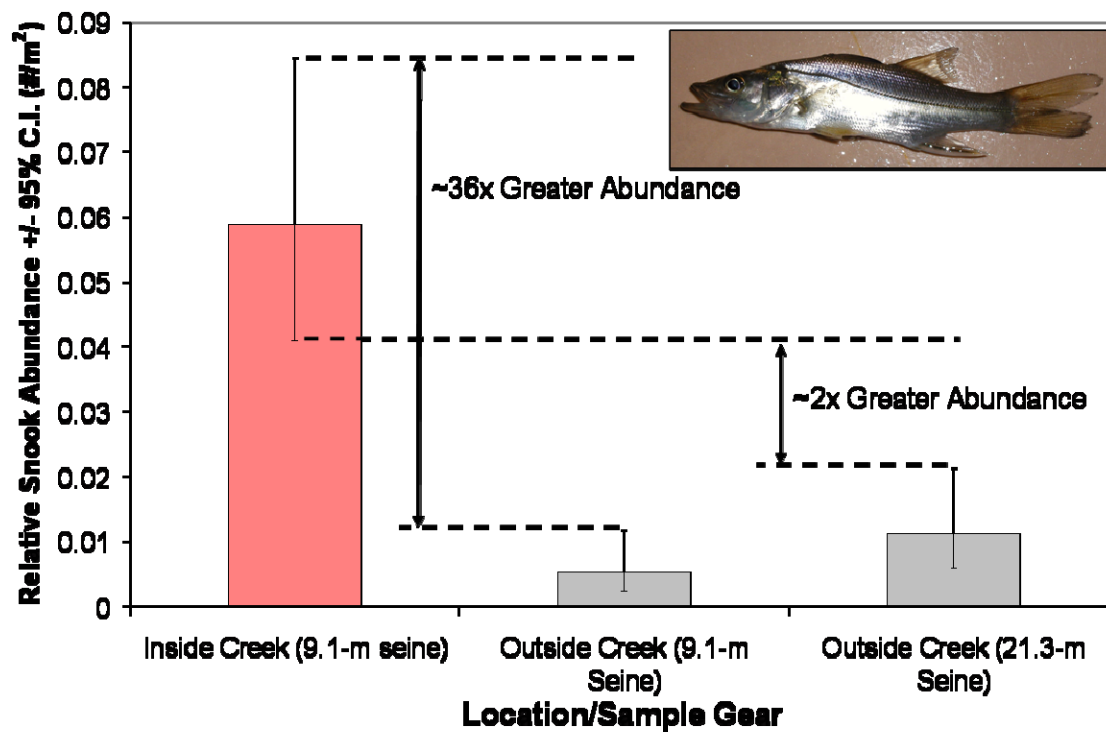


Figure 9. Comparison of snook abundance (number per m²) inside the sampled tidal tributaries relative to riverine or embayment areas adjacent to the tributaries (Greenwood et al. 2008).

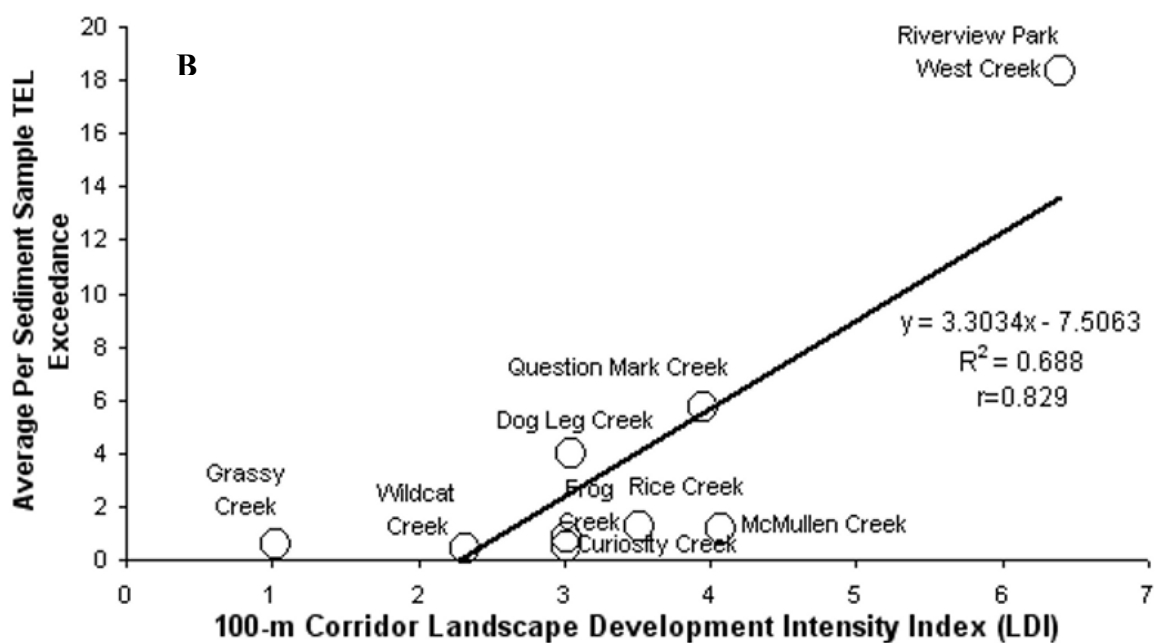
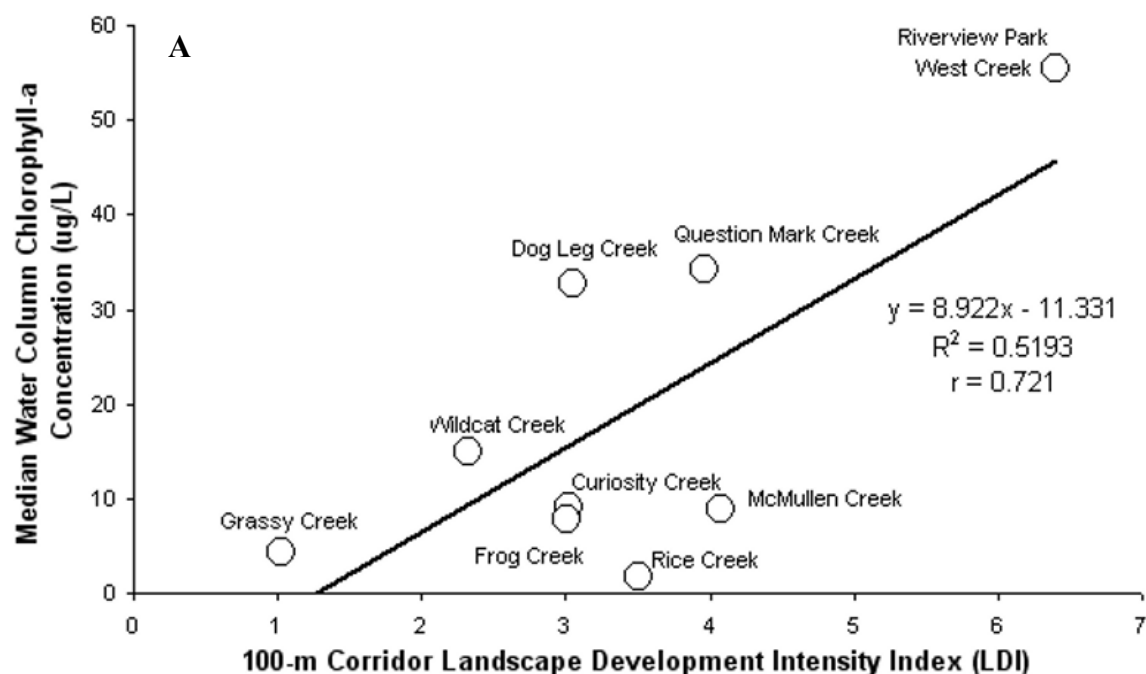


Figure 10. Relationship of landscape development intensity along the tributaries corridors and median observed chlorophyll-a concentrations (A) and average number of threshold effects level sediment contaminant exceedances (B) in the tidal tributaries during 2006.

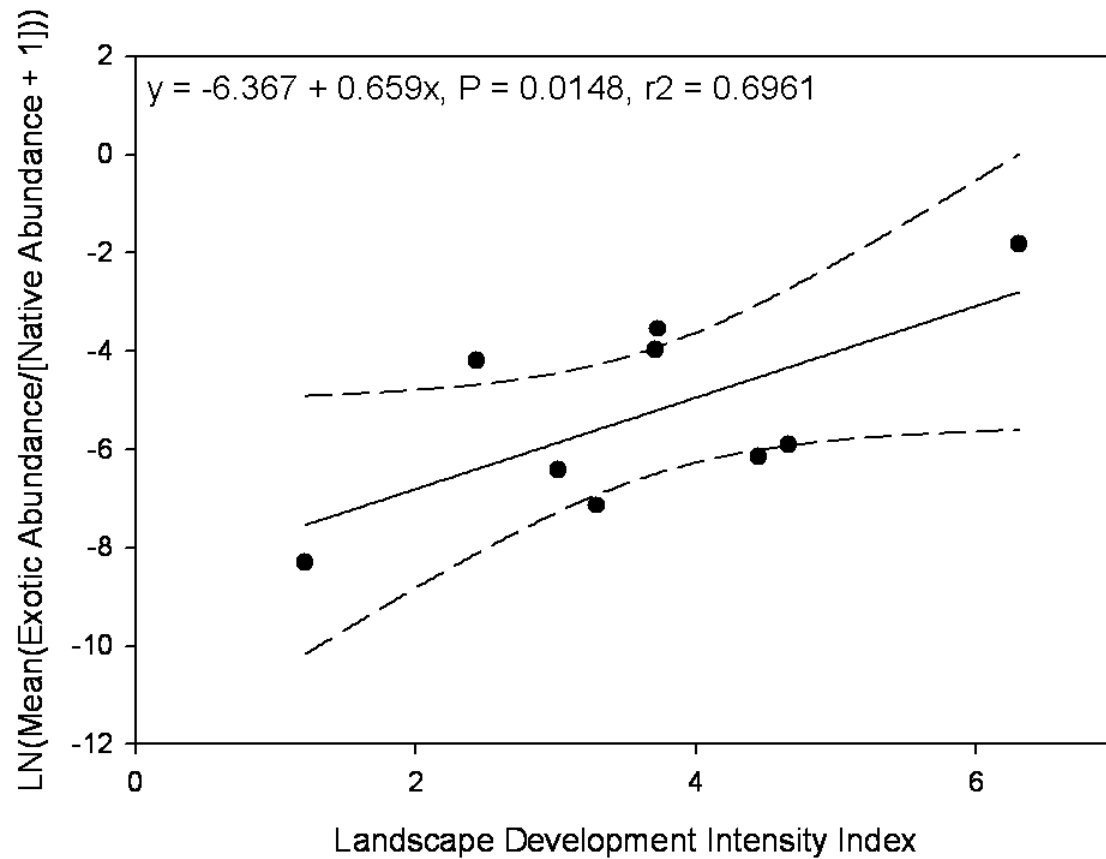


Figure 11. Relationship of exotic:native nekton species richness to landscape development intensity in the tidal tributaries (Greenwood et al. 2007).

DEVELOPING AN INDEX OF PHYTOPLANKTON BIOMASS FOR TAMPA BAY'S LOWER ALAFIA RIVER

M.R. Wessel & A.J. Janicki

ABSTRACT

Phytoplankton production and deposition is a major source of organic carbon that drives secondary productivity in tidal river systems supporting a base in the food chain for higher trophic levels. Tampa Bay's Alafia River is one of the most nutrient-enriched rivers in southwest Florida and subsequently has some of the highest phytoplankton concentrations (as measured by chlorophyll-a) among southwest Florida tidal rivers. The objective of this project was to develop an index of chlorophyll biomass that could be used to describe the spatial variability in the distribution of phytoplankton in the estuarine portion of the Alafia River. A nonparametric model was developed to estimate a chlorophyll response surface as a smooth multivariate function of inflow and location within the river. Development of this model was a necessary first step in more expansive effort to characterize the spatial concordance between phytoplankton biomass, and planktonic and benthic invertebrate organism distribution and abundance in the lower Alafia River. Previous efforts to describe the chlorophyll biomass in the lower Alafia River have been limited to simple logistic regression models used to predict the probability of a chlorophyll threshold (e.g. $15 \mu\text{g l}^{-1}$) exceedance, or linear regression models used to predict the location of the maximum chlorophyll value in the river. For this study, we expand on previous efforts by developing a model capable of estimating the biomass at any point in the lower river. Model predictions were generated to estimate chlorophyll concentrations at 0.5-kilometer intervals throughout the lower Alafia River. This spatially-explicit model compared well against both existing logistic regression models and previously-developed models describing the location of the chlorophyll maximum. Identified advantages of the spline model included the ability for predictions to be integrated over various time scales to derive an index of chlorophyll biomass describing the spatial-temporal variability in the distribution and biomass of phytoplankton as a function of freshwater inflow. Linking the spatial-temporal distribution of phytoplankton biomass with the distribution and abundance of zooplankton and benthic macroinvertebrates is currently being pursued based on the index development work described in this paper.

INTRODUCTION

Primary production is a critical component of the estuarine food web (Wolanski 2007) and the Alafia River has some of the highest observed chlorophyll-a concentrations (a measure of phytoplankton) among southwest Florida tidal rivers (SWFWMD 2008). The Alafia River is nutrient-rich. High nutrient concentrations in surface waters create ecosystems which are driven by primary productivity (Ornolfsdottir et al. 2003; Caffrey et al. 2007). Deposition and decomposition of phytoplankton is a major source of organic carbon that is utilized by detrital feeding organisms in the benthos driving secondary productivity and providing a base in the food chain for higher trophic levels (Dauer et al. 2000; Herman et al. 1999). Particulate organic carbon (POC) is comprised mainly of phytoplankton but also includes bacteria, the smaller zooplankton, fecal matter, allochthonous material and phytodetritus (Pinckney et al. 2001). The presence of detrital matter and fine sediment in estuaries makes [estuarine] food webs much more complex by adding a microbial loop that is centered on organic detritus, coupling the pelagic food web with the benthic food web (Wolanski 2007).

Freshwater inflows deliver nutrients from watershed and atmospheric sources. The rate of freshwater inflow influences hydraulic residence time, and hence the time available for nutrient uptake in lotic systems. Residence time is a key physical variable determining the extent to which the health of a particular estuary is affected by human-induced stresses. Freshwater inflows

combine with tidal forces and geomorphology to create hydraulic control points and discontinuities in estuarine salinity gradients resulting in Estuarine Turbidity Maxima (ETM) and areas of increased primary and secondary production (Roman et al. 2005). Secondary production of benthic communities is affected by the amount of POC that reaches the sediment-water interface and is thereby available for consumption by benthic detritivores (Chanton and Lewis 2002; Bouillona et al. 2004; Wang et al. 2004).

Tampa Bay Water is a regional public water supply authority supplying drinking water to much of the Tampa Bay region. Tampa Bay Water's supply sources include groundwater, desalination and surface water withdrawals from the Hillsborough River, Tampa Bypass Canal and the Alafia River. A comprehensive Hydrobiological Monitoring Program (HBMP) that was established as part of Tampa Bay Water's water use permit has supported data collection efforts in the Alafia River since 2000. This sampling program is comprehensive in that it includes monthly sampling efforts for water quality, plankton, benthic macroinvertebrates, and juvenile and adult fishes. The HBMP developed a focus group that holds regular meetings to summarize data collection efforts, present results of scientific research conducted with HBMP-derived data, and discuss potential modifications to achieve monitoring program goals as set forth in the HBMP design document (PBS&J 2000).

As part of these meetings, the ecological paradigm described above was forwarded by Dr. Ernst Peebles as relevant to Tampa Bay tidal rivers including the Alafia River: namely that spatial correlation exists between areas of high primary productivity and areas of high zooplankton and benthic macroinvertebrate abundance. It is these areas of predominant phytoplankton production and deposition of POC, particularly that derived from phytoplankton, that were of interest for this study with respect to longitudinal distribution within the Alafia River and how this relates to the distribution of the zooplankton and macrobenthos.

To-date, models developed to predict chlorophyll in the lower Alafia River have been limited to either predicting the location of the peak chlorophyll value as a function of freshwater flow or the probability of exceeding some threshold chlorophyll value as a function of freshwater flow. The objective of this study was to develop an index of phytoplankton biomass that could serve as the foundation for examining the spatial concordance between phytoplankton distribution and the abundance of primary and secondary consumers in the lower Alafia River in support of the process-based hypotheses of tidal river function described above.

METHODS

Water quality data for this study were provided by the HBMP. Monthly samples beginning in May of 2000 were collected in each of strata AR1 – AR6 of the lower Alafia River (Figure 1). Each stratum in this estuarine portion of the river is approximately 2.5 kilometers in length, and two samples for each of the three monitoring elements were collected within each stratum each month. All water quality samples are collected at randomly-generated sites each month. One additional upstream freshwater stratum was also sampled monthly for water quality. Data from this freshwater stratum was used to give the statistical model relevance to upstream water quality conditions; however, the analysis for this study is focused on the estuarine portion of the river (i.e., AR1- AR6).

The analytical phase of this project required the development of an empirical model to estimate the chlorophyll-a concentration throughout the lower Alafia in such a way that predictions could be integrated over time to correlate with zooplankton and macrobenthos abundance. The model was required to have a stochastic component such that uncertainty in model predictions could be

quantified. Therefore, a statistical model was developed to estimate the concentration of chlorophyll-a as a function of inflow and location within the river. The general model formulation is expressed as:

$$Y_i = f(x_i) + e_i$$

Where:

Y_i = Chlorophyll a concentration at time of sample

$f(x_i)$ = Unknown function of location in the river and inflow at time of sample

e_i = prediction error

This model belongs to a family known as generalized additive models and the function $f(x_i)$ was not assumed to follow any particular distribution. Therefore, this model is non-parametric. To approximate this function $f(x_i)$ the thin plate smoothing spline function was used to estimate the response surface as a smooth multivariate function as described in the documentation for the TPSPLINE procedure (SAS v9.2).

Freshwater flows were estimated at the Bellshoals Road location in the Alafia River based on measured flows at the United States Geological Service (USGS) station on the Alafia River at Lithia (02301500) and Lithia Springs, with adjustments for the small ungaged portion of the river downstream (Janicki Environmental 2005). Tampa Bay Water withdrawals were subtracted from flows at Bellshoals Road to create the final flow term which was then averaged over the sample date and the previous two days to create a 3-day average flow term for use in the model. An interaction term between freshwater inflow and river kilometer was also included in the model to allow for the relationship between inflow and chlorophyll to vary depending on location within the tidal portion of the river. Model predictions were generated by seasonal blocks which were operationally defined for this study using calendar months as follows:

- Block 1 = April - June
- Block 2 = November – March
- Block 3 = July – October

Spline model predictions were then compared to the other previously used methods predicting chlorophyll in the lower Alafia River described in SWFWMD (2008). Specifically, previously developed logistic regression models were used to predict the stratum-specific probability of a chlorophyll exceedance as a function of the 3-day average freshwater flow for 15 $\mu\text{g l}^{-1}$ and 30 $\mu\text{g l}^{-1}$ chlorophyll thresholds previously established (PBS&J 2006). These regressions were stratum-specific. To compare the model predictions, the specificity and sensitivity of model predictions were calculated for the logistic regression models and the spline model for each stratum where the logistic regression model results were statistically significant (i.e., strata AR3 through AR6). Sensitivity was expressed as the percent of observed exceedances correctly predicted while specificity was defined as the percent of non-exceedances correctly predicted. The overall percent correct predictions were then calculated and compared for both chlorophyll threshold values. We also compared spline model predictions to another model developed to predict the location of the maximum chlorophyll value as a function of freshwater flow (SWFWMD 2008). This model defined the relationship as:

$$\begin{aligned} \text{Location of Chl-a max} &= 16.48 - 1.96 \times \text{Ln (3 day mean flow)} \\ R^2 &= 0.40 \quad p < 0.0001 \end{aligned}$$

After model comparisons were completed, the concentration of chlorophyll was predicted using the spline model at 0.5-kilometer intervals for the estuarine portion of the river for each daily record between January 1 2000 and September 30 2008. These predictions were restricted to the maximum flow value associated with the empirical water quality data (i.e., the data domain of the regression). The principal objective of this study required the integration of these model predictions to describe the spatial distribution of phytoplankton throughout the lower Alafia River. Therefore, an index was constructed that integrated chlorophyll predictions at each 0.5-kilometer interval over various temporal scales. This was accomplished by summing daily values at 0.5 Rkm intervals and normalizing the values to the highest single value observed over the temporal period (e.g. calendar month). This generated an index of chlorophyll biomass ranging between 0 and 1.

Several assumptions were required to translate this index to be representative of phytoplankton dynamics in the Alafia. We assumed that the chlorophyll concentration was related to phytoplankton biomass. Assuming a constant rate of deposition per unit chlorophyll allowed for this index to represent deposition of POC from phytoplankton to the benthos. These assumptions would then allow us to utilize the index values to draw inferences when correlating the spatial distribution of index values with the distribution and abundance of zooplankton and macrobenthos over varying temporal scales including month, year and seasonal blocks.

RESULTS

The Alafia River flows at Bellshoals Road reflected the large-scale variability in southwest Florida's climatological conditions over the study period (Figure 1), including: an intense drought in 2000 and early 2001; an abnormally wet winter in 2002-2003, and the landfall of four hurricanes in 2004. From late 2006 through the end of the study period, drought conditions returned to Southwest Florida and the Alafia River watershed basin. These climate variations resulted in a robust set of inflow conditions for the study period exemplifying the range of conditions likely to be encountered within the basin as well as representing the typical seasonality of Alafia River inflows. River flows at Bellshoals Road ranged from 6 cubic feet per second (cfs) to just over 11,000 cfs over the study period.

Empirical chlorophyll data collected between 2000 and 2008 displayed similar characteristics to those described elsewhere for lower Alafia River minimum flows and levels (MFL) studies (SWFWMD 2008) with tendencies for higher chlorophyll values to be recorded at lower flows and a downstream movement of the location of the maximum chlorophyll value with increasing flows (Figure 2). Our model compared well against other models developed to predict patterns in chlorophyll distribution in the lower Alafia River. Based on the empirical data, the spline model predictions had a higher predictive accuracy compared to the empirical data than the logistic regression models used to predict either the 15 $\mu\text{g l}^{-1}$ threshold or the 30 $\mu\text{g l}^{-1}$ threshold in any stratum (Table 1). The comparison included accurately predicting an exceedance when it occurred in the empirical data and also accurately predicting values below the exceedance threshold. Based on these results, the spline model had superior predictive capacity to the logistic regression models. The spline model also compared well to the location of the chlorophyll maximum with the predicted location generally within 2 kilometers (Figure 3) with no bias evident in the differences except in seasonal Block 3. In Block 3, spline model predictions suggested that the peak chlorophyll values would occur over a relatively broad area of the first 4 kilometers of the Alafia and may reflect the influence of chlorophyll concentrations in Hillsborough Bay.

Chlorophyll model predictions matched the empirical data well. Temporal variability in index values was closely related to seasonality in chlorophyll values with the location of peak values located around Rkm 7 during January and moving upstream during the low flow months of April

and May where the highest biomass of phytoplankton was estimated to occur (Figure 4). The increase in magnitude of chlorophyll concentrations and upstream movement of peak chlorophyll values coincided with the presence of a second chlorophyll peak evident near Rkm 12. With the onset of the rainy season in July, index values diminished and the peak locations were located downstream near Rkm 5. Overall predicted biomass declined during these months as a function of the reduced residence times associated with the higher inflows of seasonal Block 3. By November, the index values increased and the peak values were again located near Rkm 8.

DISCUSSION

This study makes an important contribution to the development of a predictive model that is robust and flexible in describing the distribution of chlorophyll throughout the lower Alafia River. To date, modeling tools to estimate chlorophyll biomass have focused on the location of the chlorophyll maximum or the probability of chlorophyll exceeding a specified threshold value. An advantage of the model developed for this study is the ability to predict the chlorophyll concentrations throughout the lower river and integrate those predictions for various portions of the river to develop an index of phytoplankton biomass. These results confirmed that the predominant area of chlorophyll biomass was located near Rkm 8, but that freshwater flows had a predictable effect on the location and concentration of chlorophyll in the lower river. The model captured the variation in the shape of distribution as a function of flows. For example, during the low flow season (i.e. seasonal Block 1), the shape of the distribution changed dramatically with the peak chlorophyll values located more upstream and increased likelihood for extremely high values (i.e. $>50 \mu\text{g l}^{-1}$). The distribution flattened dramatically with the onset of the wet season (Block 3) when peak chlorophyll values were much lower and located near the mouth of the river. While these features have been described previously, the spline model approach has allowed for the dynamics of location and magnitude to be predicted for the entire lower river using a single predictive tool.

The TPSPLINE procedure used in model development is a relatively new estimation tool. It has demonstrated potential for use to characterize the spatial-temporal distribution of chlorophyll in the lower Alafia River which is advantageous to other models currently developed. The spline model compared well with other models developed to predict chlorophyll in the lower Alafia River. The spline model had higher overall correct predictions for both the $15 \mu\text{g l}^{-1}$ threshold and the $30 \mu\text{g l}^{-1}$ threshold in all strata and predicted similar location of the maximum chlorophyll value to the log linear model described in SWFWMD (2008). It should be noted that all of these models have uncertainty that may be substantial, but the results of this study represent a significant advancement for current modeling approaches. More complex models such as mechanistic water quality models have not been developed to-date, and the spline model represents a step toward a better understanding of the complex phytoplankton dynamics in the lower Alafia River.

Translating the chlorophyll predictions into a biomass index will help support investigations of process-based hypotheses of tidal river ecosystem functions by correlating zooplankton and macrobenthos distribution and CPUE with the index values of various temporal scales, which is the subject of a related study currently underway. The spline model predictions were used as the basis for a biomass index that described the distributional aspects of chlorophyll biomass in the water column. This index can then be examined in correspondence to the distribution and abundance of zooplankton in the water column. Assuming a constant rate of deposition may allow for the index values to serve as a surrogate for organic deposition to the benthos; this will be investigated for spatial concordance with the distribution and abundance of macrobenthos.

The spline model remains to be validated with independent data prior to potential use as an evaluation tool within the regulatory framework for freshwater flows, but it has shown advantages in describing dynamic fluctuations observed in phytoplankton distributions in the lower Alafia River. For this study, several simplifying assumptions were necessary in order to accept that the integrated model estimates represented phytoplankton productivity in the estuary, and the best spatial and temporal timescale for model inference have yet to be determined. However, the model remains a significant contribution to understanding tendencies in phytoplankton dynamics in the lower Alafia River. The predictions suggest that areas of predominant phytoplankton biomass occur in the middle portions of the lower Alafia River near RKm 8 but there is a highly seasonal pattern to the location of peak biomass which is related to seasonal inflows.

The study described in this paper was part of an on-going effort to develop objective criteria for evaluating the effects of freshwater inflows and withdrawals from the Tampa Bay Water facility at Bellshoals Road on the estuarine biota of the lower Alafia River. This study provided a new tool that can be used for estimating the effects of inflows on chlorophyll levels throughout the lower Alafia River. Continued research in this area by validating the chlorophyll model as well as investigating the spatial independence of water quality sampling efforts in the lower Alafia River may yield a model with greater predictive capacity to estimate phytoplankton population dynamics and energy transfer to higher trophic levels in the lower Alafia River.

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Table 1. Correct prediction percentages for logistic regression and spline model predictions by stratum in the Lower Alafia River.

Stratum	15 $\mu\text{g l}^{-1}$ threshold		30 $\mu\text{g l}^{-1}$ threshold	
	Spline	Logistic	Spline	Logistic
AR3	83%	64%	90%	61%
AR4	82%	73%	89%	77%
AR5	88%	86%	94%	88%
AR6	95%	95%	99%	96%

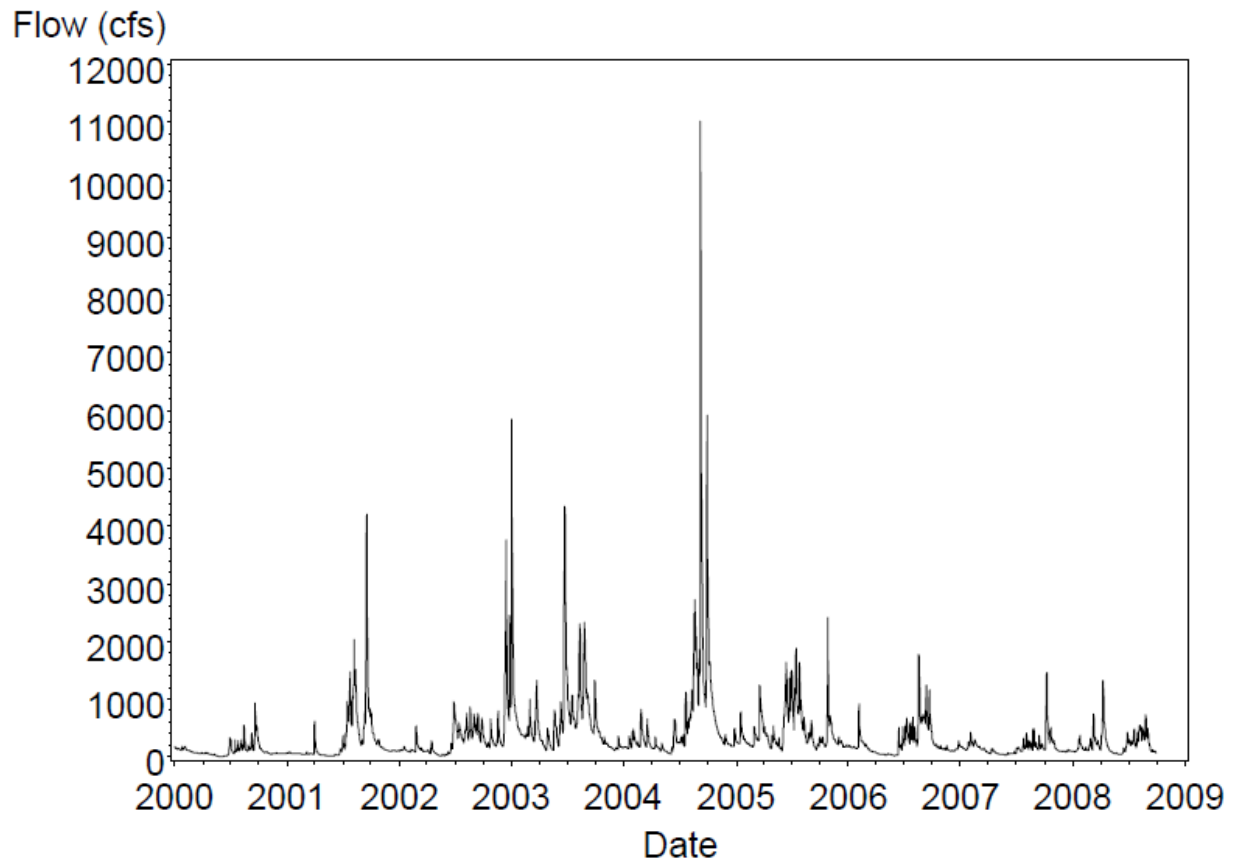


Figure 1. Time series of freshwater inflows at Bellshoals Road in the Alafia River from January 2000 through September 2008.

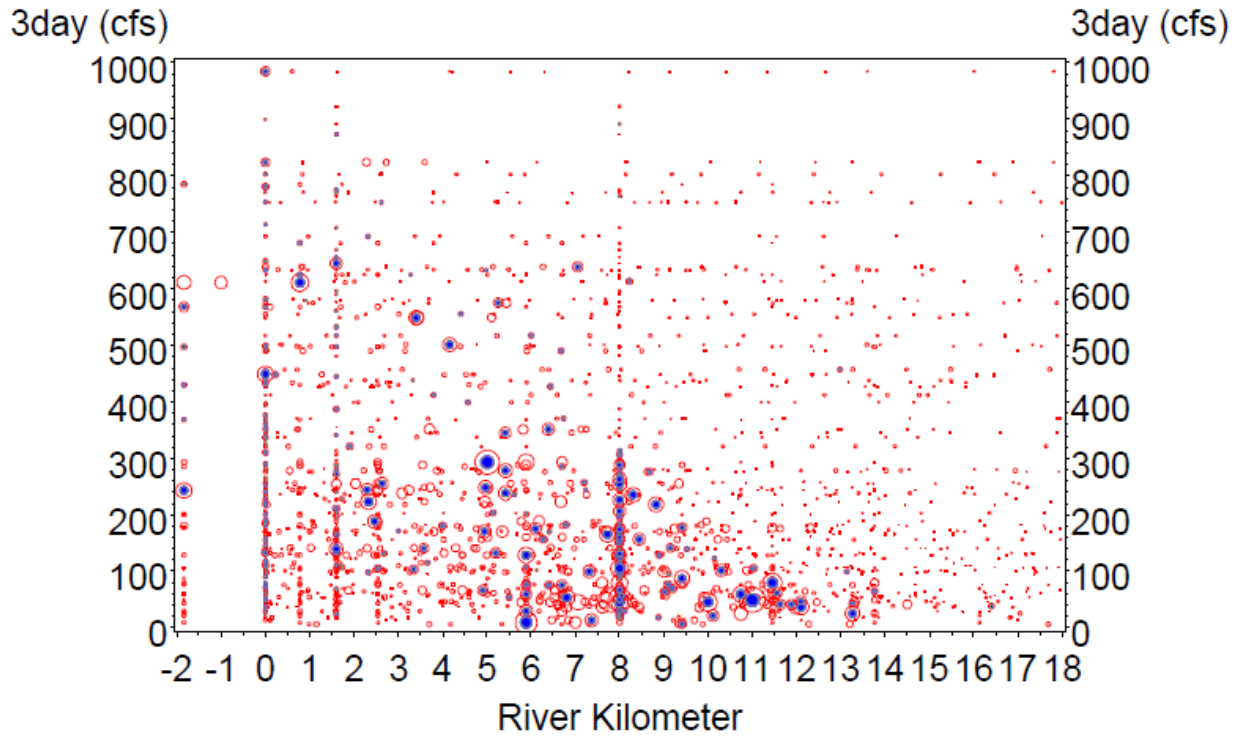


Figure 2. Distribution of chlorophyll a values in the Alafia River between 2000 and 2008. Bubble-size is related to chlorophyll biomass and filled circle indicates location of peak chlorophyll for the sample's date.

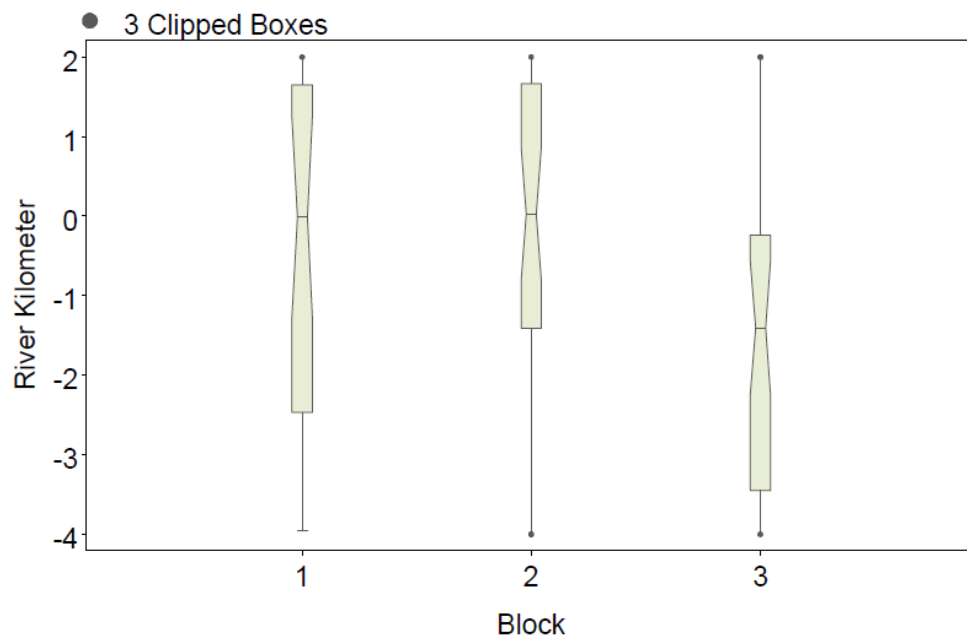


Figure 3. Difference in location of the maximum chlorophyll comparing spline model and log linear model provided in SWFWMD (2008) by seasonal blocks. Vertical axis represents the difference between model predictions (i.e. spline model – log linear model). Seasonal blocks: Block 1: April-June; Block 2: November-March; Block 3: July-October.

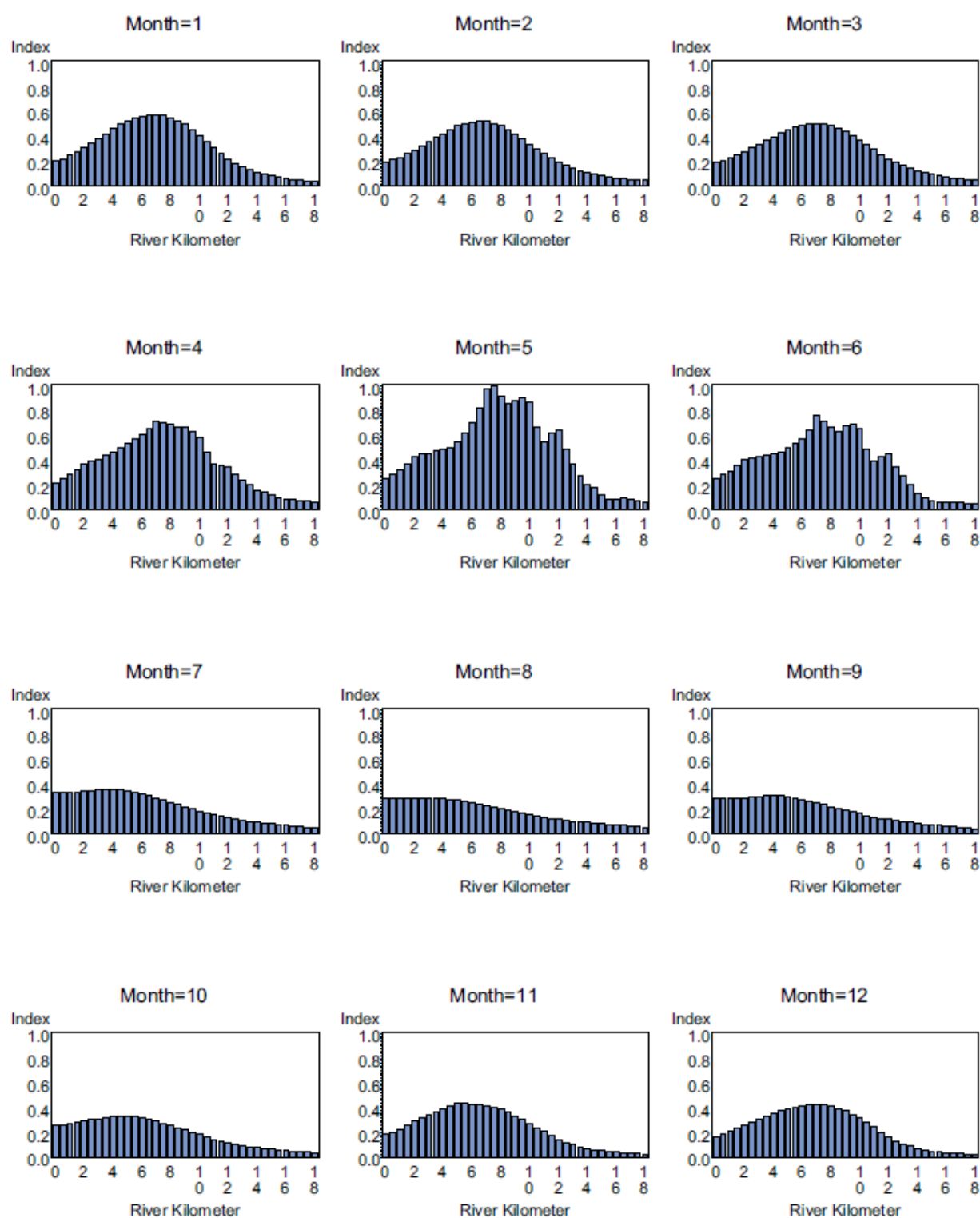


Figure 4. Monthly chlorophyll biomass index values (0 to 1) for the lower Alafia River from January to December from top left to bottom right.

ABSTRACTS

Session 8: Integrated Assessments – Tidal Tributary Studies

PRESENTATIONS

A More Comprehensive Approach for Determining Juvenile Snook Nursery Habitat in a Tampa Bay Wetland. A. Brame, C. McIvor, (United States Geological Survey), E.B. Peebles, and D. Hollander, (University of South Florida). It is widely accepted that wetlands serve as nursery habitat for numerous estuarine fish species, however few studies look past simple measures of abundance to define these habitats. In the fall of 2006 we set out to better define nursery habitat of common snook, *Centropomus undecimalis*, by sampling a tidal creek and connected ponds in both upstream and downstream portions of a single Tampa Bay wetland. We collected snook and measured habitat variables in each habitat, taking standard fish measurements (i.e., abundance, condition, size) to determine relationships between habitat, density, and growth. We also retained whole snook for stable carbon and nitrogen isotopic analysis. We found higher densities of juvenile snook in ponds (5.2 ± 1.06 snook/100m²) and upstream locations (5.5 ± 1.10) compared to creek sites (2.2 ± 0.47) and downstream portions of the wetland (2.0 ± 0.34). Snook collected from ponds were on average smaller (55.3 ± 1.17 mm SL) than those collected from the creek (66.7 mm SL ± 1.71), but condition (K) and growth rate did not differ between the two site types. Isotopic compositions of snook were distinctly different between ponds and the creek suggesting little movement between habitat types. Snook collected in ponds were isotopically enriched in nitrogen and depleted in carbon ($\delta^{13}\text{C} = -25.36 \pm 0.10$, $\delta^{15}\text{N} = 12.06 \pm 0.11$) relative to those collected in creeks ($\delta^{13}\text{C} = -24.77 \pm 0.15$, $\delta^{15}\text{N} = 10.93 \pm 0.18$). This result suggests more nutrient recycling occurred in the ponds, likely due to reduced flushing in these habitats. Based on isotopic composition, juvenile snook feed on small fishes or invertebrates at the 2nd trophic level and obtain carbon from benthic microalgae and particulate organic matter. Stable isotope ratios in conjunction with more frequently used variables such as abundance and growth seem to produce a clearer description of snook nursery habitat through the interpretation of fish diet and movements.
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What Have We Learned From 10 Years of Hydrobiological Monitoring on the Alafia River? R. Woihe, K.L.M. Jenkins, A. Willis, D. Robison, R. Montgomery (PBS&J) and R.G. McConnell (Tampa Bay Water). When the Alafia River Hydrobiological Monitoring Program (HBMP) was first designed in 1999, multiple stakeholder, research, and regulatory groups came to the consensus that data and monitoring programs then in existence were not sufficient to define baseline conditions or assess potential impacts of freshwater withdrawals in the Alafia River. One purpose of the Alafia River HBMP was to address these data gaps. Ten years of data and analyses are now available to determine how well the Alafia River HBMP has achieved its objectives. HBMP components have included water quality indicators (sampled via fixed, stratified-random, and automated continuous-recording stations) and biological indicators (vegetation, benthic macroinvertebrate, fish, ichthyoplankton, and birds). Results from the monitoring program have been used both for regulatory monitoring of potential adverse impacts and resource management purposes (e.g. developing minimum flows and levels). This presentation summarizes the current state of knowledge for the Alafia River relative to resource management and regulatory assurance purposes.
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The Determination of Minimum Flows and Levels for Tributary Rivers to Tampa Bay. M.S. Flannery, M.G. Heyl, X. Chen, and M.H. Kelly (Southwest Florida Water Management District). The Southwest Florida Water Management District has adopted, or is in the process of adopting, minimum flows and levels (MFLs) for all the major rivers that flow to Tampa Bay. MFLs establish the limit at which further withdrawals would be significantly harmful to the natural resources of the area. MFLs are determined separately for the non-tidal and tidal reaches of individual rivers using hydrobiological analyses and metrics that are appropriate for freshwater and estuarine systems, respectively. MFLs have been adopted or proposed for the fresh and estuarine portions of the Hillsborough and Alafia rivers and the tidal reach of the Tampa Bypass Canal, and are currently under evaluation for the Little Manatee and Manatee rivers.

In establishing MFLs for the Lower Hillsborough River and the Tampa Bypass Canal, the District took into account the extensive, existing structural alterations to those water bodies. For rivers that have not been so

altered, the District has employed the percent-of-flow method to determine MFLs. This method determines daily percentages that the natural baseline flow of the river can be reduced. These percentages can be determined within flow ranges or seasons and can be combined with a low-flow threshold below which no withdrawals are allowed. Ecological analyses that have been used for the determination of MFLs for tidal estuarine rivers have emphasized relationships of freshwater inflow with: the area, shoreline length and volume of biologically important salinity zones; the abundance and distribution of key fish and invertebrate species; and chlorophyll-a and dissolved oxygen concentrations. [sid.flannery@swfwmd.state.fl.us]

REVISITING AND UPDATING OUR KNOWLEDGE ON THE SCIENCE OF TAMPA BAY'S TIDAL TRIBUTARIES: SESSION SYNTHESIS

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ABSTRACT

More than 100 natural tidal creeks and rivers, and numerous man-made tidal channels, exist along the shores of the Tampa Bay estuary, but until recently, little was known of their habitat value for faunal communities, their contribution of primary and secondary production, or their role within the larger ecosystem. The need for scientific study of these systems was identified in 2003 when small tidal tributaries were ranked as the highest priority for research and management by the Tampa Bay Estuary Program and the Tampa Bay Regional Planning Council. At that time, ongoing studies of the larger tidal rivers continued to support science-based management decisions related to consumptive water uses in the Tampa Bay watershed.

As a result of the new attention, there was a reinvigoration of research on small tidal tributaries which has generated a great deal of new information that can be applied to better manage and conserve these potentially vital systems.

The 5th Bay Area Scientific Information Symposium (BASIS 5) provided a forum in which to present and discuss the new research and to provide an update on resource management activities related to the larger tidal rivers. The purpose of the Tidal Tributaries session, in following with the theme of BASIS 5, was to revisit and update our knowledge of the science of Tampa Bay's tidal tributaries. A panel discussion followed to summarize the major findings from studies presented during the session, identify information gaps in our understanding of tidal tributaries, and propose the logical next steps. The purpose of this paper is to present a session synthesis that will contribute to the development of a management plan for Tampa Bay's tidal tributaries.

Tampa Bay's Tidal Tributaries

Tampa Bay is fed by more than 100 tidal tributaries whose freshwaters maintain the salinity gradient of the estuary. Small tidal tributaries, often referred to as tidal creeks, range in size from less than one-tenth of a kilometer to ten kilometers along the main channel, while the five larger tidal rivers, the Hillsborough, Palm River/Tampa Bypass Canal, Alafia, Little Manatee and Manatee Rivers are on the order of 20-100 kilometers. Tidal tributaries in other estuaries are known to be productive systems whose ecological role and contribution of primary and secondary production to estuaries is poorly known. Based on recent studies, including those presented during the session on tidal tributaries at the recent BASIS 5 meeting, and the rapid rate of coastal development within the Tampa Bay watershed, there is an urgent need to better understand these small systems and their contribution to the ecological function of Tampa Bay so that, ultimately, a management plan can be drafted and adopted.

History of Research and Management of Tidal Tributaries in Tampa Bay

The 5th Tampa Bay Area Scientific Information Symposium (BASIS 5) was held from October 20-23, 2009 during which a special session on Tampa Bay's tidal tributaries was convened. The need for this session was first acknowledged in 2003 around the time of the BASIS 4 meeting when small tidal tributaries were ranked as the highest priority for research and management by the Tampa Bay Estuary Program (TBEP) and the Tampa Bay Regional Planning Council (TBRPC). At that time, demand for potable water was growing within the Tampa Bay watershed and Tampa Bay Water's Hydrobiological Monitoring Program had recently begun to monitor the effects of water withdrawals on the hydrology and ecology of the large tidal tributaries (i.e., tidal rivers).

However, save for a study by the TBRPC in the late 1980s (TBRPC 1986, 1987), very little was known about the role of Tampa Bay's smaller tidal tributaries (i.e., tidal creeks, man-made ditches, canals). Shortly after the conclusion of the TBRPC study, a framework was proposed for the management and conservation of Tampa Bay's tidal tributaries (Clark 1987), though efforts to continue studying these systems and the development of a management plan never came to fruition.

Although management issues for Tampa Bay's tidal tributaries were discussed at the BASIS 2 meeting in 1991 (Clark 1991), it was not until BASIS 4, more than a decade after tidal tributaries had last taken center stage, that the issue once again gained attention. During the six years between BASIS 4 and BASIS 5, there was a reinvigoration of research on small tidal tributaries which has generated a great deal of new knowledge that is being applied to better manage and conserve these potentially vital components of the Tampa Bay ecosystem. In 2003, the U.S. Geological Survey (USGS) initiated a study as part of the Tampa Bay Integrated Science Study to assess the role of tidal creeks and man-made tributaries as fish habitat (Krebs et al. 2007; Yeager et al. 2007). The Tidal Tributaries Working Group, a multi-agency collaboration led by the TBEP, was formed shortly thereafter to move forward with the study of small tidal tributaries in Tampa Bay. The Tidal Tributaries Habitat Initiative (TTHI) resulted from the Working Group and further contributed to our knowledge of tidal tributaries by conducting a study of nine tidal creeks, the results of which are detailed below (Greenwood et al. 2008a; Sherwood et al. 2008; Sherwood, these proceedings; MacDonald et al., these proceedings; Malkin et al., these proceedings). Between 2005 and 2009, several additional studies were initiated by the Florida Fish and Wildlife Conservation Commission (Greenwood et al. 2007, 2008b; MacDonald et al., these proceedings; Ley et al., these proceedings) and joint efforts from the University of South Florida and the USGS (Brame et al., these proceedings; Krebs et al., these proceedings) that have further advanced our understanding of the role of tidal tributaries as fish habitat, sources of nutrient inputs to the estuary, and areas of primary and secondary production. Results of these studies have collectively begun to characterize the ecology of nearly half of the Bay's 100+ tidal tributaries. Most recently, subsequent re-evaluation of research priorities by the TBEP Technical Advisory Committee in early 2009 confirmed that tidal tributaries remain a priority among Tampa Bay scientists and resource managers.

Purpose of the Tidal Tributaries Session

The purpose of the Tidal Tributaries session, in following with the theme of BASIS 5, was to revisit and update our knowledge of Tampa Bay's tidal tributaries. Ten invited speakers presented new science conducted with the purpose of better understanding the ecology of tidal tributaries, both small tidal creeks and large tidal rivers. Most of the talks presented during the session represented the culmination of six years of research on tidal tributaries in the Tampa Bay area. A panel discussion followed the session with the objectives being to: 1) summarize the major findings from studies presented during the session, 2) identify information gaps in our understanding of tidal tributaries that could be filled by considering studies from other estuaries or by undertaking new studies, and 3) propose the logical next steps towards the development of a management plan for Tampa Bay's tidal tributaries. The purpose of this paper is to present a session synthesis that will update our knowledge of tidal tributaries in Tampa Bay.

Synthesis of Recent Science on Tidal Tributaries

The ecological value of various estuarine habitats (e.g., seagrass beds, hardbottom, emergent wetlands) to the function and maintenance of faunal populations is widely recognized among scientists and resource managers. It was assumed that tidal tributaries in the Tampa Bay area had substantial ecological importance based on studies of tidal creeks in other estuaries (Mallin and Lewitus 2004). In an early effort to establish the ecological importance of Tampa Bay's

tributaries, the TTHI took an ecosystem-level approach to study the water and sediment quality, macrobenthos and nekton communities and the nitrogen pathways that support and connect the various trophic levels that characterize tidal tributaries (see Sherwood, these proceedings; MacDonald et al., these proceedings). The purpose of this collaborative, multi-agency study was to improve protection and management of fish populations in the Tampa Bay system by:

- 1) Determining the relative importance of tidal tributaries as fish habitat in Tampa Bay;
- 2) Determining the effects of habitat parameters (watershed condition, water quality, structural habitat) on fish habitat use in impacted and unimpacted tidal tributaries;
- 3) Developing measurable goals, management recommendations, and a pilot Tidal Tributaries Management Strategy based on study results; and
- 4) Communicating results to managers and the public to support informed decision-making regarding preservation or restoration of tidal tributary habitats.

Many of the water and sediment quality variables measured during the TTHI were positively correlated with land development intensity surrounding tidal tributaries. Elevated nutrients, chlorophyll- *a* and turbidity concentrations were observed in tributaries with more intense land development, as were more fine sediments and higher concentrations of heavy metals, organic contaminants, and pesticides in the sediments. However, the effects of the observed physicochemical changes were not as obvious in the biotic community. Abundance and richness of benthos and nekton were not well correlated with land development intensity, possibly as a result of the relatively low development intensity surrounding the studied tributaries. Stable isotope techniques used during the TTHI and follow-up studies of additional tidal creeks dominated by agricultural and urban land uses were able to more clearly establish a connection between tidal creeks and land use in the surrounding watershed (Malkin et al., these proceedings). From these studies it was determined that the nitrogen-isotope ratio from fish tissue is a useful tool for discriminating sources of nutrient inputs within a specific tributary and reflects the dominant land-use classes contributing to the nutrient load of a given tidal tributary. For example, undeveloped wetlands, residential lands and agricultural land uses were found to have distinct influences on the isotopic signatures in downstream fish tissues that reflected the uptake of nutrients in wetlands, septic/sewage inputs from urban lands, and runoff of fertilizer from agricultural crops, respectively. An empirical model was developed that identifies the relative contribution of nutrients from various land uses within the watershed surrounding a tidal tributary. This model has practical applications for land management and the control of nutrient inputs to tidal tributaries. Results from these isotope studies also identified the source of primary production supporting upper trophic levels as benthic microalgae and phytoplankton (rather than larger macrophytes like mangroves and emergent vegetation), with seasonal hydrological cycles factoring significantly in the location, type, and extent of the supporting primary producers.

Much of what has been learned recently about tidal tributaries has focused on the value of these systems as fish habitat. As a result, we now know a great deal about the nekton community from over half of Tampa Bay's tidal creeks and man-made tidal tributaries (i.e., mosquito-control and stormwater-drainage ditches). In general, the nekton community seems to vary most among tidal tributaries, types of tributaries (i.e., creeks, ditches) and seasonally, but less between the tributary and adjacent habitats (MacDonald et al., these proceedings; Krebs et al. 2007; Yeager et al. 2007; Greenwood et al. 2008a). This finding underscores the geographic uniqueness of tidal creeks and the potential differences among them that may result from the various mosaics of land uses surrounding each tributary. Early work suggested that the community structure of nekton in nine tidal tributaries was poorly correlated with land development intensity in the surrounding watersheds (MacDonald et al., these proceedings; Greenwood et al. 2008a). The lack of correlation may be explained in part by the relatively low development intensity characteristic of the studied tributaries. In a follow-up study aimed specifically at the assessment of land use on

the quality of fish habitat (Krebs et al., these proceedings), high community similarity was again observed among non-urban tributaries with lower intensity land development in the surrounding watersheds. However, urban tributaries, characterized by significantly greater land development intensity, had nekton communities that were dominated by high densities of the poeciliid fishes sailfin molly (*Poecilia latipinna*) and mosquitofish (*Gambusia holbrooki*) and very low densities of palaemonid grass shrimp (*Palaemonetes* spp.); an otherwise dominant member of the nekton community in tidal tributaries. Many of the common nekton taxa were also found to be in poorer condition in urban tributaries compared to non-urban tributaries, based on length-weight relationships. In contrast to community structure and body condition, the abundance (total and species-specific) and species richness of nekton was found to vary widely among tidal tributaries and appeared to be independent of land use in the surrounding watershed, even for the most highly urbanized tributaries.

Though the importance of tidal tributaries as habitat for the entire nekton community cannot be overstated, one fish species in particular has received significant attention as a result of recent research on tidal tributaries - the common snook (*Centropomus undecimalis*). Use of nursery habitats by juvenile snook in Tampa Bay has not been well understood until recently. Historically, juvenile snook have been nearly absent in the open estuary but in slightly greater abundance in Tampa Bay's tidal rivers, based on long-term data from the State of Florida's Fisheries-Independent Monitoring (FIM) program. The highest abundances have recently been observed in small tidal creeks, where juveniles tend to be considerably more abundant (2-36x) compared to the shoreline habitat just outside the mouth of the creek (Greenwood et al. 2008a; Sherwood, these proceedings). Within one particular tidal creek, Frog Creek, juvenile snook appeared to be more common in the upstream portions of the creek and even more so in the permanently connected, shallow tidal ponds in the upper reaches (Brame et al., these proceedings). The observed differences in isotopic composition of C and N of juvenile snook between the creek and ponds suggest limited movement by juveniles between habitats and the use of different trophic resources, and further support the idea of potential differences in the value of microhabitats for juvenile snook within small tidal tributaries.

To better understand the role of individual tidal creeks as nursery habitat for juvenile snook and red drum (*Sciaenops ocellatus*), otolith microchemistry is being used to identify those tidal tributaries that may contribute a disproportionate number of snook and red drum to the adult populations in Tampa Bay and the Gulf of Mexico (Ley et al., these proceedings). Juveniles of these species have been collected for several years from tidal creeks and rivers around Tampa Bay for extraction of otoliths. The elemental fingerprints from these juveniles will be used to characterize each of the tidal tributaries and will be compared to that of adults collected from adjacent estuarine and Gulf habitats to determine which tidal tributaries support the largest proportion of adults that enter the spawning population. In contrast to juvenile snook, which seem to be most abundant in the tidal creeks, juvenile red drum are thought to be most abundant in the larger tidal rivers, with relatively low abundances observed historically in the small tidal tributaries (Krebs et al. 2007; Greenwood et al. 2008a, b). By identifying those tributaries that are "essential fish habitat" for snook and red drum, this study will provide more accurate information on the role of tidal tributaries as nursery habitat so that steps can be taken to better manage both the habitat and the fish populations. Ultimately, habitat comparisons could be made among tidal tributaries deemed "essential fish habitat" and between these tributaries and those that appear to be less suitable, in order to determine the minimum habitat requirements necessary to support early life history stages of snook and red drum, and thus direct management goals for these fishery species.

What can we learn from the science and monitoring efforts that have been conducted in the larger tidal rivers? Several examples of science-based management strategies and a predictive ecological model for the tidal rivers were presented during the session. Tampa Bay Water's Hydrobiological Monitoring Program (HBMP) has been ongoing for 10 years to monitor changes to the ecology of Tampa Bay's tidal rivers and to provide guidance related to the demand for potable water at a time of limited supply and increasing demand (Woithe et al., these proceedings). An ecosystem-level approach has been accomplished by the HBMP as a result of a semi-synoptic sampling design including elements of water quality and hydrology, submerged aquatic and emergent shoreline vegetation, benthic, planktonic, and nektonic fauna, and wading birds. The HBMP has provided a good understanding of the relationships between tributary flow and the distribution of salinity zones between the mouth and freshwater extent of several of Tampa Bay's tidal rivers. A prominent management strategy for Tampa Bay's water resources, Minimum Flows and Levels (MFLs), has benefited from science conducted during the HBMP. The Southwest Florida Water Management District has worked to establish MFLs for the tidal rivers in Tampa Bay and has completed flow reduction criteria for the Hillsborough, Palm and Alafia Rivers with MFLs currently under development for the remaining tidal rivers in Tampa Bay (Flannery et al., these proceedings). What we've learned from the tidal rivers may eventually be used for the development of MFLs for the smaller tidal tributaries as the demand for potable water increases. Ecological models have been developed to predict changes in the center of distribution of fish and crustaceans in response to primary production across a range of tributary flows (Wessel et al., these proceedings). The presentations on tidal rivers were based on the use of scientific data to address the implications of water resource use on the dynamics of the natural systems and provide a good example of how similar science-based strategies might be applied to manage the smaller tidal tributaries. The question remains, though: is it possible to downscale what we've learned from the tidal rivers to better manage and predict the outcome of potential management scenarios for the small tidal tributaries?

What can we learn from studies of small tidal tributaries in other estuaries? A three-year field survey of 16 tidal tributaries in Sarasota Bay has identified biotic variables characterizing bivalves, pelagic, epibenthic, and infaunal abundances, as well as percent cover of periphyton and filamentous algae as indicators of pollution and has resulted in a Tidal Creek Condition Index for these creeks (Estevez et al., these proceedings). The index correlated well with species richness and total faunal densities, but was not as well correlated with land development intensity, watershed age, or total metals. Beyond the state of Florida, ongoing studies of the effects of coastal land use on the ecological condition of South Carolina's tidal creeks (Holland et al. 2004) can serve as a useful reference for the study and management of Tampa Bay's tidal tributaries. Relatively simple measures of coastal development for South Carolina, population density and impervious surface, have been shown to be good indicators of the health of tidal creeks at many levels from water quality to faunal communities. These measures are readily available for tidal creek drainages in the Tampa Bay watershed and are probably suitable indicators of tidal creek condition there as well. Given the research in Tampa Bay tributaries on nutrient inputs, trophic processes, water and sediment quality, and fish communities, it would be possible to incorporate these aspects into the development of an index of ecological condition for Tampa Bay tributaries. If a habitat condition index were to be developed, however, careful attention should be given to the selection of metrics and the limitations associated with ecological indices, in general. We need to realize that the effectiveness of a condition index for Tampa Bay's tidal tributaries is constrained by the metrics that define the index and that critical aspects of ecological condition may be overlooked, resulting in a less than useful product.

Towards a Comprehensive Management Plan for Tampa Bay's Tidal Tributaries

The information that we have gained over the last six years of research on Tampa Bay's tidal tributaries has contributed to our knowledge of the ecology and function of these systems and will ultimately be the foundation of a draft management plan for the smaller tidal tributaries. At BASIS 2, several management strategies were recommended for tidal tributaries (Clark 1991). These strategies emphasized: 1) the maintenance and restoration of natural function, 2) the development of consistent and compatible land use standards, 3) and the management of tidal creeks as an important public asset. More specifically, they included:

- 1) Maintenance and restoration of natural function - Improvement of water quality, minimization of point- and non-point source pollution, protection of natural freshwater inputs, and the establishment of a monitoring and enforcement program for water quality, habitat and species. Preservation of natural habitat, maintenance of geomorphology of tributary channels, natural vegetation and wildlife communities, and protection of archeological sites.
- 2) Development of consistent and compatible land use standards - Public land acquisition, conservation easements, compatible low-density development and clustering of water-related activities on lands surrounding tidal tributaries.
- 3) Management of tidal creeks as important public assets - Active public outreach to convey the ecological and societal value of tidal tributaries, as well as encouraging compatible public access to minimize conflicts among user groups.

More recently, the Tampa Bay Tidal Tributaries Working Group listed four recommendations based on work from the 2006 Tidal Tributaries Habitat Initiative. Many of the same elements from the original framework (i.e., ecological function, hydrology, habitat, faunal communities, monitoring, and public outreach) were reflected in the new recommendations. These recommendations included:

- 1) Maintain connectivity between open bay waters, tidal rivers and smaller tidal tributaries to allow fish movement, water flow and nutrient flux between these backwaters and larger estuarine areas.
- 2) Reduce "flashiness" of water flow to tidal tributaries to promote natural flow patterns and foster the productivity of fish food sources within these systems.
- 3) Monitor the condition of fish nursery functions and physical water quality parameters in additional backwaters to further assess the uniqueness of Tampa Bay's tidal tributaries.
- 4) Improve public education and stewardship of tidal tributaries by promoting the importance of these systems as key habitats to estuarine fish species (e.g. snook).

In addition to the recommendations made during BASIS 2, and more recently by the Tidal Tributaries Working Group, several additional elements were suggested during the BASIS 5 Panel Discussion or have been raised repeatedly in past discussions and warrant consideration in order to address gaps in our understanding of tidal tributaries.

- Tidal tributaries need to be characterized by condition (e.g., stressed, natural, restorable), linear extent, drainage size, hydrology, bay segment, dominant land uses and impervious surface within drainage basins, presence of sediment contaminants, and channel modifications/maintenance activities (e.g., dredging). Ecological value as critical habitat for various species could also be used to prioritize tidal tributaries for conservation and/or restoration efforts. Given current GIS capabilities, this task should not be difficult to accomplish and has been done to some extent as part of the tributaries research by FWRI, TBEP and USGS.

- One of the most basic pieces of information required to understand the physical and ecological processes within tidal tributaries is their hydrology. Variables such as flow, discharge, and tidal prism need to be measured and could be used to characterize the tidal tributaries. The USGS has permanent stream gauge stations on freshwater and estuarine tributaries throughout the United States. It would make sense for gauge stations to be established and maintained following an identical protocol to allow consistency among data collected. This recommendation could be addressed as part of a larger monitoring program for tidal tributaries (outlined below).
- Removal of salinity barriers is one way to re-establish tidal connectivity to some of the tributaries in which natural flow patterns have been restricted. Impediments to flow such as road crossings, flood control structures and weir structures established to prevent marine waters from intruding on agricultural lands all act as salinity barriers and restrict the natural flow of tidal tributaries. Salt Creek in southern Pinellas County, for example, had its tidal connection blocked by the construction of a tide gate in 1941, though freshwater is still able to drain to Tampa Bay when levels are high. The weir constructed on Alligator Creek in 1947 resulted in the conversion of a tidal bayou to a freshwater system, Alligator Lake. Much of the surrounding lands are now maintained by Pinellas County as a Management Area making this site an ideal location for tidal tributary restoration. The initial impetus for construction of these barriers should be considered, though, prior to their removal to avoid any unintended consequences such as flooding of residential or commercial lands adjacent to tributaries or increasing the salinity of residential irrigation wells.
- Implementation of a monitoring program for hydrology, water and sediment quality, and nekton communities in the tidal creeks and man-made tributaries. The infrastructure exists to establish such a monitoring program. A cooperative effort led by local, state, and federal agencies with the capabilities to monitor these elements already exists. Pinellas, Hillsborough, and Manatee County environmental departments are currently monitoring water and sediment quality in Tampa Bay, its tidal rivers and some of the smaller tributaries. The USGS maintains an array of stream gauges in freshwater tributaries and some estuarine systems across the nation, including the Tampa Bay watershed, and would be a suitable choice for establishing a stream-gauge network in the smaller tidal tributaries. The FWC-FIM program has monitored fish and decapod crustaceans in estuaries and large tidal rivers state-wide since 1989 and has been involved with the short-term studies in tidal creeks reported here and could very effectively take the lead on a nekton monitoring program in Tampa Bay's small tidal tributaries and backwaters.
- Continued research and monitoring efforts for the small tidal tributaries need to take an ecosystem-level approach. This could be accomplished by considering metrics of productivity, nutrient inputs, faunal abundance and community structure, pollution indicators, and land use which would be inclusive of many levels of the ecosystem. In addition, we may now know enough about the basic ecological elements of tidal tributaries to start asking directed questions and addressing those questions using an experimental approach during future studies. One important application of experimental studies would be for the assessment of habitat restoration projects.
- Development of a condition index for Tampa Bay's tidal tributaries might be considered as a tool for monitoring the ecological status of these systems, though the limitations related to the subjective selection of index metrics must be kept in mind. The Tidal Creek Condition Index for Sarasota Bay (Estevez et al., these proceedings), Tampa Bay Benthic Index (Malloy et al. 2007) based on diversity and abundance of benthic fauna, and the Estuarine Index of Biotic Integrity (Deegan et al. 1997) could be used as models. At the very least, potential indicators of habitat condition for tidal tributaries (e.g., nutrient loadings, indicator species) should be identified to assist with the development of management goals and targets for these systems.

- Beyond the structural aspects of the tidal tributary ecosystem (e.g., salinity range, levels of freshwater inflow, composition of the shoreline vegetation community, faunal species richness and abundance), the ecological function of tidal tributaries needs to be understood. Understanding processes such as nutrient cycling, rates of microbial activity, and export of secondary production to the greater estuary, for example, is central to the effective management of tidal tributaries.
- The potential impacts of exotic species in terms of their presence in Tampa Bay's tidal tributaries (Paperno et al. 2008), displacement of native species and competition for resources (Greenwood et al. 2008b), also seem important though was not an issue discussed in great detail during the BASIS 5 session.
- Understanding recruitment dynamics and migratory pathways for fish and shellfish that utilize tidal tributaries as nurseries as a function of existing hydrodynamics and circulation patterns and potential future alterations due to anthropogenic influences, such as urbanization and port development.
- The incorporation of sustainability science into research and management practices should be a central objective of future decisions regarding tidal tributaries as well as the entire Tampa Bay ecosystem (Weinstein and Krebs, these proceedings).

Over the course of the last six years, we have made substantial progress towards understanding the ecology of Tampa Bay's tidal tributaries. We hope that the information gleaned from these studies, and from this synthesis, will be useful in moving forward with a draft management plan for Tampa Bay's small tidal tributaries.

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TAMPA BAY NITROGEN MANAGEMENT CONSORTIUM: A COLLABORATIVE APPROACH TO MEET WATER QUALITY TARGETS AND SUPPORT SEAGRASS RECOVERY IN TAMPA BAY

H. Greening

In 1996, the Tampa Bay Estuary Program, local government and agency partners adopted numeric management targets to restore and protect seagrass beds in the estuarine bay segments of Tampa Bay. These resource-based targets were developed using empirical and mechanistic models and a long-term water quality database initiated in 1974 (Janicki and Wade 2006; Wang et al. 1999). Numeric management goals for seagrass acreage, and numeric targets for light attenuation, chlorophyll *a* concentrations, and the TN loads necessary to meet and maintain water quality targets that protect the designated use were adopted. A multi-pronged management strategy was implemented to meet these targets (Greening and Janicki 2006).

In 1998, the FDEP proposed and USEPA approved a regulatory limit (Total Maximum Daily Load) for nitrogen for Tampa Bay required by Section 303(d) of the federal Clean Water Act. The TMDL nitrogen loads were based on the resource-based management targets (water clarity, chlorophyll *a* concentrations and the nitrogen loads observed to meet these targets) developed by the Tampa Bay Estuary Program partners to support seagrass recovery in Tampa Bay. Since approval of the TMDL, chlorophyll *a* thresholds have been met in all four major bay segments, with the exception of one year in Lower Tampa Bay and two years in Old Tampa Bay. Seagrass acreage has increased by more than 4,800 acres bay-wide over this same period (Figures 1 and 2).

In December 2007, 40+ public and private participants in the Tampa Bay Nitrogen Management Consortium proactively committed to develop an equitable process to propose nitrogen load allocations to all sources consistent with the TMDL. The Consortium participants (Table 1) developed a set of nitrogen wasteload and load allocations that equitably distribute the burden of nitrogen management across all sectors and sources of nitrogen loading within the basin, as well as the total maximum loading of nitrogen to each major bay segment (TBNMC 2010). In September 2009, the Consortium participants finalized their technical process, and proposed TN allocations to all 189 point and nonpoint sources within the Tampa Bay watershed.

The technical approach was accompanied by a landmark agreement between government and private industry representatives to limit nitrogen pollution in Tampa Bay. The agreement spells out how much nitrogen can enter Tampa Bay through stormwater, air pollution, treated wastewater, and industrial discharges through 2012. The limits will maintain nitrogen loadings to the bay at existing levels; additional nitrogen associated with growth must be offset through additional pollution controls.

The Tampa Bay Nitrogen Management Consortium's collaborative approach to meeting water quality targets is unique in the country in that public and private nitrogen dischargers worked together to define the technical process and nitrogen load limits for each of the sources within the watershed. FDEP and USEPA worked with the Consortium and provided concurrence at each major step, but did not lead the Consortium's process.

The Consortium's approved technical document, including nitrogen load allocations to each source within the watershed, can be found at www.tbep.tech.org

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Year	Old Tampa Bay	Hillsborough Bay	Middle Tampa Bay	Lower Tampa Bay
1974	No	No	No	Yes
1975	No	No	No	Yes
1976	No	No	No	Yes
1977	No	No	No	No
1978	No	No	No	Yes
1979	No	No	No	No
1980	No	No	No	No
1981	No	No	No	No
1982	No	No	No	No
1983	No	No	No	No
1984	Yes	Yes	No	Yes
1985	No	No	No	Yes
1986	No	No	Yes	Yes
1987	Yes	No	No	Yes
1988	Yes	Yes	Yes	Yes
1989	Yes	No	Yes	Yes
1990	Yes	No	Yes	Yes
1991	Yes	Yes	Yes	Yes
1992	Yes	Yes	Yes	Yes
1993	Yes	Yes	Yes	Yes
1994	No	No	No	No
1995	No	No	No	Yes
1996	Yes	Yes	Yes	Yes
1997	Yes	Yes	Yes	Yes
1998	No	No	No	No
1999	Yes	Yes	Yes	Yes
2000	Yes	Yes	Yes	Yes
2001	Yes	Yes	Yes	Yes
2002	Yes	Yes	Yes	Yes
2003	Yes	No	Yes	Yes
2004	Yes	No	Yes	Yes
2005	Yes	Yes	Yes	No
2006	Yes	Yes	Yes	Yes
2007	Yes	Yes	Yes	Yes
2008	Yes	Yes	Yes	Yes

Figure 1. Compliance with FDEP-approved annual mean chlorophyll *a* concentration thresholds for the four major bay segments, 1974-2008. Green= annual chl *a* concentration at or below threshold values; red= annual chl *a* concentration exceeds threshold values. Threshold values: Hillsborough Bay 15.0 ug/l; Old Tampa Bay 9.3 ug/l; Middle Tampa Bay 8.5 ug/l; Lower Tampa Bay 5.1 ug/l. Data source: EPCHC.

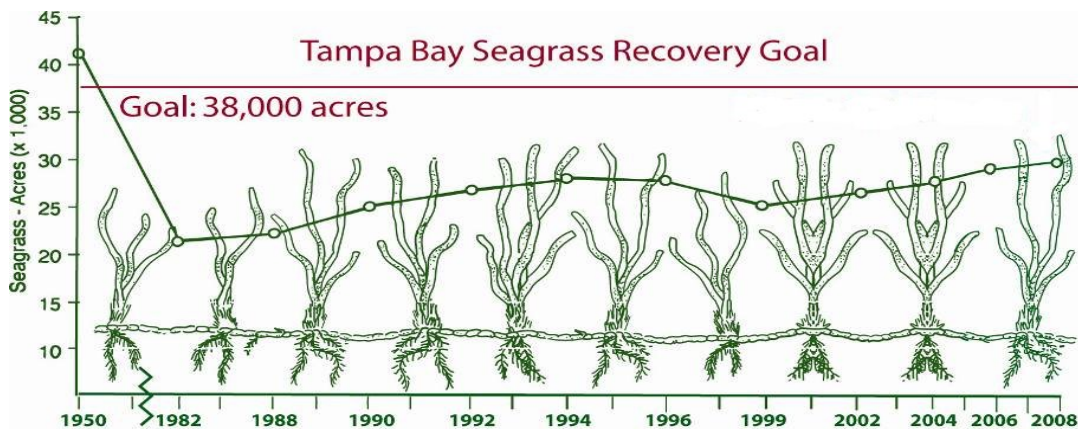


Figure 2. Seagrass acreage since 1950 in Tampa Bay. Data source: SWFWMD.

Table 1. Tampa Bay Nitrogen Management Consortium Participants

Alafia Preserve (Mulberry), LLC	CF Industries
City of Bradenton	City of Clearwater
City of Gulfport	City of Lakeland
City of Largo	City of Mulberry
City of Oldsmar	City of Palmetto
City of Plant City	City of Safety Harbor
City of St. Petersburg	City of Tampa
CSX Transportation	Eagle Ridge (Mulberry), LLC
Eastern Associated Terminals	
Environmental Protection Commission of Hillsborough County	
Florida Department of Agriculture and Consumer Services	
Florida Department of Environmental Protection	Florida Department of Transportation
Hillsborough County	Kerry I & F Contracting
Kinder Morgan Bulk Terminals, Inc.	LDC Donaldson Knoll Investments, LLC
MacDill Air Force Base	Manatee County
Mosaic Company	Pasco County
Pinellas County	Polk County
Southwest Florida Water Management District	Tampa Bay Regional Planning Council
Tampa Bay Water	Tampa Electric Company
Tampa Port Authority	Trademark Nitrogen
Tropicana Products	U.S. Environmental Protection Agency
Yara North America	

Janicki Environmental, Inc. (technical support)
Tampa Bay Estuary Program (coordinator and facilitator)

USING A ‘DECISION MATRIX’ APPROACH TO DEVELOP A FECAL COLIFORM BMAP FOR IMPAIRED WATERS IN THE HILLSBOROUGH RIVER WATERSHED

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ABSTRACT

This report provides excerpts from a series of documents that were prepared by the authors as part of an FDEP-funded project to develop a basin management action plan (BMAP) to address fecal coliform impairments and total maximum daily loads (TMDLs) for six impaired stream segments in the Hillsborough River watershed. The project developed and applied a decision-support tool, which is conceptually similar to the ‘decision matrix’ that is currently used by the Tampa Bay Estuary Program to assess water quality in Tampa Bay, to help guide the BMAP process. The decision-support tool is based on technical approaches and resource management strategies recommended by the World Health Organization (WHO) and the U.S. National Research Council (NRC). In addition to fecal coliform levels and information on the frequency of exceedance of State water quality standards, the approach also includes an assessment of the types of fecal contaminant sources that may affect each stream segment and an estimate of their potential human health risks. A weight-of-evidence approach based on the ‘Annapolis protocol’, which was developed by the WHO to address the limitations of fecal coliforms and other bacterial indicators for assessing and managing health risks associated with recreational water use, is used to organize and summarize the indicator and source category information. In addition to this application to Class III (recreational) waters, it appears that similar conceptual approaches based on the Annapolis protocol could also be used for managing health risks of waterborne disease associated with Class I (potable supply) and Class II (shellfish harvesting) waters.

BACKGROUND

From 2006 through 2009 a broadly-based stakeholder group worked to develop a basin management action plan (BMAP) to address fecal coliform impairments in six sub-basins within the Hillsborough River watershed (HRBWG 2009). The BMAP focuses on reducing fecal coliform levels and protecting the health of recreational users of these stream segments, which have been designated as impaired by the Florida Department of Environmental Protection (FDEP) under the Florida Watershed Restoration Act (FWRA) (Chapter 403.067, Florida Statutes) and the Impaired Surface Waters Rule (IWR) (Rule 62-303, Florida Administrative Code), and for which total maximum daily loads (TMDLs) have been established.

The Hillsborough River BMAP was developed using an adaptive management approach applied at a watershed scale. Most of the public and private sector organizations participating in the BMAP process are also active members of the Tampa Bay Nitrogen Management Consortium, which was initiated in 1998 under the auspices of the Tampa Bay Estuary Program (TBEP). The consortium’s success in using a collective approach to maintain nitrogen loads to Tampa Bay at or below target levels was the model used by the Environmental Protection Commission of Hillsborough County (EPCHC) in proposing a Watershed Management Initiative (WMI) for waterbodies in Hillsborough County in 2005. When the Florida Department of Environmental Protection (FDEP) approved a number of TMDLs in 2006, the affected parties in the Hillsborough River Basin and FDEP accepted the WMI concept as an appropriate framework for BMAP development.

For administrative purposes the BMAP stakeholders are organized as two working groups. The Technical Stakeholders (TS) group is open to all interested parties, and is primarily comprised of

technical staff from local, regional, state, and federal government agencies and private sector organizations. It collates and evaluates water quality data and other pertinent information, and develops resource management options based on that information. These technical and resource management options are then evaluated by FDEP and presented to the Basin Working Group (BWG), whose members have been authorized by the organization they represent to select preferred management actions which will be implemented through the BMAP to achieve the relevant TMDLs. Organizations participating in the Hillsborough River BWG included each of the cities and counties in the watershed, the EPCHC, several state agencies (Florida Agriculture and Consumer Services, FDEP, Florida Department of Transportation), the Southwest Florida Water Management District, the TBEP, and staff from two universities (University of Florida and University of South Florida).

PROJECT AREA

The BMAP addresses the following six hydrologic units (identified by their “water body IDs” or WBID numbers), whose locations are shown in Figure 1:

- Blackwater Creek (WBID 1482), located in northern Hillsborough County, is 13.6 miles long, has a 113 square-mile drainage area, and discharges to the Hillsborough River;
- New River (WBID 1442), located in southeastern Pasco County and northern Hillsborough County, is 11.1 miles long, has a 20.9 square mile watershed, and discharges to the Hillsborough River;
- Spartman Branch (WBID 1561), located in north-central Hillsborough County and the City of Plant City, is 4.5 miles long, has a 27.4 square-mile watershed, and discharges to Pemberton Creek, which is a tributary of Baker Creek and Lake Thonotosassa;
- Baker Creek (WBID 1522C), located in north-central Hillsborough County, is 2 miles long, also has a 27.4 square-mile watershed, and discharges to Lake Thonotosassa, which is a tributary of the Hillsborough River via Flint Creek;
- Flint Creek (WBID 1522A) has a >60 square mile watershed which includes the Spartman Branch, Lake Thonotosassa and Campbell Branch sub-basins. The creek is 2.3 miles long, extending from Lake Thonotosassa to the Hillsborough River; and
- Lower Hillsborough River (WBID 1443E), located in the City of Tampa between Sulphur Springs and the river mouth, has a length of 7.8 miles and a 675 square-mile watershed which includes all the WBIDs listed above.

TECHNICAL ISSUES AFFECTING THE USE OF FECAL COLIFORMS AS INDICATORS OF WATER QUALITY AND HUMAN HEALTH RISK

While regular monitoring of fecal coliform bacteria or other microbial indicator organisms (IOs) is a key component of the water quality management process, the monitoring data must be augmented with other types of information to provide a comprehensive framework for managing microbial water quality and protecting human health. Health risks associated with exposure to pathogens in recreational waters have been investigated under a variety of environmental conditions in a number of geographic areas on a worldwide basis (summarized by NRC 2004; WHO 2003, 2005; EPA 2007). However, it is difficult to detect human pathogens, particularly viral and protozoan pathogens, in water samples collected from recreational waters. Methods for detecting and identifying many infectious viruses or parasites are often expensive and technically complex to perform, when they exist at all. Tests for bacterial pathogens are more readily available, but these organisms typically have specialized nutritional requirements and susceptibilities to environmental stresses that make this task difficult as well (WHO 2003; NRC 2004).

As a result, water quality monitoring programs have traditionally relied on the detection of fecal (or “thermotolerant”) coliform bacteria, or other easily-monitored IOs, to indicate the potential presence of fecal contamination and its associated human health risks (e.g., Alonso et al. 1999). However, many questions exist concerning the effectiveness of these IOs as health risk indicators, and a number of environmental and physical factors are known to affect their usefulness for this purpose (WHO 2003; NRC 2004).

In response to the known limitations of fecal coliforms and other existing IOs, the U.S. EPA recently convened a “state of the science” workshop of technical and regulatory experts to evaluate the future use of indicator bacteria and other IOs in recreational water quality criteria (EPA 2007). The purpose of the workshop was to provide information to EPA concerning the “critical path research and science needs for developing scientifically-defensible new or revised... recreational ambient water quality criteria (AWQC) in the near term.” Near-term activities were defined as those that could be accomplished in 2 to 3 years, so that results would be available to EPA in time to support the development of updated criteria. The new criteria, which EPA hopes to develop by 2012, “must be scientifically sound, protective of the designated use, implementable for broad Clean Water Act purposes, and when implemented, provide for improved public health protection” (EPA 2007).

Given the limitations of the existing IO-based water quality criteria, and the fact that the U.S. EPA is currently working to revise and strengthen them, it is clear a water quality management strategy that relied solely on compliance with the existing IOs and criteria as a means to minimize human health risks in recreational waters would have a number of drawbacks. Several of those drawbacks have been summarized by the WHO (1999, 2003) and the NRC (2004):

- If management actions are implemented only in response to exceedances of the existing IO-based criteria, those actions will often be reactive (implemented *following* human exposure to potential health hazards) rather than proactive or preventive;
- In terms of the human health risks posed by fecal contamination of recreational waters, the highest risks are usually associated with human fecal material. Due to the “species barrier” (the reduced ability of a pathogen from one host species to infect individuals of a different host species), which affects the transmission of many pathogens, the density of pathogens of public health importance is generally assumed to be lower in non-human than in human excreta. Waters that have come into contact with non-human fecal material (e.g., from sources such as livestock, birds and wildlife) are thus thought to pose a lower risk to human health;
- Nevertheless, there are human health risks associated with pollution of recreational waters from animal sources, and some pathogens, such as *Cryptosporidium parvum*, *Campylobacter* spp. and *E. coli* O157:H7 can be transmitted through this route. Local knowledge of possible sources and environmental pathways of animal pathogens to humans should therefore be a component of the management effort;
- In addition to human and non-human fecal sources, coliform bacteria and several other currently-used IOs can arise from a number of non-fecal sources (e.g., soils, vegetation), particularly in sub-tropical and tropical waters (e.g., Fujioka 2001; NRC 2004), reducing their reliability as indicators of human health risk;
- Health risks posed by recreational waters typically show gradients of severity, correlated with the types and densities of pathogens present at different locations. Given the limitations of the existing IOs, in many circumstances these risks may not be adequately characterized by IO monitoring data alone.

THE “ANNAPOLIS PROTOCOL” ASSESSMENT APPROACH

To help address these limitations the World Health Organization and the U.S. EPA co-sponsored an international workshop, which was held in Annapolis, Maryland, to develop improved strategies for managing the microbial quality of recreational waters (WHO 1999, 2003). The workshop produced an innovative information assessment framework, designated the “Annapolis protocol”, which combines quantitative IO counts (provided by water quality monitoring programs) with site-specific evaluations of the potential health risks posed by local IO sources (provided by “sanitary inspections” or “contaminant source surveys”) to classify recreational waters based on their estimated suitability for whole-body contact. An overview of the “decision matrix” site assessment framework used in the Annapolis protocol is provided in Table 1.

The Annapolis protocol provides a technically-sound conceptual approach for managing human health risks among recreational water users (NRC 2004; U.S. EPA 2007), but some modifications are needed to make it applicable to Florida waters. For example, the protocol uses enterococci as the IOs on which the microbial water quality assessment is based, while fecal coliforms are the IOs that are currently used to assess the microbial quality of Florida’s recreational waters. Also, the protocol uses the 95th percentile IO value observed at a monitoring site to determine IO exceedances, while monitoring sites in Florida are evaluated by FDEP based on the percentage of samples that exceed the state’s 400 CFU/100 mL fecal coliform criterion (Chap. 62-303, Florida Administrative Code).

In addition, the “sanitary inspection” categories outlined in the protocol, while conceptually useful, de-emphasize management responses to situations in which non-human fecal sources may pose health risks in recreational waters. Adjustments need to be made to these categories to address situations in which non-human fecal sources (e.g., livestock, which are potential sources of *Giardia* and *Cryptosporidium* infections) are also an issue of concern for local resource managers.

In order to address these issues, the Hillsborough River BMAP participants developed a modified version of the WHO (2003) assessment matrix (TCC 2008), which was then applied to the six impaired WBIDs in the Hillsborough River basin (PBS&J 2008).

A MODIFIED MATRIX FOR USE IN THE HILLSBOROUGH RIVER WATERSHED

The site assessment approach developed for the Hillsborough River BMAP involves the following steps:

- First, microbial water quality conditions within each WBID are categorized based on fecal coliform concentrations observed in the available monitoring data;
- For characterization purposes, each sampling station within a WBID is assigned to a microbial water quality assessment (MWQA) category. The MWQA categories are symbolized as letter grades (A through E) reflecting how frequently the State’s fecal coliform criterion of 400 CFU/100 mL is exceeded at a given site. Because sites with higher frequencies of criterion exceedances also tend to exhibit higher overall concentrations of fecal coliforms and enterococci, based on long-term monitoring data provided by the EPCHC (EPCHC; Figure 2), MWQA categories A through E also represent progressively higher indicator organism concentrations and increasing levels of potential human health risk;
- Next, for sites at which more than 10% of the samples exceed the State’s 400 CFU/100 mL criterion, contaminant source surveys (CSS) are carried out in order to identify the types of probable sources that could contribute to the elevated bacterial concentrations occurring at the site and to characterize their potential human health risks;

- Following the concepts outlined in the Annapolis protocol, each surveyed site is placed in a CSS category (ranging from “very low” to “very high” levels of potential risk; see below) reflecting the types of probable bacterial sources found in the vicinity of the site and the estimated likelihood that those sources pose human health risks;
- Following a “phased monitoring” concept recommended by the NRC (2004), the intensity of CSS investigation that a site receives is based on its MWQA classification. That is, sites that exhibit more frequent (and higher magnitude) exceedances of the State’s 400 CFU/100 mL fecal coliform criterion (e.g., sites in MWQA categories C, D or E) are subject to more intensive CSS investigations than sites exhibiting less frequent (and lower magnitude) exceedances;
- Once the MWQA and CSS analyses are completed, each site receives a two-part classification (summarized in Table 2), based on the MWQA and CSS categories into which it has been placed;
- These classification outcomes can be provided to policy-makers and the public, along with a summary of the management actions that have been implemented in the impaired WBIDs and any changes in water quality that have occurred there.

The modified assessment matrix is shown in Table 2. Following the approach used in the Tampa Bay water quality decision matrix (e.g., Sherwood 2009), cells in the modified matrix are color-coded (green, yellow, orange, red or magenta) to provide a visual indication of the gradient of potential human health risks associated with different matrix outcomes. The “sanitary inspection” component of the matrix - which is used to rank sites based on the types and magnitudes of potential fecal contaminant sources observed in their hydrological catchments - has been revised to incorporate non-human fecal sources and has been renamed the “contaminant source survey” (CSS) component in the modified matrix. The CSS categories developed for the Hillsborough River WBIDs have been defined as follows (PBS&J 2008) to provide a qualitative assessment of the likelihood that fecal contamination posing human health risks would be encountered by recreational users at a given site:

1. Very Low: No visual evidence of potential sources of human pathogens; natural environment; no or minimal anthropogenic land uses; wildlife present (any density);
2. Low: Low density agricultural and residential sources, including pets, livestock (without direct access to surface waters), or poultry operations; residences on septic systems;
3. Moderate: Urban stormwater sources (including pet waste) present; well-functioning wastewater infrastructure (both sewer and septic); episodic/low volume sanitary sewer overflows (SSOs) reaching surface waters; moderate-density livestock with little direct access to surface waters; Class A residual and/or septage spreading areas may be present;
4. High: Major stormwater outfalls present; history of failing wastewater infrastructure (central sewer or onsite systems); episodic or chronic/high volume SSOs reaching surface waters; concentrated livestock without direct access to surface waters; residual/septage spreading (Class B) may be present; and
5. Very High: Current failing wastewater infrastructure; chronic/high volume SSOs reaching surface waters; concentrated livestock with direct access to surface waters; evidence of direct sewage inputs (e.g., confirmed illicit discharges).

MICROBIAL SOURCE TRACKING (MST) FINDINGS

Because the Hillsborough River BMAP was a FDEP-supported pilot study, state funds were also available to support the collection and analysis of microbial source tracking (MST) data at a number of project sites, providing additional information on possible IO sources and their potential health risks. As noted earlier, epidemiological evidence indicates that, as a general rule, water contaminated by human fecal material poses a substantially higher risk to public health than

water contaminated by fecal material from non-human organisms (WHO 2003; NRC 2004). Among non-human sources, livestock and household pets are thought to pose the next highest levels of risk. This is due to their periodic excretion of potential human pathogens (e.g., *Giardia*, *Cryptosporidium*, and several pathogenic bacteria), and the fact that bacterial pathogens excreted by these animals can exhibit resistance to antibiotics that are commonly used in human and veterinary medicine. Wildlife species can also act as sources of *Giardia*, *Cryptosporidium* and other human pathogens. However, wildlife populations are usually present at lower densities than livestock or pets in many agricultural, suburban and urban areas, and are not treated with the antibiotics that can encourage the development of antibiotic-resistant strains. Under most circumstances the presence of fecal material from wildlife sources in recreational waters is therefore thought to represent a lower risk to human health than the other source categories (WHO 2003; NRC 2004).

Several biological MST markers were used in the Hillsborough River project (PBS&J 2008) to identify sources of fecal contamination, including:

- Human polyomavirus (HPyV), a nonpathogenic virus with a high carrier rate among humans;
- A genetic marker for a bacterial virulence factor (an enterococcal surface protein [*esp*] found in the bacterium *Enterococcus faecium*) which is specific to human sources (PBS&J 2008); and
- Genetic markers for human-specific, ruminant-specific and horse-specific forms of *Bacteroidetes*, a family of anaerobic bacteria found in human and animal intestines (and which includes the well-known genus *Bacteroides*);

The presence of one or more of these markers in surface water samples provides evidence of contamination by human, horse, or ruminant (e.g., cattle, deer) fecal sources. The percentage of sampling dates on which the human, horse, and ruminant-specific markers were detected can be used to estimate the relative frequency of fecal contamination from these sources at a given site.

Markers for additional livestock-related sources, such as hogs or poultry, were not included in the MST study. It appears that neither of these species is currently raised on a commercial scale in the project area. However, chicken manure is applied to pasturelands as a fertilizer source and soil amendment in portions of the Blackwater Creek WBID, and inclusion of a marker capable of identifying this source (if available) would be appropriate if future MST studies are conducted in the basin.

Because of the generally higher public health risk posed by human fecal material, monitoring locations exhibiting high frequencies of one or more of the human markers (HPyV, *esp*, or human-specific *Bacteroides*) should be given the highest priority for follow-up management actions to identify and address specific contaminant sources. Locations showing elevated frequencies of the ruminant specific or horse-specific *Bacteroides* markers should be given the second highest priority. Because the ruminant-specific *Bacteroides* marker may be an indicator of a wildlife-based (e.g., deer) source, additional information will be needed to confirm the types of sources present at locations showing high frequencies of this marker.

At locations where the CSS provides no evidence of human or livestock-based sources of fecal contamination, wildlife or other environmental factors are assumed to be the primary sources of any indicator bacteria or ruminant-specific MST markers observed in monitoring data. These locations can be given the lowest priority for follow-up management action, and it is assumed that under most circumstances no actions will be taken to address wildlife sources affecting Class III waters.

WEIGHT-OF-EVIDENCE ASSESSMENT RESULTS

A prioritized list of WBIDs, and of MST monitoring locations within WBIDs, which was produced using a weight-of-evidence approach combining all the information outlined above, is shown in Table 3. Within a WBID, individual monitoring sites were ranked from higher to lower priority based on classification matrix (TCC 2008) outcomes, which were determined using CSS category assignments and MWQA scores, and on the percentage of samples showing evidence of the human-specific or ruminant-specific markers (PBS&J 2008). (Confirmed detections of the horse-specific *Bacteroides* marker were uncommon in this study, occurring only once each in samples from the Spartman Branch and Blackwater Creek WBIDs, and were therefore not used in the prioritization process.) Summaries of the most probable sources of fecal contamination (as reported by PBS&J 2008) and recommended management actions for each site are provided in Table 4.

Among six WBIDs examined in this project, the Lower Hillsborough River (WBID 1443E) appears to be the highest priority for follow-up management action. This section of the river receives a great deal of recreational use, had a large number of sites falling in CSS category 4 and generated the largest mean CSS rating in the updated surveys, exhibited high frequencies of human MST markers at several monitoring sites, and appears to receive inputs from a number of sources of human fecal contamination. Sanitary sewer infrastructure, malfunctioning septic systems, homeless residents and stormwater sources appear to be the highest priorities for management action in this WBID (Table 4a).

The Blackwater Creek (WBID 1482) and Baker Creek (WBID 1522C) WBIDs form a second priority group, with a preponderance of MST monitoring sites falling in CSS categories 3 and 4 and high frequencies of human and ruminant fecal markers detected at several locations. The Hillsborough River Reservoir, a Class I waterbody from which public supply withdrawals are made by both the City of Tampa and Tampa Bay Water, is located downstream of these WBIDs. In addition, Baker Creek also discharges to Lake Thonotosassa, a highly-eutrophic Class III waterbody that is on the State's Surface Water Improvement and Management (SWIM) priority list due to poor water quality (FDEP 2005). Primary sources of MST markers in the Blackwater Creek WBID appear to be septic systems, livestock, and discharges of treated effluent from either publicly or privately owned wastewater treatment plants, while malfunctioning septic systems and livestock appear to be the primary sources of markers detected at sites in the Baker Creek WBID (Table 4b).

The Flint Creek (WBID 1522A), Spartman Branch (WBID 1561) and New River (WBID 1442) WBIDs form a lower priority group, with most MST monitoring sites classified in CSS group 2 but with elevated frequencies of human and ruminant fecal markers sporadically detected at some locations (Table 4c). All three discharge to the Class I Hillsborough River Reservoir. Sources in the Flint Creek WBID have not yet been clearly defined but appear to include a mixture of human, agricultural, and wildlife contributions. Primary sources in the New River WBID appear to be septic systems, livestock and wildlife. A mixture of human and wildlife sources are suspected in the Spartman Branch WBID.

Regardless of the priority group in which a WBID falls, it is recommended that significant human and livestock fecal sources identified at any of the MST monitoring locations should be addressed as a means of reducing pollutant loads to downstream receiving waters such as the Hillsborough River Reservoir and Tampa Bay.

BMAP MANAGEMENT ACTIONS AND PROJECTS

An important result of the BMAP process in the Hillsborough River Basin has been the very high level of local stakeholder participation and commitment. Enhanced communication and cooperation among basin stakeholders will have benefits beyond the BMAP. Together, stakeholders identified solutions to several of the basin's water pollution issues and are taking active steps toward achieving those solutions.

Signatories to the BMAP share a common goal of restoring the designated uses of impaired waterbodies in the Hillsborough River Basin. Their management actions/projects, including more than 75 specific projects identified in the BMAP, are anticipated to improve water quality within the impaired waterbodies. These management actions/projects cover a wide variety of pollutant sources and are categorized as follows:

- Basic Stormwater Management Programs;
- Education and Outreach Efforts;
- Agricultural BMPs;
- Regulations, Ordinances, and Guidelines;
- Special Studies, Planning, Monitoring and Assessments;
- Restoration and Water Quality Improvements; and
- Wastewater Infrastructure Management.

A generic summary of the number of projects by category and WBID is provided in Table 5. Individual projects may apply to more than one management category, and the numbers and types of projects vary from WBID to WBID. Basinwide management actions/projects are assumed to apply to all WBIDs. More detailed project information is provided in the BMAP document (HRBWG 2009) and supporting information, which is available through the FDEP website (<http://dep.state.fl.us/water/watersheds/bmap.htm>).

Considerable effort has been expended by the BMAP partners to understand the locations and types of potential fecal contaminant sources in the six WBIDs included in the BMAP project. Even so, the precise source(s) of fecal IOs and MST markers are still not well understood for some locations. Consequently, projects and actions identified in the BMAP also include continuing source identification efforts, and efforts to determine whether sources are primarily anthropogenic or animal based.

PROGRAM EVALUATION AND TRACKING

BMAP implementation will be a long-term process. Many unknowns remain with respect to fecal coliform sources, and the TMDLs established for the basin's rivers and creeks may not be achieved in the near term. However, several initial management actions/projects taken to identify and address sources of fecal coliform bacteria and fecal contamination have been successful.

In the initial phase of BMAP implementation the TS is tracking its projects and continuing to monitor water quality in the six WBIDs of interest (through existing water quality monitoring programs) to ensure that BMAP projects are carried out and to measure their effectiveness. The TS and BWG plan to meet annually to discuss implementation issues, consider new information, determine whether additional management actions or projects are needed in some locations, document progress, and summarize this information for forwarding to FDEP.

If significant changes have occurred during the year, a progress report will be created to discuss those changes and highlight potential alternatives for TMDL and BMAP implementation. Each

entity responsible for carrying out management actions/projects as part of the BMAP will submit information to the TBEP for incorporation into the progress report for submittal to the BWG and FDEP. The progress report will track the implementation status of any management actions/projects listed in the BMAP and document additional management actions/projects undertaken to further the water quality improvements in the basin.

As part of the BMAP development effort, the BWG has designed a water quality monitoring program which will be used to determine if water quality is improving and TMDLs are being met. Observations of water quality conditions and trends provided by the monitoring program will be reported to the BWG and FDEP as part of the annual BMAP meeting. The water quality data will be used to support the adaptive management process, assess projects, and identify possible needs for new management strategies or actions. A more complete analysis of trends in progress towards achieving designated uses in each of the impaired WBIDs will be made on a five-year basis, corresponding with FDEP's watershed management cycle.

In addition to this application to Class III (recreational) waters, it appears that similar conceptual approaches based on the Annapolis protocol could also be used for managing waterborne health risks associated with Class I (potable supply) and Class II (shellfish harvesting) waters. To do so, modifications would need to be made to the MWQA and CSS classification systems shown in Table 2 to better reflect the different types of potential health risks associated with those waterbody uses.

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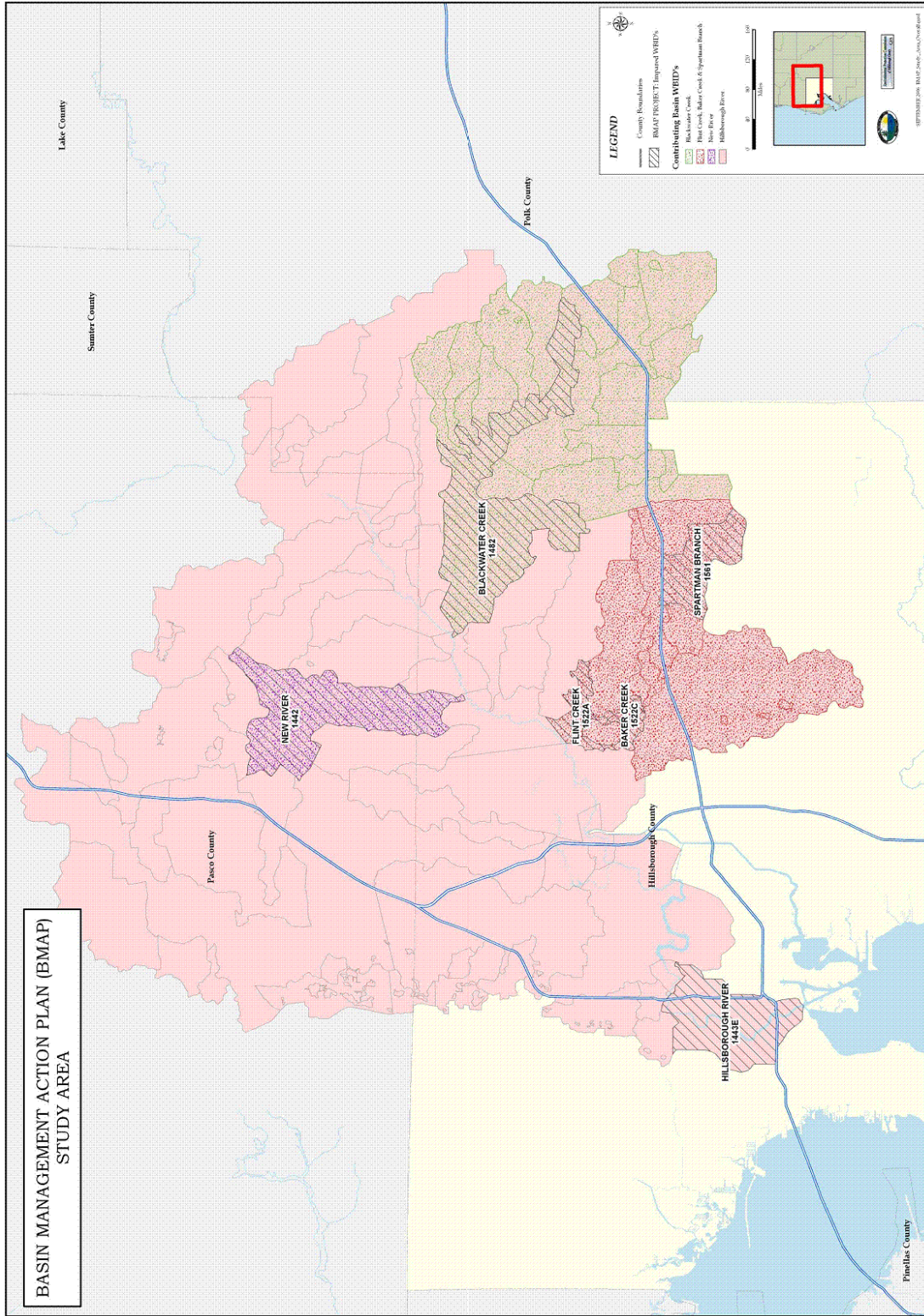


Figure 1. Project location map. (Source: HRBW G 2009)

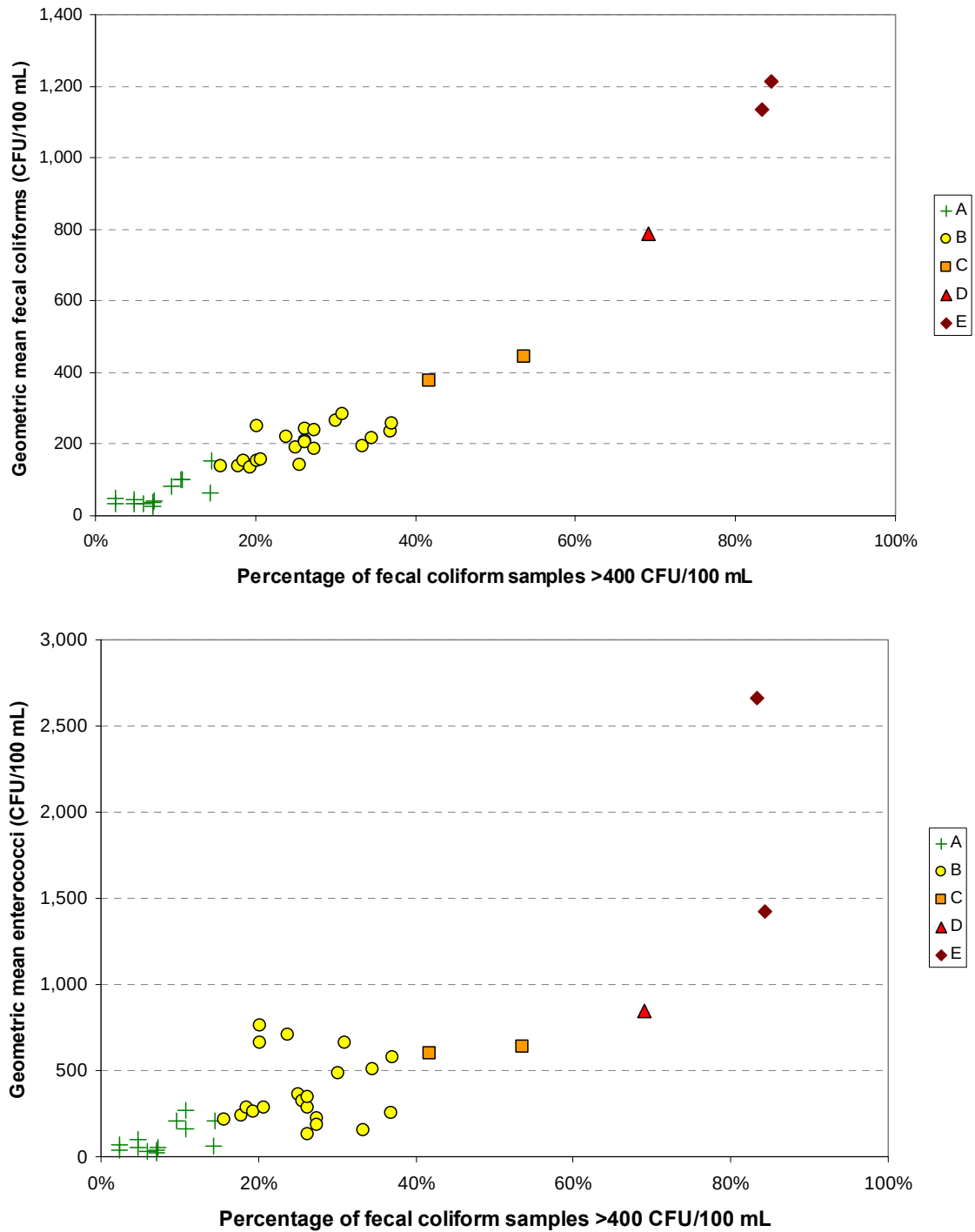


Figure 2. Geometric mean fecal coliform (upper panel) and geometric mean enterococci (lower panel) concentrations at the EPCHC monitoring stations shown in Figure 5. MWQA categories A through E were defined using the exceedance frequency break points of 10%, 30%, 50% and 75% described in Table 2. (Source: TCC 2008)

Table 1. WHO classification matrix summarizing the “Annapolis protocol”, which integrates microbial water quality and sanitary inspection categories. (Source: WHO 2003)

Microbial water quality assessment category (95 th percentile intestinal enterococci/100 mL)				
Sanitary Inspection Category (susceptibility to fecal influence)	A	B	C	Exceptional circumstances ³
	≤40	41-200	201-500	D >500
Very low	Very good	Very good	Follow up ¹	Follow up ¹
Low	Very good	Good	Fair	Follow up ¹
Moderate	Good ²	Good	Fair	Poor
High	Good ²	Fair ²	Poor	Very Poor
Very High	Follow up ²	Fair ²	Poor	Very Poor
Exceptional circumstances ³				
Action				

Notes:

- 1 Implies non-sewage sources of fecal indicators, which should be verified.
- 2 Indicates possible episodic contamination (e.g., rainfall- and runoff-driven), which should be investigated further, including sampling during and following storm events.
- 3 “Exceptional circumstances” relate to known periods of higher risk, such as sanitary sewer overflows (SSOs) in the vicinity of recreational water bodies. Under such circumstances the matrix may not represent risk/safety accurately.

Table 2. An alternative classification matrix, based on the WHO (2003) Annapolis protocol approach, using a combination of fecal coliform measurements (represented by the MWQA group) and contaminant source survey (CSS) information to rank recreational sites based on the apparent likelihood of human health risk. (See text for explanation of CSS assessment categories.) (Source: TCC 2008; modified from WHO 2003)

		MWQA group (based on binomial assessment of frequency of 400 CFU/100 mL fecal coliform exceedances)					Exceptional Circum- stances (e.g., sewer line break) ^c
		A (≤ 10%)	B (>10% - 30%)	C (>30% - 50%)	D (>50% - 75%)	E (>75%)	
Contaminant source survey (CSS) assessment category (likelihood of fecal contamination posing human health risks)	1. Very Low	A1	B1	C1 ^a	D1 ^a	E1 ^a	Immediate Action
	2. Low	A2 ^b	B2	C2	D2 ^a	E2 ^a	
	3. Moderate	A3 ^b	B3	C3	D3	E3	
	4. High	A4 ^b	B4 ^b	C4	D4	E4	
	5. Very High	A5 ^b	B5 ^b	C5 ^b	D5	E5	
Exceptional Circumstances (e.g., sewer line break) ^c		Immediate Action					

Notes:

- These outcomes imply that the CSS may be providing an overly optimistic rating of water quality, or the fecal coliform sources in the area may be relatively low-risk or primarily environmental (e.g., wildlife, sediments, soils, vegetation), and the cause(s) of the discrepancy should be verified.
- These outcomes imply that the fecal coliform indicator may be providing an overly optimistic MWQA rating, or the CSS may be providing an overly negative assessment, and the cause(s) of the discrepancy should be verified.
- As explained by WHO (2003), exceptional circumstances involve acute situations known to be associated with higher public health risks, such as sewer line breaks and other SSOs that contaminate surface waters, which require immediate remedial action.

Table 3. Priority rankings for further management action (from higher to lower priority) of WBIDs and assessment sites within WBIDs, based on classification matrix outcomes (using CSS and MWQA categories), and frequencies of MST marker detection. Site classifications are color-coded, ranging from green (least potential for human health risk) to purple (greatest potential for human health risk); sites in gray represent those areas where the CSS and MWQA scores provide different assessments and require further evaluation (TCC 2008). Only stations with MST data are included. (Source: PBS&J 2008)

Sub-Basin	WBID	Monitoring Location	Updated Classification Matrix Outcome	Sample Size (n) for Updated MWQA Score	Human Fecal Markers		Ruminant Fecal Marker	
					No. Sampling Dates	% Dates Marker(s) Detected	No. Sampling Dates	% Dates Marker Detected
Lower Hillsborough River	1443E	DHR4A	D5	5	5	100 %	5	20 %
		HR3	D4	8	5	80 %	5	20 %
		DHR5	D4	8	2	50 %	2	0 %
		DHR8	D3	5	4	50 %	4	0 %
		HR2	C4	8	3	67 %	3	0 %
		HR4	C4	8	3	67 %	3	0 %
		DHR7	C3	5	5	60 %	5	60 %
		DHR6	C3	6	3	33 %	3	0 %
		DHR9	C3	3	2	0 %	1	0 %
		HR1 REF	B3	8	3	100 %	8	0 %
		HR5	B3	8	5	60 %	5	0 %
		DHR10	A4	2	1	100 %	1	0 %
		DHR4B	A4	1	1	0 %	1	0 %
Blackwater Creek	1482	BW3	D4	4	4	75 %	1	0 %
		BW2	D4	8	3	67 %	3	100 %
		BW4	C3	2	1	0 %	1	0 %
		DBW4	A3	7	1	100 %	1	0 %
		BW5A	A3	2	2	50 %	2	0 %
		BW5	A3	2	1	0 %	1	0 %
		BW1 REF	A2	8	7	43 %	6	17 %
		BK2	D3	8	6	67 %	6	67 %
		BK1	C3	8	7	57 %	7	29 %
Baker Creek	1522C	BK3	C2	6	2	50 %	2	0 %
		BK4	A4	1	1	100 %	1	0 %
		BK5	A3	2	2	100 %	2	0 %
Flint Creek	1522A	FL2	C2	8	6	50 %	6	0 %
		FL1	C2	8	5	20 %	5	0 %
		FL3	A3	8	6	33 %	6	0 %
Spartman Branch	1561	SB1	D3	6	6	17 %	6	0 %
		SB2	D2	7	5	20 %	6	0 %
		DSB3	C2	5	2	0 %	2	50 %
		SB3	B2	8	4	25 %	4	0 %
New River	1442	NR2	C3	7	7	29 %	7	29 %
		NR1	C3	6	2	0 %	2	50 %
		NR3	C3	4	2	0 %	2	50 %

Table 4a. Priority ranking, most probable source categories, and recommended management actions for Lower Hillsborough River (WBID 1443E) MST sampling locations. (Source: PBS&J 2008)

Sub-Basin	WBID	Monitoring Location	Classification Matrix Outcome	Fecal Source(s) of Concern	% of Sampling Dates Markers Detected	Most Probable Source Categories and Recommended Management Actions
Lower Hillsborough River	1443E	DHR4A	D5	Human	100 %	Septic and possibly sanitary sewer; confirm and address specific source locations and provide public outreach explaining potential presence of health risk at site
		HR3	D4	Human	80 %	Septic and sanitary sewer; confirm and address specific source locations and provide public outreach explaining potential presence of health risk at site
		DHR8	D4	Human	50 %	Multiple sources suspected; identify and address specific source locations; provide public outreach explaining potential presence of health risk at site
		DHR5	D3	Human	50 %	Multiple sources suspected; identify and address specific source locations; provide public outreach explaining potential presence of health risk at site
		HR2	C4	Human	67 %	Sanitary sewer and stormwater; identify and address specific source location(s)
		HR4	C4	Human	67 %	Sanitary sewer and stormwater; identify and address specific source location(s)
		DHR7	C3	Human	60 %	Multiple sources suspected; identify and address specific source locations
		DHR6	C3	Human	33 %	Unknown; identify and address specific source location(s)
		DHR 9	C3	Human	0 %	Unknown; identify specific source location(s)
		HR1 REF	B3	Human	100 %	Homeless camp, sanitary sewer and possibly stormwater; confirm and address specific source locations
		HR5	B3	Human	60 %	Sanitary sewer; confirm and address specific source locations
		DHR10	A4	Human	100 %	Multiple sources suspected; identify and address specific source locations
		DHR4B	A4	Human	0 %	Unknown; identify specific source location(s)

Table 4b. Priority ranking, most probable source categories, and recommended management actions for Blackwater Creek (WBID 1482) and Baker Creek (WBID 1522C) MST sampling locations. (Source: PBS&J 2008)

Sub-Basin	WBID	Monitoring Location	Classification Matrix Outcome	Fecal Source(s) of Concern	% of Sampling Dates Markers Detected	Most Probable Source Categories and Recommended Management Actions
Blackwater Creek	1482	BW3	D4	Human	75 %	Public or privately owned wastewater facilities; confirm and address specific source locations; provide public outreach explaining potential presence of health risk at site
		BW2	D4	Human Ruminant	67 % 100%	Unknown human source(s); confirm and address specific source location(s); provide public outreach explaining potential presence of health risk at site Cattle; install fencing around watercourse
		BW4	C3	Human	0 %	Unknown; identify specific source location(s)
		DBW4	A3	Human	100 %	Sanitary sewer; ensure that upstream WWTP (City of Plant City) maintains adequate disinfection of treated effluent
		BW5A BW5 BW1 REF	A3 A3 A2	Human Unknown Human	50 % 0 % 43 %	Septic; confirm and address specific source locations Unknown; identify specific source location(s) Septic; confirm and address specific source locations
Baker Creek	1522C	BK2	D3	Human	67 %	Septic; confirm and address specific source locations; provide public outreach explaining potential presence of health risk at site
		BK1	C3	Ruminant Human Ruminant	67 % 57 % 29 %	Cattle and/or deer; install fencing around watercourse if cattle are confirmed Septic; confirm and address specific source locations Cattle; install fencing around watercourse if confirmed
		BK3	C2	Human	50 %	Septic; confirm and address specific source locations
		BK4	A4	Human	100 %	Septic; confirm and address specific source locations
		BK5	A3	Human	100 %	Septic; confirm and address specific source locations

Table 4c. Priority ranking, most probable source categories, and recommended management actions for Flint Creek (WBID 1522A) and Spartman Branch (WBID 1561) MST sampling locations. (Source: PBS&J 2008)

Sub-Basin	WBID	Monitoring Location	Classification Matrix Outcome	Fecal Source(s) of Concern	% of Sampling Dates Markers Detected	Most Probable Source Categories and Recommended Management Actions
Flint Creek	1522A	FL2	C2	Human	50 %	Likely septic; identify and address specific source location(s)
		FL1	C2	Human	20 %	Possibly upstream septic and livestock sources; identify and address specific source location(s)
		FL3	A3	Human	33 %	Not clearly defined; identify and address specific source location(s)
Spartman Branch	1561	SB1	D3	Human	17 %	Unknown; identify and address specific source location(s); provide public outreach explaining potential presence of health risk at site
		SB2	D2	Human	20 %	Septic; confirm and address specific source locations; provide public outreach explaining potential presence of health risk at site
		DSB3	C2	Ruminant	50 %	Wildlife sources suspected; no management actions suggested
		SB3	B2	Human	25 %	Unknown; identify and address specific source location(s)
New River	1442	NR2	C3	Human	29%	Septic; confirm and address specific source locations
				Ruminant	29%	Cattle; install fencing around watercourse upstream if confirmed
		NR1	C3	Ruminant	50%	Cattle and/or wildlife; install fencing around watercourse where cattle present
		NR3	C3	Ruminant	50%	Cattle; install fencing around watercourse upstream if confirmed

Table 5. Numbers of projects to be conducted by BWG members to implement the 2009 BMAP, by waterbody and project type. (Source: HRBWG 2009)

Waterbody	Project Category	Number of Projects
Blackwater Creek (WBID 1482)	Basic Stormwater Management Program	3
	Education and Outreach Efforts	5
	Agricultural BMPs	2
	Regulations, Ordinances and Guidelines	6
	Special Studies, Planning, Monitoring and Assessment	12
	Restoration and Water Quality Improvement	4
	Wastewater Infrastructure Management	13
New River (WBID 1442)	Basic Stormwater Management Program	2
	Education and Outreach Efforts	4
	Agricultural BMPs	1
	Regulations, Ordinances and Guidelines	2
	Special Studies, Planning, Monitoring and Assessment	9
	Wastewater Infrastructure Management	7
Spartman Branch (WBID 1561), Baker Creek (WBID 1522C), and Flint Creek (WBID 1522A)	Basic Stormwater Management Program	2
	Education and Outreach Efforts	10
	Agricultural BMPs	1
	Regulations, Ordinances and Guidelines	1
	Special Studies, Planning, Monitoring and Assessment	5
	Wastewater Infrastructure Management	13
Lower Hillsborough River (WBID 1443E)	Basic Stormwater Management Program	15
	Education and Outreach Efforts	10
	Regulations, Ordinances and Guidelines	1
	Special Studies, Planning, Monitoring and Assessment	7
	Restoration and Water Quality Improvement	3
	Wastewater Infrastructure Management	12

INDUSTRIAL STORM RUNOFF AND FLORIDA RECEIVING WATERS: ADEQUACY OF DATA FOR WATERSHED PLANNING

L.D. Duke

ABSTRACT

Effective decision-making for watershed planning and the BMAP (Basin Management Action Plan) process, in urban or industrialized watersheds, needs to include realistic estimates of pollutant loads originating with industrial storm runoff. This research investigates the sufficiency of available data on pollutants in industrial runoff, using: 1) statewide data about compliance with requirements under the NPDES program of the U.S. Clean Water Act; 2) information about municipal programs in various parts of Florida; and 3) a telephone survey of a sample of facilities in Pinellas County. Results demonstrated that a small portion of industrial facilities subject to NPDES stormwater regulations in Florida comply with the basic requirements of self-identification and monitoring of a limited number of constituents in facility runoff. Among those that self-identify, exemptions and allowed exclusions allow most to avoid monitoring. Data are insufficient to form a reasonable estimate of loadings to receiving waters or to reliably identify the location of the largest potential sources in any given watershed. Municipal programs in various parts of Florida are very unevenly implemented. Some watersheds have effective information and institutions in place to reduce pollutants if needed, while others have neither. Examples of active and effective outreach and technical support include Hillsborough County's efforts for the Port of Tampa and Jacksonville's inspection program for all industrial facilities. Other municipalities implement a diligent best-judgment effort to focus on facilities considered likely to contribute pollutants to receiving waters. Others implement the minimum activities that satisfy regulations. Finally, interviews with industrial facility operators suggest a few use monitoring data to identify and reduce pollutants in runoff, but many do not. Many newer facilities escape monitoring requirements by discharging to on-site retention or detention ponds, and do not substitute site-specific observations that would characterize pollutants that may enter groundwater.

INTRODUCTION

Regulations for industrial stormwater under the NPDES (National Pollutant Discharge Elimination System) program include, among other things, requirements for industrial facilities to monitor their discharges to quantify chemical constituents and possible pollutants in their facility's runoff. In most states of the U.S., state agencies implement NPDES, subject to oversight by the U.S. Environmental Protection Agency (USEPA). States vary in the kind of monitoring required, the frequency with which the regulated community is required to monitor, and the uses to which they put the resulting data, as well as other features of the implementation. In Florida, NPDES is implemented by the Florida Department of Environmental Protection (FDEP), which specifies monitoring requirements subject to approval by USEPA. Data acquired about constituents in industrial storm runoff via monitoring by the regulated community could be helpful in programs to protect watersheds, especially watersheds with extensive urban development and substantial proportions of industrial land uses. Information on loads of various constituents to Tampa Bay could be extremely useful in efforts by the Tampa Bay Estuary Program (TBEP), and others, to protect or restore the Bay's natural aquatic systems.

It is not clear that information gathered under the monitoring portion of NPDES is successful at providing data for any watershed planning purpose in the Tampa Bay watershed. A variety of factors limits the extent, representativeness, and reliability of the data collected, including exemptions for some facilities; loose guidelines that lead to very different data collection protocols, so that data cannot be aggregated in a meaningful way; guidelines designed to reduce the burden on businesses leading to very infrequent data collection; and highly incomplete compliance among the regulated

community. This paper explores the possible usefulness of data on stormwater runoff constituents collected under NPDES, and the reasons for which those data as currently collected are inadequate to meet any of the useful purposes intended under the regulations or identifiable as possible goals of watershed management agencies.

Possible Uses of Industrial Runoff Monitoring Data-

- A. Identification of high polluting facilities allowing agencies to focus outreach or regulatory efforts on facilities where the greatest pollutant reduction may be gained.

Purpose stated in MS4 and MSGP permits: FDEP 2004; FS 2000b; FDEP 2005

- B. Documentation of facilities' improvement in polluted discharges allowing agencies to verify progress in reducing loads to receiving waters

Purpose stated in MSGP: FDEP 2004; USEPA 1998a

- C. Facility operators' self source identification and quantification supporting future decisions about new equipment, operational changes, or other measures to reduce pollutant loads in runoff

Purpose stated in MSGP: FDEP 2004; USEPA 1998a

- D. Assessment of pollutant loads to receiving waterbodies from industrial facilities in a given watershed

For use in TMDLs, BMAPs, watershed protection planning, etc.

Possible Use A: Identification of High-Polluting Facilities--- It is reasonable to expect that some industrial facilities generate greater loads of specific constituents in their runoff than other facilities, by the nature of industrial activities: plants that handle substantial quantities of various chemicals, in equipment exposed to stormwater, are likely to have some of those substances enter the stormwater, while other facilities with only indoor activities such as assembly likely will have much less. The NPDES regulations for stormwater, like those for wastewater, recognize this variation (USEPA 1998a; USEPA 1998b). Monitoring results would be of value to local watershed managers if they were able to identify those facilities in a given watershed where runoff loads of a particular pollutant were higher than at other facilities. Agencies could then focus their time and resources for outreach, compliance, and enforcement on the "high-polluting" facilities to get the best return on resources in their efforts to reduce loads of selected constituents that may be causing problems in receiving waters. In Florida, the NPDES permits held by municipalities for discharges from municipal separate storm sewer systems (MS4s) in fact recognize this feature. The MS4 permits specify that permittees identify "high priority facilities" as part of their pollutant reduction programs. (FDEP 2005; Cross and Duke 2008). Reliable and complete monitoring data about discharges from facilities subject to the MSGP would facilitate that effort.

A single problem of the MSGP as currently implemented is sufficient to render it incapable of supporting that purpose. Compliance by the regulated community is known to be highly incomplete, including in the number of facilities that are identified as subject to the regulations and therefore complete any of the requirements. The structure of NPDES stormwater regulations is such that facilities self-identify, which leads to very poor understanding of the number of businesses subject to the regulations (Duke et al. 1999); and has led also to an unquantified, but very small, proportion of facilities complying with even the initial step of identifying themselves. Research has estimated the non-complying proportion at between 75% and 90% of those regulated (Duke and Shaver 1999; Duke 2001). Industries within the Tampa Bay watershed are no exception. Figure 1 shows results of research for Pinellas County industrial facilities (Cross and Duke 2008), which found the proportion of compliance to be between 10% and 25% of facilities required to comply in four targeted industrial

sectors. The estimate varies somewhat between industrial sectors and depending on interpretation of each facility's duty to comply, but each of the estimates suggests a startlingly small proportion of the regulated community is in fact acting to comply with regulations. The low compliance perhaps reflects in part resource limitations on the implementing agencies, who find the need to focus enforcement efforts on regulations with more direct and more powerful immediate impacts on the environment. In any case, few facilities in a typical urban watershed are actively complying with any of the permit, and therefore those conducting monitoring constitute quite a small proportion of those discharging stormwater associated with industrial activities. We cannot say with any confidence that the facilities with the greatest discharge of any pollutant of interest would be included in the small proportion conducting monitoring, so the monitoring program is not of value for this purpose.

Possible Use B: Documentation of facilities' improvement--- The MSGP specifies that another potential use of monitoring data is to allow agencies to determine any changes over time (FDEP 2004). That is a rational goal even if only a small number of facilities comply, because if FDEP or local agencies should identify facilities known or suspected to discharge pollutants in amounts that cause problems for receiving waters, they can use the regulations as a tool to verify changes over time.

However, facilities that do comply with the permit, and do conduct sampling in accordance with the permit requirements, do not generate sufficient data for this purpose because the Florida MSGP can be satisfied with very limited sampling. Table 1, adapted from Gleaton (2006), shows three types of monitoring required by complying facilities. The most frequent requirement is visual monitoring: facility operators are required to look at their stormwater discharges and report whether they see anything of concern. If they see anything suggesting further data would be advisable, they are expected to then collect a sample for laboratory analysis. Facility operators are considered to be in compliance if they report they have in fact visually observed their runoff once every three months. Visual examinations are intended to "provide a simple and inexpensive means of obtaining a rough assessment of stormwater quality at the facility" (Gleaton 2006, from USEPA 1999), and might detect such trends, but of course those results are limited to gross pollutants clearly visible at a glance and are not useful for identifying changes in chemical composition.

Table 1. Monitoring requirements under the Florida multi-sector general permit for stormwater discharges associated with industrial activity (FDEP 2004). From Gleaton (2006).

Monitoring	Sampling Frequency
Visual	Quarterly through the duration of the permit
Analytical	Quarterly sampling during the 2 nd and 4 th year of the 5-year permit (with exceptions)
Compliance	Annually

Analytical monitoring, in which facility operators collect a sample of runoff and have it tested for chemical constituents, is conducted very infrequently. The specified procedures, which is to collect a grab sample from any given storm (of rainfall greater than 0.1 inch) within each quarter, are very likely not representative of the facility's ongoing industrial operations or the runoff that results from them; this issue is fully discussed elsewhere. That feature is intended to reduce the burden on the regulated community, and is typical of permits implemented by USEPA and the states nationwide, but in return it is recognized to sacrifice any use of the data that requires an assumption they be

representative. In many states the data are required to be collected annually (see for example Duke et al. 2000), but the Florida requirements – even less burdensome to businesses – require analytical sampling only for two years out of five, and exempt facilities from one of those two years if their (non-representative) data show they do not exceed concentrations identified as atypically high by USEPA guidelines (USEPA 1995).

For purposes of identifying changes over time, the results are far too sparse. As an example, Figure 2 plots the entire set of collected data by a Pinellas County facility over a five-year permit span for one constituent, the metal zinc. There are seven data points. The data are shown to vary substantially. The research in Gleaton (2006) acquired data for all facilities reporting monitoring results under the MSGP in Pinellas County – a total of fewer than thirty facilities – and this sample facility had a greater number of data points for this particular constituent than any other facility.

Table 1 identifies “compliance monitoring” as a regulatory category affecting some facilities. This is required of facilities that are subject to NPDES wastewater regulations for their runoff – a small number of facilities, mostly limited to particular industrial categories such as primary metals smelting, petroleum refining, and others where runoff is considered to be a wastewater issue. That sampling is of course integral to documenting compliance by those facilities with the far more rigorous wastewater regulations, which happen to extend to runoff in their cases. It is not, in practice, used for facilities subject only to stormwater regulations.

Possible Use C: Use by facility operators to identify major pollutant sources on site--- Another stated intent of the monitoring requirements is so that a facility operator might use the results to identify any potential problems at the facility, and thus could be encouraged to take action to reduce pollutants. The infrequency of data collection described above is an important barrier to this: operators who see the chemical constituents in their runoff only once per quarter (limited to one year out of five) are likely to miss any short-term episodes of higher pollutant discharge caused by equipment malfunctions or varying production practices; and operators who see data only in one or two years out of five will receive very slow, if any, feedback about discharge pollutant changes resulting from a change in industrial operations.

A further limitation is that facility operators are instructed to take samples at the point where stormwater is discharged from the site. In Florida, sites operated for business purposes are subject to regulations that specify they treat stormwater by collecting it in retention or detention ponds on-site, thus reducing peak discharges, removing some pollutants with settling sediments, and directing some runoff into groundwater via seepage in the ponds. Data from facilities sampling at the discharge from these ponds to waters of the U.S. may be appropriate to estimate pollutant loadings into the receiving waters, which is the resource intended to be protected by the permit. However, sampling at that point does not give information about the originating source of the pollutant, because pond discharge includes water and pollutants that have been gathered over some period of time prior to a storm event, and also represents runoff that has been collected from all parts of the facility. In such a case it is not possible to attribute either flow or pollutants to source activities, and so an operator has no information about where, or when, any pollutants originated on the site. The operator cannot use this information to change industrial activities in a way that would reduce pollutants in runoff.

Table 2 displays results of a phone survey to manufacturing facility operators in Pinellas County, Florida. The table shows responses only from those who were identified as subject to the MSGP, and thus were expected to conduct monitoring in accord with that permit. Nearly all (97%) reported they routinely conducted visual observation monitoring. But even those visual observations were routinely limited to points of discharge from the facility (or “outfalls”), as stated by 70% of respondents. For half of these respondents, the outfall where monitoring was conducted was at the discharge from an

onsite detention or retention pond. Data of that sort – even if frequently-collected chemical concentration data, which these are not – will not give information about location or time of origin of any pollutants detected in the monitoring. Therefore these data are not useful for this purpose.

Table 2. Phone questionnaire to 42 industrial facilities, Pinellas Co. FL: What kind of stormwater runoff monitoring do you conduct? From Gleaton (2006).

<u>Visual Observations</u>
97% conduct visual observations
<u>Locations</u>
81% monitor at outfalls
50% “outfall” = from onsite retention / detention
30% on-site locations of production equipment, material handling, material storage
70% <u>do not</u> monitor other than at outfalls

Possible Use D: Quantification of pollutant loads from industrial runoff to particular receiving waters--- Watershed management efforts such as TMDL findings and waterbody protection plans require an effective quantification of the load of a target constituent, and a reasonable attribution of those loads to particular sources. Understanding the contribution of industrial activities can be a crucial component in some urban watersheds, perhaps including the Tampa Bay watershed. Monitoring data could be expected to provide an effective estimate. The sparsity, and the non-representative nature, of the monitoring data are such that any estimates based on specific industrial facilities in a watershed would be highly imprecise. What is more, the data do not support development of a ‘loading factor’ for industrial land use generally, or for industry sectors in particular, of sufficient precision for watershed assessment.

Figure 3 (adapted from Duke 2001) illustrates this imprecision, plotting the monitoring results for a reasonably large sample of facilities in Los Angeles County, California. The figure plots the median and quartiles of two substances in four separate industry sectors. Variation among sectors is not particularly meaningful, as the difference in median concentration is much smaller than the interquartile variation within each sector. Comparison of the means using a t-test (not shown) demonstrates no statistically significant difference among the means of the separate sectors. But the variation of median concentrations at all facilities, within each sector and with the four-sector sample as a whole, is so large as to render it useless for estimating total annual loads of any watershed: the interquartile measurements span a range of a factor of four to 10. That means, for example, that 75% of all metal-fabricating industrial facilities in this sample reported a concentration of copper in their runoff ranging from 5 mg/L to 35 mg/L. In a watershed with a small number of facilities in any one industry – typical of any urban region of less industrial intensity than a dense manufacturing district – selecting the median concentration for copper of about 10 mg/L could miss the true long-term average loading by a factor of two or three, even if we did not encounter any atypical facilities outside the 75% range, and even if we presume these data are representative and accurate for the full range of fabricated metal facilities in our target watershed. A factor of two or three has a substantial influence on attributing in-watershed loads for purposes of a TMDL or watershed management plan.

SUMMARY

Possible uses of industrial facility storm water runoff monitoring data--- The data are not useful for the four purposes identified here. Reasons are:

- Identification of high polluting facilities:
 - Compliance highly incomplete
- Documentation of facilities' improvement:
 - In Florida, many facilities among compliers are exempt from monitoring requirement
 - In Florida, monitoring is too infrequent to quantify, or even identify, change over time
- Facility operators' self evaluation:
 - Monitoring at outfalls does not identify source activities
- Assessment of pollutant loads to receiving waters:
 - Sparseness of compliance, and sparseness of monitoring among compliers
 - Questionable if representative
 - Much more variation than land use factors capture
 - No valid estimates by industry sector (concentration or load)

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Figure 1. Example of monitoring results: data summary, median and quartiles (mg/L) of four substances in four selected sectors, Los Angeles region, 1999 (adapted from Duke, 2001)

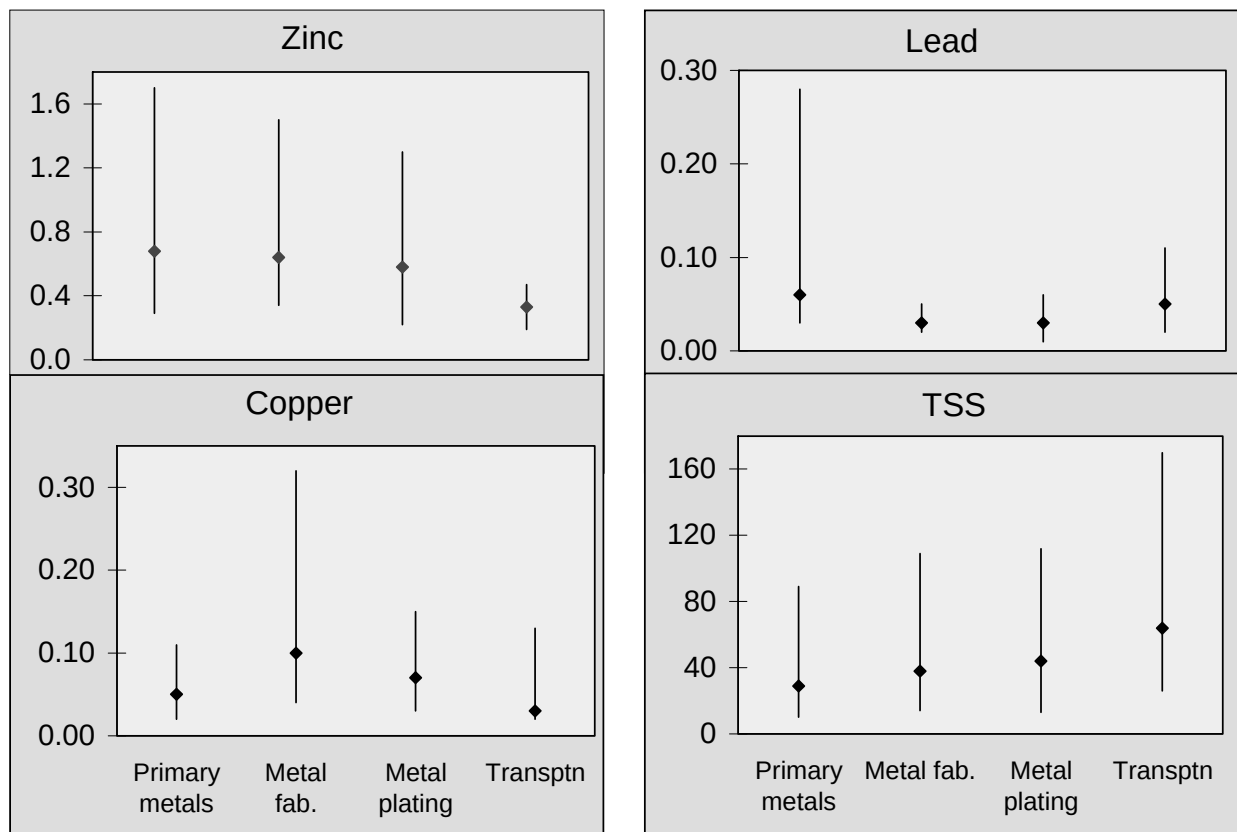
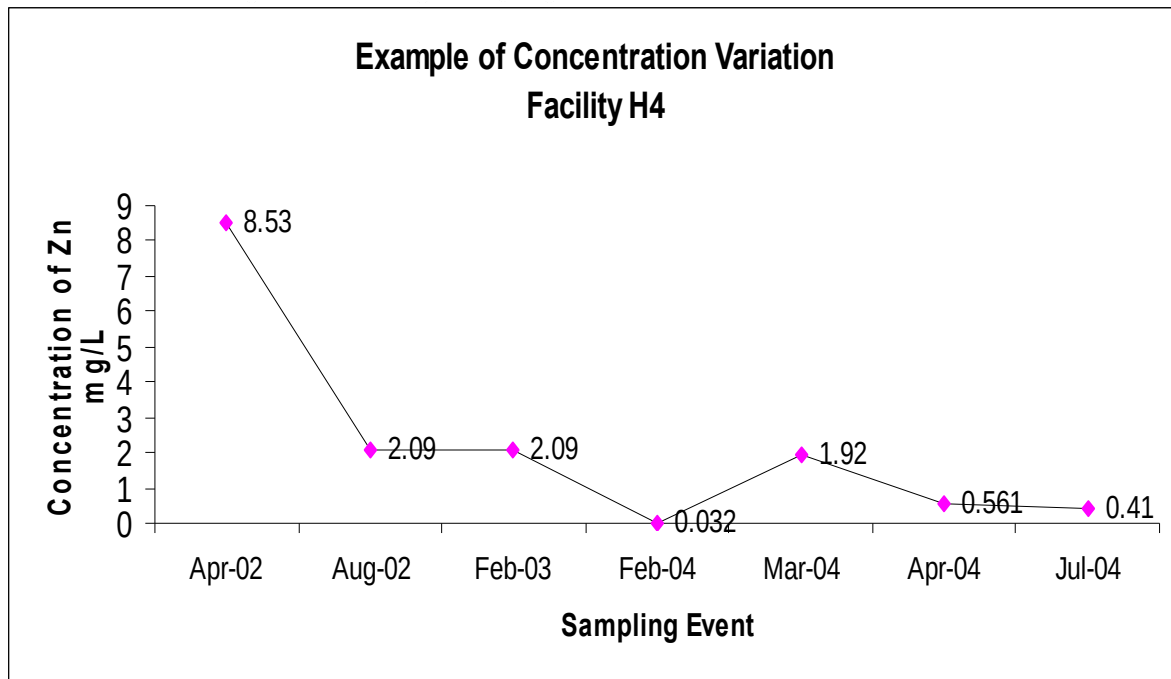


Figure 2. Compliance with Florida Multi-Sector Generic Permit by facilities in four industry sectors in Pinellas County, 2003, by “intensity of industrial activities.” Example of monitoring results: data summary, median and quartiles (mg/L) of four substances in four selected sectors, Los Angeles region, 1999. (adapted from Cross and Duke 2008)

Proportion of Facilities in Compliance with the Generic Permit

Industry Sector	Need not File (Not Industrial)	Probably need not File (Zero or Low Intensity)	Probably must File (Medium, High Intensity)	Actually Filed NOI	% Filed of Medium, High	% Filed of Total
Lumber	2	13	4	0	0%	0%
Stone	6	12	9	6	66%	29%
Metal	5	38	37	5	14%	7%
Electronics	8	12	11	3	27%	13%
Total	21	75	61	14	23%	10%

Figure 3. Example of full compliance with analytical monitoring (water quality constituent laboratory analysis) by one Pinellas County facility subject to Florida Multi-Sector Generic Permit, with no exceptions or exemptions (adapted from Gleaton, 2006)



ABSTRACT

Session 9: Integrating Watershed Management Initiatives

PRESENTATION

Regional and State Watershed Management Initiatives. S.T. Cooper, W.A. Wynne (Tampa Bay Regional Planning Council), J. Seachrist (Southwest Florida Water Management District), P. Wenner (Florida Department of Environmental Protection), and H.S. Greening (Tampa Bay Estuary Program). The Tampa Bay watershed is blessed to have a variety of agencies and organizations focused on understanding, restoring, and protecting its natural values. Research, regulation, and collaboration have been key to the Bay's health and progress toward recovery over the past 30 years. The Agency on Bay Management, formed in 1985 by the Tampa Bay Regional Planning Council, was the first quasi-governmental body to coordinate management and restoration efforts. Its membership includes all of the stakeholders and user groups within the bay in a forum to deal with Bay issues.

The Southwest Florida Water Management District (District) manages water and related natural resources to ensure their continued availability while maximizing environmental, economic, and recreational benefits. Resource management responsibilities include those for water quality, flood protection, water quality, and natural systems. As a regulatory agency, the District promotes fair allocation of the water resources, protects wetlands, enforces well construction standards, and protects against flooding and water quality degradation by new development. Additionally, the District helps fund local, state, and federal governments to implement projects that develop alternative water supplies, promote conservation, provide flood protection, and protect water quality and natural resources.

The FL Department of Environmental Protection's Division of Environmental Assessment and Restoration is tasked with the development and promulgation of water body-specific Total Maximum Daily Loads (TMDL's). Basin Management Action Plans are then developed to address how a specific basin would best be managed to ensure that TMDL's will be met. FDEP is also responsible for Aquatic Preserves, issuance of domestic and industrial wastewater permits as part of the NPDES program, and has Air, Hazardous Waste, Potable Water, Environmental Resources and Storage Tanks programs.

The Tampa Bay Estuary Program was created by Congress to protect and restore one of America's most important coastal waterways. The mission of TBEP is to build partnerships to restore and protect Tampa Bay through implementation of a scientifically sound, community-based management plan. Goals include maintaining adequate water quality to restore seagrass to 1950s levels, restoring the historic balance of coastal habitats, and creating a constituency of informed, involved citizens. The Program's Nitrogen Management Consortium is a public/private partnership formed to develop and implement a collaborative, watershed-based action plan to address nitrogen management goals and to meet regulatory water quality thresholds.

The Tampa Bay Regional Planning Council has statutory responsibility for the review of large scale developments as well as serving as a convener to resolve interlocal concerns and accomplish common goals. One Bay is a regional visioning partnership drafting recommendations and future land use strategies for sustainable growth. TBARTA is the regional organization developing a transportation corridor system to meet future growth needs. [suzanne@tbrpc.org]

MEETING RESTORATION GOALS IN THE TAMPA BAY BASIN THROUGH MITIGATION BANKING

R. Toth, L. Proenza & B.F. Birkitt

ABSTRACT

A key driver for development of mitigation banks is the new federal compensatory mitigation rule. This rule identifies the use of a mitigation bank as the preferential choice for mitigation for wetland impacts. State regulations are expected to follow this approach. This recognition of the benefits of mitigation banks provides opportunities to establish banks that meet goals identified within the Tampa Bay region. Proposed mitigation banks must demonstrate regional ecological benefits, effective management in perpetuity including financial responsibility, and a high probability of meeting success criteria, among others. Mitigation banks typically provide higher quality habitat than most on-site mitigation. With the current regulatory focus on mitigation banking and the significant ecological benefits they provide, it is anticipated that banking will play an increasing role in the future restoration and enhancement of Tampa Bay.

The Tampa Bay Mitigation Bank is the first mitigation bank established in the Tampa Bay Basin. The 161-acre bank is located in southeastern Hillsborough County adjacent to Cockroach Bay Aquatic Preserve. The headwaters of Andrews Creek are being restored, tidal creeks excavated, and approximately 42 acres of estuarine salt marsh and mangrove habitat created. Significant natural recruitment of mangroves and other estuarine vegetation has also occurred on site. Red mangroves and estuarine vegetation have been planted. Grading of oligohaline and freshwater portions of the property has been completed and native wetland vegetation will be planted in the near future, to create 85 acres of freshwater wetlands. Adaptive land management techniques are being utilized to ensure establishment of this important regional habitat.

Successful establishment of mitigation banks requires a marriage of science and economics. Many opportunities for restoration and enhancement of natural systems occur throughout the region. Other prospective mitigation banks involving significant areas of mangrove and aquatic habitat restoration within the Tampa Bay Basin are being evaluated.

INTRODUCTION AND BACKGROUND

As the population of the Tampa Bay area continues to increase and development within the region continues to impact vital wetland resources, effective means of providing compensatory mitigation and meeting regional wetland habitat goals are extremely important. While restoration efforts within the area have re-established a significant amount of regional wetlands, an additional means of promoting valuable habitat and meeting management goals is through effective mitigation within the region.

Mitigation banking has become a key component of successful compensatory mitigation, and mitigation banks have the opportunity to play an important role in counterbalancing the effects of past and continued wetland impacts in the Tampa Bay area. The following considerations are key to understanding the present and future state of mitigation banking in the region: 1) the ecological benefits of mitigation banking and the specific benefits that will affect the future of the bay; 2) the current condition of mitigation banking within the region, which includes Tampa Bay Mitigation Bank; and 3) the future of mitigation banking within the region as population and development increase. This discussion will enable us to gain an insight into the role of mitigation banking in meeting restoration goals within the Tampa Bay basin.

A wetland mitigation bank can be defined as a wetland or other aquatic resource area that has been preserved, restored, created, or enhanced for the purpose of generating credits that can be sold in order to compensate for unavoidable impacts to aquatic resource areas (EPA 2009). Mitigation banking is a practice which is gaining momentum as state and federal environmental regulatory agencies acknowledge the benefits of the concept. On June 9, 2008, the Department of the Army, Corps of Engineers (USACE) 33 CFR Parts 325 and 332 and the Environmental Protection Agency 40 CFR Part 230, implemented the federal compensatory mitigation rule which established a preference for mitigation banking. Within the State of Florida, Florida Statutes 373.4136, "Establishment and Operation of Mitigation Banks", establishes a framework for the formation and operation of mitigation banks in the state. Although the State of Florida has not yet incorporated mitigation banks as a preference in Rule or Statute, a trend toward this approach is evident.

ECOLOGICAL BENEFITS OF MITIGATION BANKS

Mitigation banks can provide significant ecological benefits to Tampa Bay that cannot be achieved through on-site mitigation. These critical ecological benefits include: 1) an opportunity for creation, preservation, restoration, and enhancement of critical and regionally significant habitat by restoring natural hydrologic regimes resulting in improved water quality within the watershed, 2) the ability to host a greater diversity of species due to the overall larger size of mitigation banks which generally contain a wider variety of habitat types, 3) the ability to support species with a greater habitat size requirement, increasing diversity in the area, 4) the ability to be less sensitive to environmental changes and natural disasters due to the larger, more diverse seed banks from the size and variety of habitats available in a bank, and 5) significant wildlife benefits by creating or enhancing foraging and nesting habitat for wading birds, ephemeral wetlands necessary for amphibian life cycles, and vegetative coverage for mammals. The grouping of several wetland habitats in a contiguous manner within a bank offers benefits over the typical on-site piecemeal approach of creating small, less productive wetlands on individual private properties. On-site mitigation projects typically are much smaller in size and are adjacent to developed areas. This situation results in comparably less wildlife use and wetland function. Typically, individual mitigation projects have a lower success rate than established mitigation banks.

There are also some practical considerations to take into account regarding how mitigation banking can offer more reliable benefits to the environment through the current regulatory frame work. The previously mentioned federal compensatory mitigation rule recognized the advantages of mitigation banking and established a preference for the use of banks in order to reduce some of the risks and uncertainties associated with compensatory mitigation. Mitigation banks are regulated through state and federal laws and are subject to a rigorous process to ensure the banks' viability in perpetuity. As projects that impact wetlands are permitted through the regulatory framework and compensatory mitigation is proposed to offset impacts, it is the responsibility of permitting agencies to confirm that the proposed mitigation is constructed as designed and that the mitigation activities achieve the required level of ecological success. Often agencies simply do not have the time or manpower to always ensure that on-site mitigation projects meet the level of success that has been proposed. Mitigation banks, however, are designed to meet more stringent rules; enforcement is more streamlined, ensuring a greater likelihood of ecological success. According to Florida Statutes, mitigation banks are also required to have a defined Mitigation Service Area (MSA) that allows mitigation only to occur within the same MSA as the wetland impact, other than a few exclusions. MSAs are typically defined by the basin limits in which the mitigation bank resides.

In addition, mitigation banking allows for the establishment of functional wetland systems well in advance of any project impacts. This advance implementation removes any "time lag" associated with compensatory mitigation. Time lag is the period of time between when functions are lost at an impact site and when those functions are replaced by the mitigation.

Finally, the creation and operation of a mitigation bank generally involves the application of more extensive planning and scientific expertise not always available to many permittee-responsible compensatory mitigation proposals. This greater concentration of scientific resources and design is more likely to lead to ecological success (EPA 2009). Recognizing these ecological benefits, the Society of Wetland Scientists (SWS) issued a position paper in 2004 supporting wetland mitigation banking to improve mitigation success and contribute to the goal of no net loss of wetlands (SWS 2004).

Additionally, the Tampa Bay Estuary Program (TBEP) has recognized a number of benefits of mitigation banking specific to Tampa Bay. In June of 2008, TBEP Technical Advisory Group issued a position paper regarding mitigation banks in the area (TBEP 2008). This document addressed the net environmental benefits and positive attributes of mitigation banks including: net benefits to wading birds in creating foraging and nesting habitat, the potential creation of Greenway Trails, the preservation of critical habitat, and the more stringent rules regulating banks versus individual mitigation that ensure ecological viability. Due to these benefits, the TBEP has supported the development of mitigation banks in the area; and, in its 2008 position paper, they stated, “mitigation banks should be encouraged on a watershed/regional scale. Both public and private mitigation banks should be developed in Hillsborough County. . . .(and) should be focused on preserving existing habitats now, rather than on complex re-engineering to acquire credits in the future (TBEP 2008)”.

The Tampa Bay Mitigation Bank, as well as other future banks, offers a number of benefits that cannot be achieved through on-site mitigation, provides a greater likelihood of ecological success through the regulatory framework, and offers specific advantages to Tampa Bay.

CURRENT MITIGATION BANKING IN THE TAMPA BAY BASIN - THE TAMPA BAY MITIGATION BANK

Currently, Tampa Bay Mitigation Bank (TBMB), located in southern Hillsborough County adjacent to the Cockroach Bay Aquatic Preserve, is the only permitted mitigation bank within the Tampa Bay Basin and is, therefore, the only bank available to compensate for impacts within the basin (Figure 1) as defined by the MSA. The MSA for TBMB includes portions of Hillsborough, Pasco, Pinellas, and Manatee Counties.

In its 1996 Management Plan “Charting the Course: The Comprehensive Conservation and Management Plan for Tampa Bay”, the TBEP documented significant loss of mangrove and salt marsh habitats along Tampa Bay’s southeastern shoreline (TBEP 1996). They also noted a decline in wading bird populations, such as the white ibis, due to the loss of freshwater marshes and ponds. Goals were identified to restore estuarine and freshwater wetlands habitat and small freshwater ponds which are critical foraging habitat. Tampa Bay Mitigation, L.L.C., the creators of TBMB, designed the bank to address the goals of the TBEP and to mirror the successes of the adjacent Cockroach Bay Habitat Restoration Project. The Tampa Bay Mitigation Bank site itself was also identified by the estuary program as a key location for restoration efforts.

Historically, the 161-acre TBMB site included an expansive mosaic of hydric hammock, swamp, pine flatwoods, and upland hammocks. Approximately 50 years ago, however, the site was converted into irrigated row crops and improved pasture (Figure 2). The headwaters of Andrews Creek were channelized for agricultural irrigation, and ditches were cut throughout the property to drain and lower surface and groundwater levels, significantly altering site hydrology. A concrete weir was also constructed across Andrews Creek, altering the hydrology of this tidal creek system. These hydrologic alterations significantly degraded the wetland habitat previously present on the TBMB site and nuisance exotic species such as Brazilian pepper populated much of the property.

Between 2005 and 2007, the TBMB site was graded and recontoured in order to construct the bank, and included the removal of the weir, which restored the hydrology of Andrews Creek (Figure 3). A tiered system of wetlands was established beginning with the upper freshwater wetlands, which flow into the lower saltwater marshes and mangrove forests. These varying depths provide seasonal and permanent habitats and create freshwater pond foraging habitat for protected species. Native estuarine species were planted in 2008 in tidally-influenced areas including red mangrove (*Rhizophora mangle*), seashore paspalum (*Paspalum vaginatum*), salt cordgrass (*Spartina patens*), seashore dropseed (*Sporobolus virginicus*), smooth cordgrass (*Spartina alterniflora*), saltgrass (*Distichlis spicata*), and sand cordgrass (*Spartina bakeri*). Additionally, natural recruitment of these species has been strong (Figures 4 and 5). Freshwater portions of the site have not yet been planted; however, planting is expected to occur in the spring of 2010.

An innovative aspect of the bank implementation was the use of shell rock mining for the majority of land excavation which assisted in funding the mitigation bank development. During construction, the overburden was removed and set aside. Shell rock was mined from the site and sold. The site was then recontoured to meet the design specifications for the wetlands creation and restoration plan to appropriate elevations for the establishment of wetland hydrology and planting of native wetland species. The overburden was redistributed and the site was prepared for planting.

Through creation, restoration, and enhancement, the TBMB was designed to provide a diverse variety of ecologically productive habitat types including mangrove forest, salt marsh, freshwater marsh and shallow ponds, and coastal and upland hammock. The TBMB contributes to the preservation of protected resources such as Threatened, Endangered, and Species of Special Concern, wading and migratory birds, Essential Fish Habitat, nursery habitat, and mangrove forests. These species and habitats are vital to the health and continuation of the rich freshwater and saltwater ecosystems that comprise the Tampa Bay region. The TBMB also directly influences the regional water quality and ecology in the Cockroach Bay Aquatic Preserve and Tampa Bay Estuary, and aids in the protection and preservation of vast amounts of diverse flora and fauna. Wildlife observed utilizing the bank has included the protected species: white ibis, limpkin, little blue heron, roseate spoonbill, snowy egret, and wood stork as well as many other species of wading birds (Figure 6). Other species that have been observed include the white pelican, glossy ibis, great blue heron, great egret, and marsh rabbit (Figure 7). With the amount of natural recruitment, wildlife observations, and succession of the restoration efforts, the TBMB currently appears to be trending toward success.

TBMB has been permitted to receive approximately 111 Southwest Florida Water Management District (SWFWMD) credits and approximately 103 USACE credits. To date, approximately 25 SWFWMD and 17 USACE estuarine and/or mangrove credits have been released for purchase based on the work completed. Freshwater/oligohaline systems within the bank are scheduled to be planted in spring 2010.

FUTURE OF MITIGATION BANKING IN THE TAMPA BAY BASIN

A variety of factors must be considered when attempting to formulate an outlook on mitigation banking in the Tampa Bay area. The successful establishment of mitigation banks requires a marriage of science and economics. The ability of a bank to come to fruition and develop is driven by the available market. The same laws of supply and demand that apply to any goods or services also drives the market for banks. In order to forecast mitigation banking in the area we consider the market forces at play that create demand for banked credits. The previously mentioned new federal compensatory mitigation rule establishes a preference for the use of mitigation banks for compensatory mitigation; state regulatory agencies have also indicated an interest in the use of banked credits. This preference creates demand for mitigation banks through the regulatory sector; therefore, those in need of wetland mitigation are more likely to purchase credits preferred by

regulatory agencies based on the likelihood that use of banked credits will decrease the time associated with obtaining a permit. From a supply standpoint, the highly developed nature of the Tampa Bay basin means that mitigation opportunities are limited; therefore, a need for mitigation banks in the area is apparent. Furthermore, recent sales for TBMB have also demonstrated a significant interest in utilizing a mitigation bank to meet the needs of a wide variety of projects within the Tampa Bay basin.

It is also vital when assessing the future of mitigation banking in the area to look at projected population and economics of the Tampa Bay region. According to the Florida Demographic Database, the population of the Hillsborough, Pinellas, and Manatee Counties, all within the Tampa Bay basin, increased steadily between the years of 1996 and 2006, before slowing in 2007. It is estimated that population growth will again increase, however, not to the level of growth seen within the region between 1996 and 2006 (Florida Demographic Database 2008).

From an economic standpoint, Sean M. Snaith, PhD., a widely recognized economist in the field of business and economic forecasting at the University of Central Florida, anticipates a slow recovery from the recent economic downturn within the state. Snaith predicted that new housing construction within the state would bottom out in 2009 and then climb at a very slow pace. These economic figures, along with population predictions, indicate that development within the Tampa Bay area, along with the corresponding need for compensatory mitigation, is likely to rebound from the recent economic downturn in the future (Snaith 2009).

There are also a number of major projects or expansions anticipated to take place within the region in the next ten years that could create further demand for mitigation bank credits. Transportation projects expected within this time frame include projects planned by Florida Department of Transportation (FDOT), the Tampa Bay Area Regional Transportation Authority (TBARTA), and by the local governments including Pinellas and Hillsborough Counties. Airport and port expansions planned within the area include Tampa International Airport, St. Petersburg-Clearwater International Airport, Tampa Port Authority, and Port Manatee. These large-scale projects all have the potential to impact a significant amount of wetlands habitat that may require the use of compensatory mitigation credits from mitigation banks in the basin.

In addition to these market indicators, there are financial and reduced risk-related advantages to using a mitigation bank. First, mitigation banks avoid the high costs of design and construction associated with on-site mitigation. Also, there is no long-term monitoring required for the use of mitigation credits, all responsibilities associated with meeting long-term success criteria are assumed by the mitigation bank. Due to these factors, mitigation banks are able to provide a level of certainty and banks are likely to become a preferred means of achieving compensatory mitigation. These advantages encourage the use of mitigation banking for compensatory mitigation and therefore increase the opportunities for banks within the region. Taking into consideration all of the factors that are likely to affect the supply and demand for mitigation bank credits, and as mitigation credits continue to be purchased from TBMB, it can be expected that mitigation banking will continue to expand within the Tampa Bay basin.

CONCLUSION

In conclusion, mitigation banks play a vital role in meeting wetland restoration and habitat planning goals within the Tampa Bay area. While there is currently only one mitigation bank in the Tampa Bay basin, TBMB, it is currently providing a host of ecological benefits to the area. With the amount of natural recruitment, wildlife observations, and succession of the restoration efforts, the TBMB currently appears to be trending toward success. Market indications show that demand for banked credits is likely to continue in the area and may increase in the upcoming years. Overall, it is

anticipated that we will see mitigation banks play an increasing role towards achieving restoration goals in the Tampa Bay basin.

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Figure 1. Tampa Bay Mitigation Bank Location Map.



Figure 2. Pre-construction Aerial of Tampa Bay Mitigation Bank (2004)



Figure 3. Post-construction Aerial of Tampa Bay Mitigation Bank (2008)



Figure 4. Natural recruitment and planting of native estuarine herbaceous species.



Figure 5. Natural recruitment and planting of mangrove species.



Figure 6. Juvenile white ibis foraging at the Tampa Bay Mitigation Bank.



Figure 7. White pelicans foraging at the Tampa Bay Mitigation Bank.

DISTRIBUTION OF NATIVE PLANT SPECIES ON ISLANDS OF THE TAMPA BAY AREA

T. Restom-Gaskill, J. Wolf & R. Runnels

ABSTRACT

Understanding the factors that affect the natural distribution of plants on islands is essential to the restoration success of severely disturbed insular ecosystems. The objective of this project was to understand the distribution of plants within and among islands located in the Tampa Bay area according to physical factors that could easily be measured by restoration planners. The factors chosen were soil salinity, soil texture and elevation. We surveyed vegetation on 12 islands in Pinellas (Anclote Key, Caladesi Island, "Plant Island" and Shell Key), Hillsborough (Big Pass Key, Camp Key, Egmont Key, Pine Key, Sand Key and Snake Island) and Manatee (Harbor Key and Mariposa Key) counties. Approximately 145 plant species were identified from 458 1-m² plots surveyed for ground cover (plants smaller than 1 m), and 449 25-m² plots surveyed for vegetation taller than 1 m. Species were found to occur in different ranges of soil salinity and elevation, and in different types of soil. Ordination techniques were used to identify species assemblages and how the different environmental variables affect their distribution. The results obtained will be used to formulate guidelines to improve the restoration success of island ecosystems in Tampa Bay.

INTRODUCTION

Coastal habitats have been subjected to enormous transformational pressure originating both from residential developments (including indirect effects such as hydrological changes and pollution), and from sea level rise as a result of global climate change (Vaughan 2006; Bassford et al. 2006; IPCC 2007). Coastal islands are particularly vulnerable to these changes because of their large shoreline to area ratio, their general low elevation, and their higher exposure to oceanic influence.

Islands are a major part of Florida's coastal landscape. The state ranks second in the United States in terms of number of islands, with over 4,000 islands greater than 10 acres (Fernald et al. 1992). Islands at or near the coast, specifically, provide a range of ecological functions from nursery for fish and marine invertebrates along their shore, to rookeries for birds and mammals, to protection of the mainland's shore from high energy wind and waves. Islands are also known for their high number of endemic species as a result of their isolated nature (Grant 1998).

High density of invasive species and high rates of erosion (due to vegetation changes and dredge-and-fill activities) have been jeopardizing the ecological functions of Florida's islands. In addition to the degraded naturally-formed islands, the west coast of Florida has dozens of artificially-formed islands (spoil islands), the result of dredge-and-fill activities in the 1900s. The Tampa Bay Aquatic Preserves, TBAP, is responsible for managing 70 to 80 of these islands, from restoring degraded natural islands to rehabilitating spoil islands in order to provide insular ecological functions. Whereas island mangrove ecosystems have been restored quite successfully, restoration approaches on the upland vegetation of these islands have been done on a trial and error basis due to lack of scientific data that addresses the unique characteristics of this area.

The objective of this project is to understand the distribution of upland plants in response to their physical environment in order to improve restoration success on islands of the Tampa Bay area. This paper provides preliminary results on the patterns of plant distribution according to soil salinity, soil texture and elevation. These environmental factors were selected because, in addition to their relevance to plant occurrence and distribution, they are easily measured and thus restoration planners will be more likely to adopt them.

METHODS

Study sites—This study was conducted on 12 islands on the Gulf coast of Florida, in the Tampa Bay area (Figure 1). Most of these islands are naturally-formed with a high density of native plant species, with the exception of Pine Key, a spoil island in the northeastern part of Tampa Bay (Hillsborough Bay). On Pine Key, the vegetation survey was done in the southern end of the island, in an area dominated by native plants.

Vegetation survey and environmental variables—The vegetation on each island was surveyed between March 2007 and August 2008 using the systematic plots method (Brower et al. 1998) in which plots were located 10 m apart along transects that ran across the width of each island. Each plot was 5 m x 5 m (25 m²), with 1 m x 1 m subplots (1 m²) in one of the corners. In the 25-m² plots we counted and identified all plant species with height ≥ 1 m (canopy species), whereas in the 1-m² plots we counted and identified all plant species shorter than 1 m (ground cover and saplings). Seedlings, considered not yet established, were not included in the survey.

A sample of most plant species found during the survey was collected for identification and stored in the Eckerd College herbarium. Identification was confirmed by comparing our samples to specimens in the University of South Florida Herbarium.

Elevation was measured at one of the corners of each small plot, and at two corners of each large plot, using a laser level (LB-10 Laser Beacon, Laser Alignment). The two measurements taken in the large plots were averaged as an estimate of the plot's elevation, and the relationship between them was used to calculate slope. A temporary reference mark was set up on each island prior to the elevation measurements. The real elevation of this reference mark was obtained later by relating it to an official benchmark of known elevation on the island or on the mainland.

Soil samples were taken from three areas within each plot and returned to the laboratory. Salinity was estimated using a soil paste technique (preparation of glistening pastes then the expression of interstitial water for analysis) and read with a refractometer (Richards 1954). Soil texture was also determined in the laboratory by touch (Brower et al. 1998).

Data analyses—The sampling units of this study were the plots, and data from the two plot sizes were analyzed separately. First we prepared the dataset by eliminating the three mangrove species (*Avicennia germinans*, *Laguncularia racemosa* and *Rhizophora mangle*), non-native species, individuals that were not identified to the species level, and species that occurred in less than 5% of the plots. McCune et al (2002) recommend eliminating rare species in order to reduce the noise in the community data and increase the power of the multivariate analyses.

The vegetation count data (+ 1, in order to avoid zeroes) were log transformed in order to reduce variation and skewness of the data. For the same reason, the environmental variables were also transformed as follows: soil salinity data (+ 1) were log transformed, and elevation data were square-root transformed.

We performed Cluster Analysis (R 2.9.0, R Development Core Team 2009; BiodiversityR package, Kindt et al, 2005) on the vegetation data using average as the linkage method, Bray-Curtis as the distance, and six as the pre-determined number of clusters. We used cophenetic correlation and Mantel test (100 permutations) in R to test the significance of the analysis. Discriminant Analyses (Minitab 14, Minitab Inc.) were also used to refine the clusters obtained.

The vegetation and community datasets were analyzed using Canonical Correspondence Analysis (CCA, R 2.9.0, BiodiversityR package), an ordination technique which constrains the vegetation data

by environmental variables. In addition to soil salinity, soil texture and elevation, the UTM coordinates of the plots were included in the analyses to assess spatial influences on plant distribution. Slope was also included in one of the models but did not improve the relationships between the vegetation data and the environmental variables. Because not all plots had a measurement for slope, we decided not to include slope in the models. Ordination plots were scaled based on species scores, and were tested with Monte Carlo test ($n = 100$ permutations). The log-transformed vegetation count data for the 1-m² plots were relativized by column (species) total due to the large range of abundance found within certain species.

RESULTS

We surveyed a total of 449 25-m² plots and 458 1-m² plots throughout the islands, covering an area of over 11,000 m². This area included 0.1 % (Anclote Key) to 8.3 % (Plant Island) of the upland area of the islands that confidently represents 1.1 % (Anclote Key) to 36.9 % (Snake Island) of the islands' upland vegetation area. Table 1 gives a complete summary of each island's characteristics.

After eliminating unidentified, undesirable, and rare species from the datasets, we obtained a matrix of 12 species x 330 plots for the analyses of the 25-m² plots, and a matrix of 16 species x 210 plots for the 1-m² plots. The clusters obtained for each dataset are shown in Table 2.

We plotted the data for the first two CCA axes in ordination graphs (Figures 2 and 3), where the influence of each environmental variable is represented by a vector. The length of the vector indicates the strength of the influence of that environmental variable on the vegetation data, and the direction of the vector indicates the direction of increase of that variable. Each symbol in the ordination graph represents a plot and is labeled according to the cluster to which it was assigned (Table 2).

Spatial UTM coordinates (latitude x longitude) were the variable that most influenced vegetation composition in the CCA for the 25-m² plots, with soil salinity and texture next (Monte Carlo test, 100 permutations, $p < 0.01$). Most of the vegetation clusters seemed to respond to the environmental variables involved in the analyses (Figure 2). The plots dominated by *Uniola paniculata* (UnPa) tend to occur in low salinity, sandy soils, and low to medium elevation. The plots dominated by *Serenoa repens* and *Quercus virginiana* (SeRe) tend to occur in higher elevation, low salinity, and in less sandy soils (ranging from loamy sand to silt loam). The plots dominated by *Coccoloba uvifera* (CoUv) tend to occur in high salinity, sandy soils of low to medium elevation. The plots dominated by *Caesalpinia bonduc* (CaBo) and the ones dominated by *Sideroxylon celastrinum* (SiCe) tend to occur in high salinity, medium elevation, and relatively sandy soils, with the plots within the cluster CaBo occurring in a little higher salinity than SiCe. The cluster SaPa is a very diverse and heterogeneous cluster and, as such, occurred in a wide range of environments (Figure 2).

Spatial UTM coordinates (latitude x longitude), especially longitude, and elevation were the variables that most influenced vegetation composition in the CCA for the 1-m² plots, with soil salinity and sandy soils next (Figure 3) (Monte Carlo test, 100 permutations, $p < 0.01$). Plots representative of the DiSp cluster occur in sandy soils of low elevation, in a wide range of salinity. Plots of the SePo cluster tend to occur on islands more to the west (Mariposa, Harbor, Big Pass, Sand), in high salinity, relatively sandy soil, and low elevation. Clusters represented by *Uniola paniculata* (UnPaS) tend to occur on islands to the east (Shell, Caladesi, Egmont, Anclote, but also Pine), in medium elevation, low salinity, and sandy soils. Plots of the clusters ToRa and QuVi tend to occur in high elevation, low salinity, and less sandy soils. Most of the plots of the cluster SpPa occurred on Pine Key, in high salinity, low elevation, and sandy soils.

DISCUSSION

Interestingly, spatial location (UTM coordinates) was the factor that most seemed to influence plant distribution on the islands surveyed in this study. This may reflect, however, the influence of factors not measured in the study, associated with the location of the islands. We believe that, rather than an east-west influence, plant composition differs between islands located inside Tampa Bay and islands located along the Gulf coast. These differences may be a result of factors such as distance to freshwater source (more abundant in the bay than on the coast), distance to seed sources (islands on the coast tend to be further from the mainland than islands inside the bay), and protection from wave and wind energy (islands along the coast are more exposed to these elements). All these bring up interesting hypotheses that can be further evaluated in future studies.

Even with the large influence of spatial location, the clusters were well discriminated based on soil salinity, soil texture, and elevation. Most clusters occurred in specific environments, such as clusters dominated by *S. repens*, by *T. radicans*, and by saplings of *Q. virginiana* clearly occur on less sandy soils, with low salinity and high elevation. On the other hand, clusters dominated by *S. patens*, and by *S. portulacastrum* tend to occur preferentially in high salinity, low elevation, sandy soils of the islands found inside the bay.

Our next steps will be to quantify the proportion of sand, silt and clay in each plot, and separate the islands into two groups, the ones inside the bay and the ones along the coast, in order to confirm the patterns described in this study.

We also found indications that some plant assemblages, particularly CaBo and SiCe, may tend to occur on middens which resulted from past Native Americans' activities. We did not have enough plots surveyed on middens in this study due to not being allowed to collect soil on them. It would be interesting to expand this survey to include more island middens in order to assess whether or not species assemblages associated with these features are different from the species assemblages away from the middens.

In sum, this preliminary study contributes to advances in the understanding of plant distribution on islands, and indicates promising easily measurable environmental factors that influence this distribution. Soil salinity, measured with the paste technique, and elevation, with the laser level, showed their clear influence in plant distribution. Even soil texture, measured just by feel, was enough to show the strong influence of sandy soils on plant distribution. Further analyses with quantitative data on sediment proportion will confirm whether or not categorical data for this variable is enough for the purpose of improving restoration success on islands of Tampa Bay.

CONCLUSION

In this preliminary study, we were able to identify plant species assemblages on islands of the Tampa Bay area, and the effect of soil salinity, soil texture, and elevation on the distribution of these assemblages. These relationships will be further used to improve restoration strategies for islands of the Tampa Bay area. We suggest that plant species composition of islands inside Tampa Bay differs from the composition of islands located along the coast, however, further analyses of the data are needed in order to confirm this observation.

Of all soil textures, sandy soils had the strongest influence on the distribution of the different plant clusters obtained in this study. If further analyses with quantitative data confirm this trend, estimating soil texture by feel may be enough to measure this variable, which would translate into easier procedures when elaborating a restoration plan for islands in the Tampa Bay area.

In conclusion, our data indicate that, in spite of the complexity of natural ecosystems, few environmental factors may be enough to allow us to understand the general occurrence of plant assemblages on islands of Tampa Bay in order to better plan their restoration. This understanding of island vegetation will also allow us to set restoration goals in the context of predicted changes in island area and elevation due to sea level changes.

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Table 1. Characteristics of the islands surveyed in this study.

	Andote	Big Pass	Calades i	Camp	Egmont	Harbor	Maripos a	Pine	Plant	Sand	Shell	Snake
Number of 25-m ² plots	41	32	38	35	66	36	24	36	14	27	57	43
Number of 1-m ² plots	41	33	38	38	66	34	24	37	18	27	57	45
Number of species found	37	36	50	31	17	53	33	25	16	36	36	38
Area surveyed (m ²)	1025	800	950	875	1650	900	600	900	350	675	1425	1075
Island size (km ²)	2	0.16	2	0.21	1.14	0.15	0.29	0.081	0.0042	0.25	N/A	0.15
Uplands area (km ²)	0.86	0.04	0.71	0.04	1.14	0.03	0.03	0.081	0.0042	0.04	0.66	0.04
Percentage of upland area surveyed	0.1	2	0.1	2.2	0.1	3	2	1.1	8.3	1.7	0.2	2.7
Elevation range (cm)	43 - 149	43 - 155	46 - 222	62 - 146	17 - 341	37 - 159	36 - 162	51 - 158	N/A	44 - 152	0 - 164	29 - 203
Salinity range (ppt)	0 - 2	0 - 17	0 - 18	0 - 30	0 - 3	0 - 37	0 - 29	0 - 5	0 - 40	0 - 20	0 - 22	0 - 70
Month/year surveyed	Jul 08	Jul 07	Jan 08	Jun 07	Jun 08	Mar-Apr 08	Feb 08	Jun-Jul 08	Jun 07	Jul 07	Oct-Nov 07	Apr-May 07

Table 2. Clusters obtained using Cluster Analysis (refined using Discriminant Analysis) on the vegetation data collected in plots (25 m²) and subplots (1 m²).

Cluster Code	Representative species	Common name	Number of plots
25-m ² plots			
SaPa	<i>Sabal palmetto</i>	cabbage palm	162
	<i>Eugenia axillaris</i>	white stopper	
UnPa	<i>Uniola paniculata</i>	sea oat	59
SeRe	<i>Serenoa repens</i>	saw palmetto	40
	<i>Quercus virginiana</i>	live oak	
CoUv	<i>Coccoloba uvifera</i>	sea grape	27
CaBo	<i>Caesalpinia bonduc</i>	nicker bean	25
SiCe	<i>Sideroxylon celastrinum</i>	saffron plum	17
1-m ² plots			
DiSp	<i>Distichlis spicata</i>	salt grass	74
	<i>Paspalum setaceum</i>	thin paspalum	
	<i>Fimbristylis</i> spp.	hurricane grass, fimbry	
SePo	<i>Sesuvium portulacastrum</i>	sea purslane	42
UnPaS	<i>Uniola paniculata</i>	sea oat	37
	<i>Chamaesyce mesembrianthemifolia</i>	coastal beach sandmat	
ToRa	<i>Toxicodendron radicans</i>	poison ivy	27
SpPa	<i>Spartina patens</i>	saltmeadow cordgrass	18
QuVi	<i>Quercus virginiana</i>	live oak	12

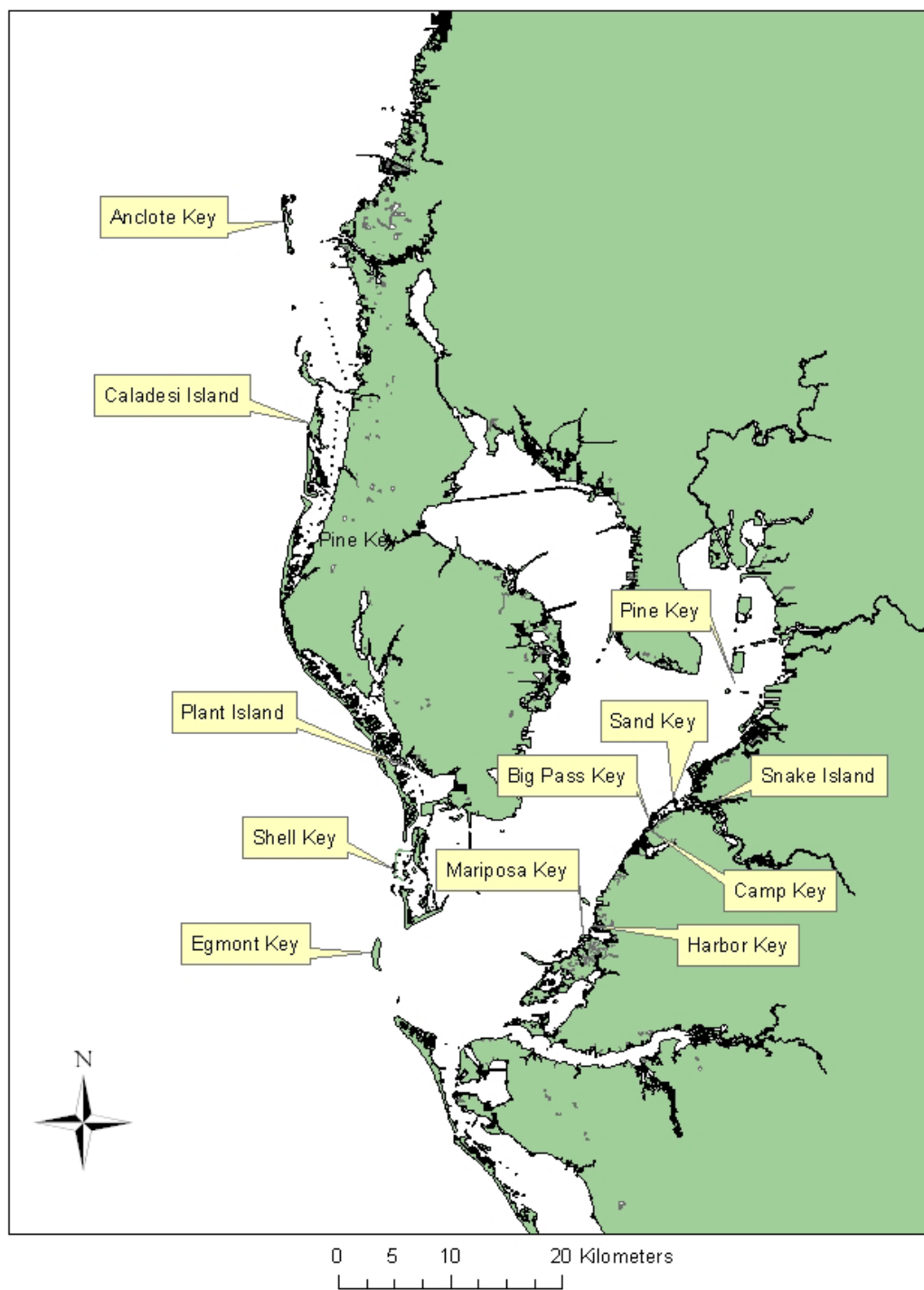


Figure 1. Location of the islands surveyed in this study, in the Tampa Bay area.

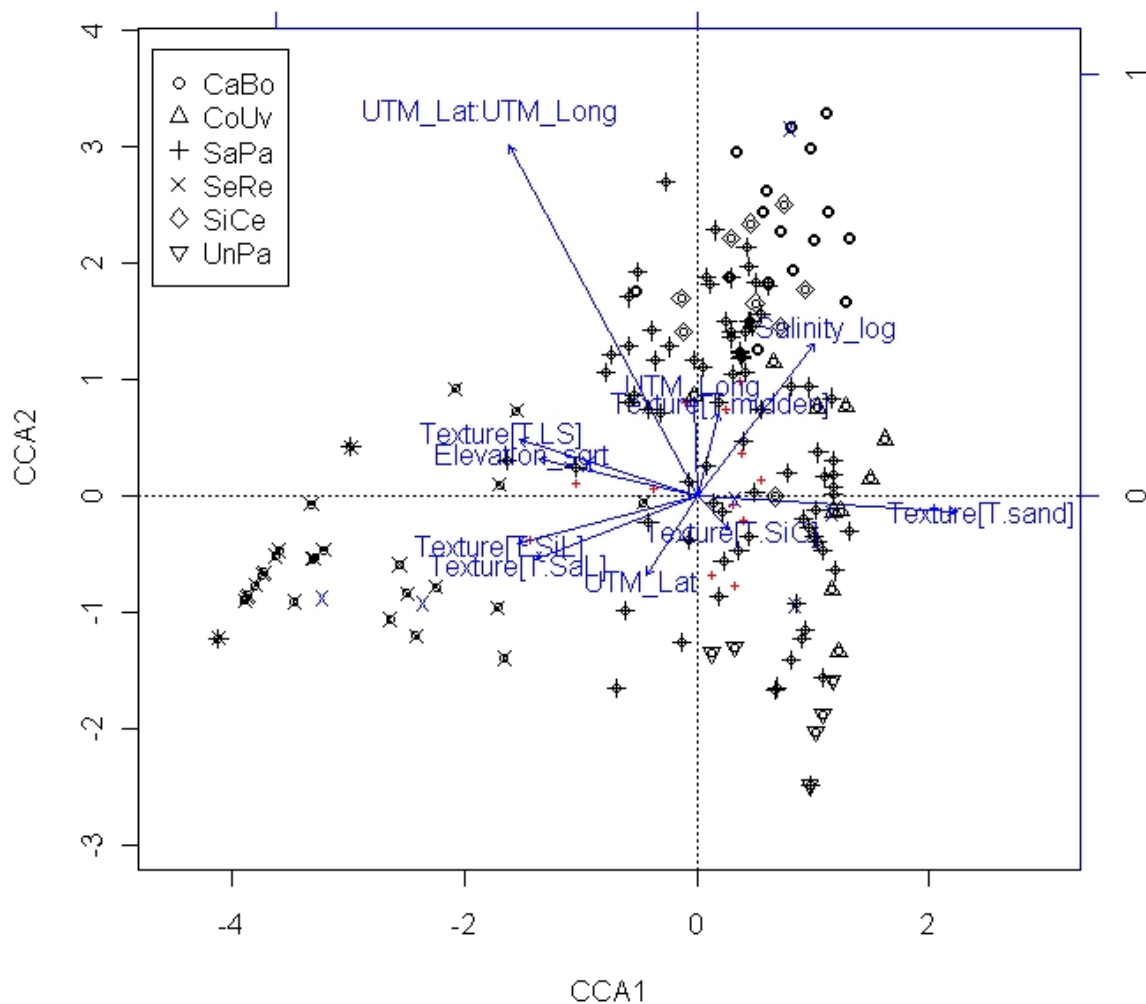


Figure 2. First two axes obtained through Canonical Correspondence Analyses of vegetation constrained by soil salinity, texture (categorical), elevation, and UTM coordinates for the 330 25-m² plots surveyed in this study. Symbols indicate clusters as explained in Table 2. The categories of soil texture included in the model are: sand, loamy sand (LS), sandy loam (SaL), silty loam (SiL), silty clay (SiC), and midden (no soil was collected from these).

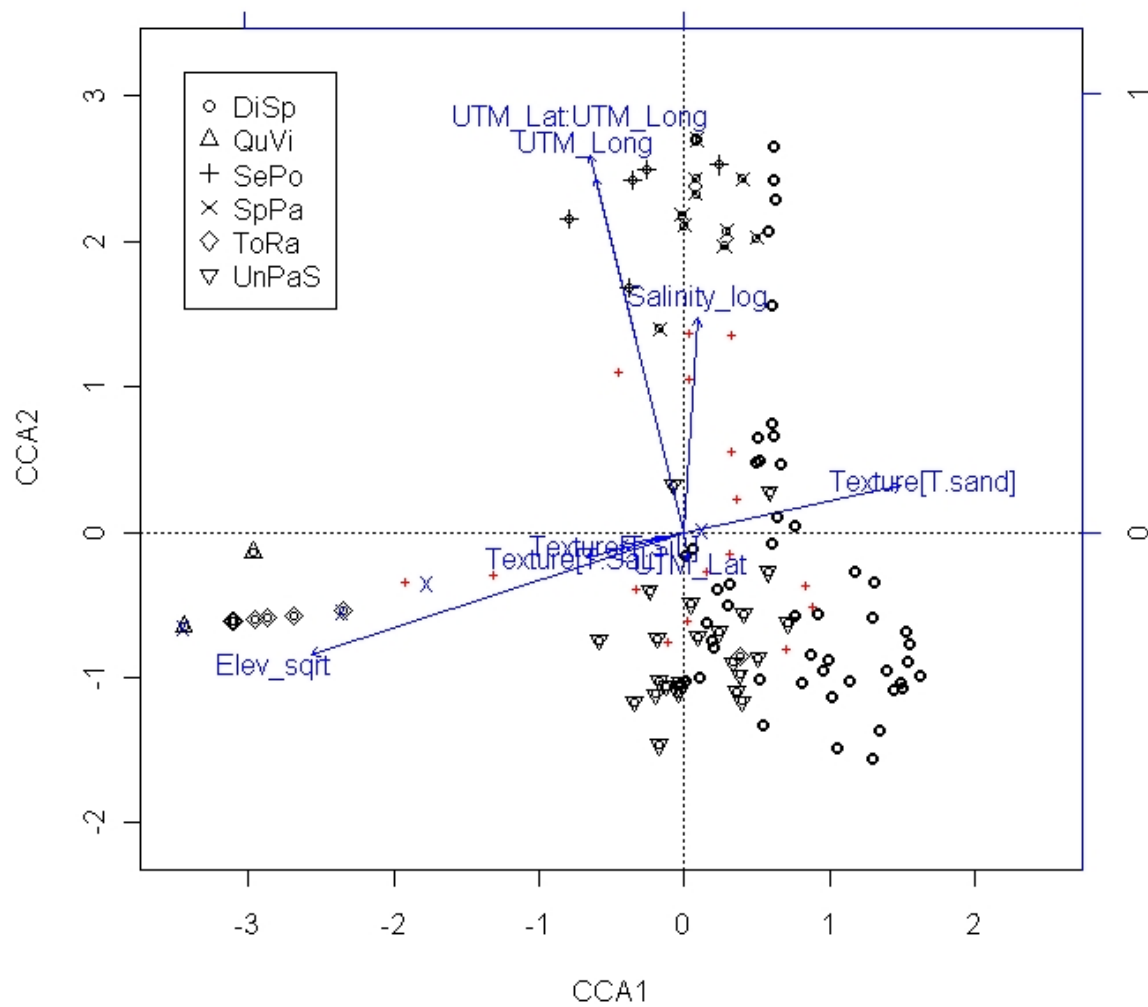


Figure 3. First two axes obtained through Canonical Correspondence Analyses of vegetation constrained by soil salinity, texture (categorical), elevation, and UTM coordinates for the 210 1-m² plots surveyed in this study. Symbols indicate clusters as explained in Table 2. The categories of soil texture included in the model are the same as for Figure 2.

FORTY YEARS OF TAMPA BAY AQUATIC PRESERVES

R. Runnels

ABSTRACT

August 1 represents the 40th anniversary of the establishment of Tampa Bay's first Aquatic Preserve in Boca Ciega Bay in 1969. Under the auspices of DEP's Office of Coastal and Aquatic Managed Areas, the Tampa Bay Aquatic Preserves program manages four areas of submerged lands designated by the legislature as state aquatic preserves in three counties.

To some extent, Tampa Bay's aquatic preserves are a microcosm of management needs and strategies of Florida's network of 41 Aquatic Preserves. Ranging from densely urbanized to relatively pristine, each of these areas presents a unique and dynamic suite of management needs. As with Florida's other aquatic preserves programs, the Tampa Bay program is place-based, science-based, adaptive and proactive. With information from scientists and citizens, the program identifies habitat degradation and other resource management issues within the preserve boundaries. Input from scientists and citizens also is used in developing and implementing strategies to address those issues.

By looking at landscape-level ecological function within relatively large areas, the Tampa Bay Aquatic Preserves program fills a unique niche that complements the missions of, and enhances the actions of, a variety of governmental and nongovernmental organizations. The program's outreach efforts include sharing information and ideas throughout the state, the Gulf of Mexico, and abroad.

Program History

As Florida's growing population in the 20th century continued to alter the coastline that had attracted so many settlers and visitors, citizens and officials began to reflect on the changes. Following the passage of the Bulkhead Act in 1957, dredge-and-fill activities continued to dramatically alter areas like Boca Ciega Bay. Out of growing concern for the unsustainable trajectories of habitat loss, the Randall Act was passed in 1967 to require more environmental oversight of these activities (Haman and Tucker 1997).

In addition to regulatory action, proprietary measures also were proposed. In a November, 1968 report to the Governor and Cabinet, the Florida Inter-Agency Advisory Committee on Submerged Land Management recommended the establishment of a system of "Aquatic Preserves". The areas proposed for this additional protection included some of Florida's remaining pristine coastal sites, as well as some urban areas that likely had been settled because of their abundant aquatic resources. On August 1 of 1969, the Florida Legislature established the Boca Ciega Bay Aquatic Preserve as the first of four aquatic preserves eventually to be established in the Tampa Bay area.

It is unlikely that the persons responsible for beginning Florida's system of Aquatic Preserves realized that their effort was decades ahead of many similar efforts in other states and countries. Individual marine protected areas (MPAs) were relatively unique, and systems of MPAs were practically unheard of.

In subsequent years, actions were taken to establish the active management of Tampa Bay's aquatic preserves. The first part-time manager was hired in 1984, and the first full-time career-service manager was funded in 1986.

Through the years, aquatic preserves have been managed under several regulatory and parks programs, but the unique management model they require to be effective eventually led to the establishment of a separate program within the Florida Department of Environmental Protection.

Present Administrative Structure and Management Responsibilities

The Tampa Bay Aquatic Preserves (TBAP) program is managed under the auspices of Florida DEP's Office of Coastal & Aquatic Managed Areas (CAMA). In addition to 41 aquatic preserves, CAMA partners with NOAA to manage three National Estuarine Research Reserves and the Florida Keys National Marine Sanctuary. The National Estuarine Research Reserves also serve as regional hubs for the aquatic preserves.

Tampa Bay's four aquatic preserves include the Boca Ciega Bay Aquatic Preserve, The Pinellas County Aquatic Preserve, the Cockroach Bay Aquatic Preserve and the Terra Ceia Aquatic Preserve. These four aquatic preserves cover nearly 400,000 acres of submerged land, with abundant seagrasses, corals and other habitats. The location of these preserves at a latitude of climatic transition places special importance on the protection of their biota.

Strategic Approach

As with other CAMA sites, TBAP plays a unique resource management role in a dynamic landscape of issues. To be effective, the program must be adaptive, science-based, place-based, collaborative and proactive.

With the primary goal of restoring and protecting ecological function and ecosystem services of vast areas of submerged land, TBAP must make efficient strategic use of its limited resources. The horizon for the program's strategic planning typically is at least ten years, but the strategic plan is reviewed on an annual basis. Within the context created by the strategic plan, staff prepare annual individual workplans, and these are reviewed frequently. This strategic planning cycle allows TBAP to maintain an unusual degree of adaptive management capacity.

While existing scientifically-derived information is vital to MPA management, TBAP also actively engages the scientific community to provide feedback on resource management informational needs. This two-way flow of information between resource managers and researchers has resulted in experimental designs that provided specific answers for resource management decisions. Fortunately, the Tampa Bay area is relatively rich in research and information. If necessary, TBAP sometimes conducts research in-house or collaborates with other researchers, but information often is already available for informed decisions. In addition to better defining baseline conditions in the preserves, scientific inquiry helps to understand the resources in the context of a changing watershed and environment. Rather than merely trying to turn back the clock, the program works to understand how habitats and species may function best in the present reality of bridges, channels and other alterations.

Much of TBAP's capacity to address issues in a timely, cost-effective manner is the result of numerous collaborative efforts with local citizens, organizations and programs. Local citizens, scientists and officials often are instrumental in identifying problems while their resolution still is relatively easy and inexpensive. Similarly, local individuals and organizations, in association with CAMA's statewide perspective of similar issues, often play an active role in finding and implementing solutions. In this manner, "cookie cutter" actions are avoided, and each issue can be addressed within the unique context of each site. An example of this approach is the program's history of gradually replacing exotic trees on islands with native trees. Using local volunteers, this can be accomplished in a low-cost, gradual process that avoids the controversies of large-scale removal of shade trees the public enjoys.

TBAP's approach fits well with the evolution of environmental protection toward proactive, preventive measures. The program is actively engaged in applying proactive tools like the Florida Clean Boating Partnership to the sites it manages. These efforts provide a means for the vast majority of Floridians who want to "do the right thing" to do so in an informed manner. Like public health issues, ecosystem health issues often are more manageable and more economically addressed with a preventive approach. While recognizing the importance of habitat restoration, the program stays actively involved in coastal conservation land acquisition, and preservation of intact habitats. With an increasing awareness that "restoration" activities do not always yield appreciable environmental gains (Hildebrand et al. 2005), the program sometimes takes a critical view of likely side effects of some proposed restoration projects.

Future Directions

Among the exciting opportunities that technology offers for the resource management capacity is the ability to better manage data and to do so in georeferenced databases. The TBAP program is actively applying cutting-edge GIS technology to improve the storage and retrieval of information. This present investment of effort is expected to greatly improve the accuracy and availability of data and metadata.

The program must maintain and expand academic ties to take advantage of the growing knowledge base relevant to the preserves. The model of engagement of the scientific community that has yielded actionable information in recent years must be applied to new academic partnerships and emerging environmental issues. New means of acquiring, storing and retrieving scientific literature are improving the program's integration of research results into everyday actions and policy advice. The prompt availability of new scientifically-derived information to policy makers is an important role of the program.

The program actively engages the public on each issue, but large-scale awareness with limited program resources remains challenging. New opportunities afforded by the Internet and other media must be aggressively pursued to enhance appreciation of the importance of ecosystem services and the overall value of healthy ecosystems to the quality of life that Florida's residents and visitors enjoy. The primary focus of aquatic preserves on ecosystem and habitat conservation often may not be as tangible as that of primarily-recreational programs, but public buy-in for these "behind the scenes" efforts is vital.

As the world gets smaller, in a practical sense, so does the world's environment. Many issues important to Florida and Tampa Bay are directly relevant to similar issues in other areas, and all are affected by the health of the world's oceans. In the course of providing advice and perspectives on coastal conservation to other states and nations, it has become apparent that these interactions also yield important knowledge and insights into local efforts. New avenues of information flow now make it easier for resource managers to learn from a global community of research and monitoring programs, rather than "reinventing the wheel". TBAP routinely draws on the scientific analysis of conservation and restoration strategies in distant places in the course of developing resource management actions and defining informational needs.

Given the scale of water quality issues, migratory species and other "big picture" considerations, it is increasingly important that individual MPAs not be managed in isolation. Global networking of MPAs has been called for by the 2002 World Summit on Sustainable Development, the 2003 G8 group of nations, the 2004 Convention on Biological Diversity and the 2003 5th World Parks Congress (IUCN World Commission on Protected Areas 2008). While Florida and Tampa Bay had an early start on MPA networking, there is much to be learned from present efforts to create

new networks based on present-day ecological knowledge. Efforts like the Gulf of Mexico Alliance afford new frameworks for MPA networking, and TBAP and CAMA staff have worked with other nations like Mexico, Britain and China in their MPA networking initiatives.

In the course of the past 40 years, Tampa Bay's Aquatic Preserves have evolved into an adaptive, science-based, place-based program that is well-suited to provide the extra protection for Florida's aquatic habitats that was originally envisioned in the 1960s. New technology and scientifically-derived knowledge offer unprecedented potential for better decision making and proactive resource protection, but the key to future maintenance of our natural heritage especially depends on the appreciation of the public for the ecosystem services provided by these remarkable areas.

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ABSTRACT
Session 11: General Call

PRESENTATION

Stormwater Re-use: City of Tampa Experience. C. Walter (City of Tampa). The City of Tampa beneficially uses 38% of the stormwater generated City-wide. The City of Tampa is 109 square miles. The population of 325,500 makes it the 55th largest city in America and the third largest in the State of Florida. The potable water customer base is approximately 500,000 as there are significantly more daytime visitors than full time residents and the service area for the potable water system is significantly larger than the City limits.

The City of Tampa is a highly urbanized area. It grades less urban and more suburban from its midpoint to the north and south respectfully. The northern 1/3 of the City drains to a City reservoir. Further, large portions of urbanized City of Temple Terrace and development in unincorporated Hillsborough, Pasco and Polk Counties all drain to the reservoir.

The water in this reservoir system is treated at the David Tippin water treatment plant and is distributed as potable water to the City's potable water customer base. Runoff calculations from the entire City of Tampa considering normal rainfall conditions are 1.5 billion gallons per year. Tampa water customers consume 2.0 billion gallons per year. A detailed water budget will demonstrate the economic use of stormwater for potable water demand in the City of Tampa. Plans to maximize performance of the stormwater catchments systems will also be discussed. [chuck.walter@ci.tampa.fl.us]

SCALING THE EFFECTS OF CLIMATE CHANGE ON FLORIDA'S OCEAN AND COASTAL RESOURCES TO THE TAMPA BAY ESTUARINE ECOSYSTEM

E.D. Estevez

ABSTRACT

In 2009 the Florida Oceans and Coastal Council (FOCC) released a report examining major mechanisms of climate change pertinent to Florida, and their effects on the state's ocean and coastal resources. Mechanisms include increases of greenhouse gases, air temperature and water vapor, water temperature, and sea level. Seventeen categories of effects produced by the mechanisms were identified. Published information that was directly concerned with or transferable to Florida was used, and then categorized under each mechanism and effect as what is known, what is probable (highly likely to occur), or what is possible (may occur). This paper provides a summary of the report and offers a perspective on the challenges that scaling and/or transferring climate effects to the Tampa Bay area will pose for scientists and resource managers. FOCC recommendations for embedding climate change in ongoing coastal research projects, from the Council's 2010-11 Research Report, are also summarized. The reports are available online at <http://www.floridaoceanscouncil.org/>.

The FOCC climate report's treatment of sea-level rise is used to illustrate the problems and opportunities scientists and resource managers face in transferring information across dimensions of space and uncertainty. Florida information on sea-level rise and its effects transfer generally to Tampa Bay but more must be learned for a complete understanding of sea-level rise and effects in the estuary. On balance, some data gaps for Tampa Bay also apply at Florida scales. Attention is drawn to the difference between effects associated with sea-level stand, and effects of rates of sea-level rise. Using the FOCC report as a guide, some specific research questions concerning sea level and Tampa Bay are suggested.

INTRODUCTION

The Florida Oceans and Coastal Council (FOCC, Council) was created by the Florida Legislature in 2005 by the Oceans and Coastal Resources Act (S. 161.7 Florida Statutes). The Council is charged with developing priorities for ocean and coastal research and establishing a statewide ocean research plan that is submitted to the Legislature each year. The Council also undertakes projects and programs deemed important for marine and coastal science and resource management, for example, by sponsoring a contemporary evaluation of Florida's ocean and coastal economy (Kildow 2008). In 2009 the Council presented a report, "The Effects of Climate Change on Florida's Ocean and Coastal Resources", to the Florida Energy and Climate Commission and people of Florida. The report was written by Council members with contributions by subject matter experts, and peer-reviewed. The report, revised in June 2009, is available from the Florida Department of Environmental Protection and the Council's website (FOCC 2009). Council intends for the report to be updated as new information becomes available. This paper describes interesting aspects of the report and comments upon its bearing on research and management challenges and opportunities relative to the Tampa Bay estuary and adjoining watershed and coastal environments.

The report identifies 4 mechanisms or drivers of climate change and 12 categories of effects pertaining to Florida's ocean and coastal resources (Table 1). Thirty statements were made about the drivers and 96 statements concerned effects. Effects statements were grouped by certainty

into three categories: what is currently known, what is probable, and what is possible. According to FOCC (2009), “‘Probable’ means that an effect is highly likely to occur, while ‘possible’ means that it may occur, but that predicted impacts must be carefully qualified to reflect the level of certainty” (page 3). The report’s treatment of sea level rise illustrates the organization:

“What We Know. (1) Around Florida, relative sea level has been rising at a slow but constant rate, about an inch or less per decade. (2) The rate of global sea level rise increased from the 19th to the 20th century.

What is Probable. (3) The rate of global average sea-level rise has increased during the late 20th century. (4) In time, the rate of global sea-level rise will accelerate because of ocean warming and contributions from land-based ice melt from glaciers and the ice sheets of Greenland and Antarctica. (5) Sea-level rise will continue well after 2100 even if greenhouse gas concentrations are stabilized by then.

What is Possible. (6) Major inputs of water from the melting of high-latitude and high-altitude ice reservoirs could cause several meters of sea-level rise over the period of centuries”.

Domain of Climate Change Effects Pertinent to Tampa Bay—Most of the drivers and effects discussed in “The Effects of Climate Change on Florida’s Ocean and Coastal Resources” are directly pertinent to Tampa Bay, its watershed, and surrounding environment. Two effects, increases in ocean acidification, and increases in coral bleaching and disease, may be less immediate concerns owing to the estuary’s chemistry and biogeography. Hermatypic, scleractinian coral species occur in natural patch reefs, at depths ≥ 15 m in the Gulf of Mexico near Tampa Bay (Rice et al. 1992) and at least one species of *Sideastrea* occurs in hard bottom communities of Lower Tampa Bay (Savercool 1994). Their abundances in Tampa Bay proper could increase with favorable conditions, and their susceptibility to acidification could likewise increase, but too little is known at present of their vulnerability to acidification or disease to allow an assessment. By the same token, sensitivity to lower pH of oysters in other estuaries (Miller et al. 2009) may not be transferable to Tampa Bay without more information on the details of the bay’s carbonate chemistry.

Effects on coastal water supplies are not as urgent for the Tampa Bay area, as they could become elsewhere in Florida. Although changes in the frequency and intensity of rainfall would affect any region’s water budget, a rise in sea level would have less of an effect on existing water supplies available to the Tampa Bay area than it would in southeast Florida. In the Tampa Bay region, water is provided by instream and offstream reservoirs generally unaffected by sea level. The instream reservoir on the Braden River, Manatee County, is impounded by a low dam (DelCharco et al. 1997) that is at greatest risk of over-topping by rising sea level. In southeast Florida the Biscayne Aquifer is an important water supply, and that aquifer would be endangered by saltwater intrusion caused by sea-level rise (STC 2008). Other effects identified by the 2009 report have greater bearing on Tampa Bay, as illustrated by the case of sea-level rise.

Tampa Bay and Sea-Level Rise—Global or eustatic sea level refers to the average vertical position of the world ocean. Mostly land-based records show that eustatic sea level rose faster in the 20th century than earlier in the historic record. Satellite-based data indicated an acceleration of sea-level rise in the decade of the 1990s (USACE 2009). Trends in eustatic sea level cannot be transferred directly to Tampa Bay, or other Florida waters. The gage record for sea levels around Florida (NOAA 2010a) describes a range in rates of sea-level rise of 1.80 to 2.78 mm/yr (mean of 2.1 mm/yr), somewhat higher than the eustatic mean of 1.7 mm/y for the 20th Century (IPCC 2007). For St. Petersburg, the mean rate of sea-level rise over a 59 year period has been 2.36 mm/y and there is no evidence of acceleration in the rate of sea-level rise around Florida, including at St. Petersburg, during the 1990s (NOAA 2010b).

Looking forward, the IPCC called for a eustatic sea-level rise of approximately 20 to 60 cm by 2100 (IPCC 2007), but recognized that their prediction did not account for recent information on the contribution of glaciers. Subsequent work estimates glacial contributions to cause an overall eustatic sea-level rise on the order of 50 to 140 cm by 2100 (Rahmstorf 2007). Other work suggests greater rates (Velicogna 2006).

How much sea level rises by 2100 or later is a critical matter for Tampa Bay's natural resources, especially in the shallow subtidal to supra-littoral zones. The extent to which sea-level rise affects infrastructure will also depend on total inundation, but another process is at work that must be reckoned for both natural and cultural resources, namely, the rate at which sea level rises. Existing data and forecasts present a potential problem relative to rate issues because the historic rate of sea-level rise in Tampa Bay (at St. Petersburg) has been stable, linear, and equal to a rise of about 24 cm by 2100. If projected stands of sea level in 2100 do occur, the longer Tampa Bay rates remain low and linear, the greater an acceleration must be in the balance of this century once the acceleration begins.

A simple example illustrates the rate problem. Allowing a continued rate of about 2.4 mm/y of sea-level rise in Tampa Bay, a projected sea-level rise of 1.0 m by 2100, and linear terms: in 2025, the rate of sea-level rise would have to accelerate from 2.4 mm/y to 13 mm/y; or in 2050, the rate of sea-level rise would have to accelerate from 2.4 mm/y to 18 mm/y; or if the acceleration only began in 2075, the rate of sea-level rise would have to accelerate from 2.4 mm/y to 33 mm/y. Curvilinear solutions result in even greater rates of acceleration. On the one hand, then, recent sea-level rise in Tampa Bay has been relatively low and stable but, on the other hand, sea levels projected for 2100 have been increasing. This situation probably means that acceleration in the rate of sea-level rise must occur in the coming decades and bring significant adverse impacts to bay resources and infrastructure, once it begins. Or, it could mean that sea level in Tampa Bay may not track eustatic sea level with fidelity in the future. Eustatic sea level is a global mean value and Tampa Bay and Florida waters generally may be situated in a region of the world where local sea-level rise naturally differs from global averages.

Effects and Adaptation—The 2009 FOCC report on climate change enumerated effects of sea-level rise on beaches, coastal wetlands, and bays and estuaries. These effects may be taken as a first-order estimate of effects likely to occur in Tampa Bay, at least with respect to inundation (a 2009 report on the Charlotte Harbor region's vulnerability to climate change including sea-level rise may be consulted for a more detailed resource assessment (SWFRPC 2009). The 2009 report acknowledged that the rate of sea-level rise can be as or more important than its eventual stand but the report did not identify effects related to rates except to state as probable that, "Many tidal wetlands in areas with low freshwater and sediment supplies will 'drown' where sea-level rise outpaces their ability to accrete vertically" (p. 27). The response capacity of natural systems thus sets upper limits to the resource management ideas of mitigation or adaptation, which are only practical at low (to moderate?) rates of sea-level rise; by definition, mitigation and adaptation will not be practical under catastrophic rates of sea-level rise.

The time-course of future sea-level rise is a critical factor to learn in order for adaptation strategies to be effective in Tampa Bay. Management of the bay's natural resources especially will depend on knowing whether future sea level and rates of sea-level rise will be comparable to or significantly greater than historic rates; whether the changes will be continuous or abrupt (CCSP 2008), and how the effects of sea level will interact with other effects of general climate change or local anthropogenic forces. In this regard, it would be useful to revisit the question of whether a sea level highstand of about 0.5 m occurred in the Gulf of Mexico and southeast

Atlantic Ocean during the late Holocene (Froede 2002). Diverse evidence (Stapor et al. 1991; Walker et al. 1995) has been interpreted to support such a high stand lasting from about 1750 to 1450 years before present. Such a temporary high stand would represent an “abrupt change” of the kind foreseen as a consequence of climate change (CCSP 2008), and its evolution and effects in Tampa Bay would be instructive in terms of understanding future sea level accelerations described in Section 1.2. Other research recommendations follow.

FUTURE RESEARCH

Climate Recommendations of the Florida Oceans and Coastal Council—The Council’s 2009 report informed Council’s 2010-11 Research Plan (FOCC 2010) by identifying knowledge gaps at broad and specific levels: “Providing guidance to minimize effects of climate change on Florida’s population and natural resources will entail investigation into these key areas:

Scaling: Methods to apply global, hemispheric, and continental data and models to Florida and its adjoining oceanic environments, for example by modeling sea-level rise based on Intergovernmental Panel on Climate Change (IPCC) scenarios. Emphasis is on collaborative, statewide efforts with peer review.

Baselines: Programs to define existing conditions and eventually status and trends for the major categories of effects identified in the 2009 Council report, for example by monitoring and mapping the effects of climate change on coral reef communities.

Processes: Defining cause-and-effect relationships between climate drivers or effects and the explicit responses of coastal and ocean ecosystems, for example by developing and testing models that assess the impact on fisheries productivity from changes in Florida’s estuarine habitats due to climate change.

Risk: Predicting changes in the sustainability of natural resources and their values or beneficial uses for humans, using robust and understandable methods, for example by adopting standard statewide projections of beach endangerment by climate change.”

Application to Tampa Bay—The FOCC recommendations deserve to be evaluated with respect to the many avenues of Tampa Bay research bearing on climate change. Key areas identified by the FOCC can be illustrated with respect to sea-level rise in Tampa Bay.

Scaling: From Section 1.3, two areas deserving attention include a (1) better understanding of how Florida and Tampa Bay sea levels are affected by the Yucatan, Loop, and Florida currents, and the Gulf Stream, under different regimes of sea-surface temperature and climate, and (2) projections of seasonal steric effects (NOAA, 2010b) on sea level under scenarios of shelf warming.

Baselines: How coherent are sea levels around the bay? Has sea-level rise affected tidal characteristics in Tampa Bay and how might future sea levels affect tides? Have past anthropogenic changes in bay geometry affected tidal ranges and harmonics and how might tides be affected by future changes?

Processes: Two central questions arise. The first concerns the ability of Tampa Bay’s tidal wetlands (and oyster reefs) to track sea level. We need to learn how wetlands in different bay environments maintain their elevations. Conversely, more mechanistic explanations are needed to understand whether parts of Tampa Bay have deepened or will deepen as sea level rises, as adumbrated in a Sarasota Bay study by Tomasko (these proceedings).

Risk: The endangerment of barrier and bay islands, gulf and bay beaches, inlets, and bay causeways should be assessed using physical, social, and economic models, ideally for multiple rates of sea-level rise as well as projected stands in 2100. Vulnerability of the Braden River reservoir to salt-water intrusion, and probable tidal over-topping should also be considered.

SUMMARY AND CONCLUSIONS

The 2009 FOCC report on climate change was based in part on the transfer of knowledge from planet, ocean, and continent-scaled information. This Tampa Bay assessment extends the sea level content of the state report to a finer scale and necessarily involved greater uncertainty, but there may be merit in adopting the state report's method of acknowledging levels of certainty in future discussions of sea level and climate generally, as more local insights develop. The issue of sea-level rise was examined in the context of Tampa Bay. Historic sea-level rise has been relatively slow and linear. Some resources will be affected by simple inundation, but the rate at which sea level rises will also be important to understand. The longer local rate acceleration is retarded, the greater the rates must have to be if present-day projections of sea level in 2100 are realized. Thus the time-course of future sea-level rise, and its affect on bay geometry, shorelines, natural resources, and human infrastructure, are critical factors to discover. Preliminary recommendations for some sea-level research in Tampa Bay are offered with respect to scaling, baselines, processes, and risk. The same considerations could inform research needs concerning other aspects of climate change in Tampa Bay and environs.

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Table 1. Climate change drivers and effects pertaining to Florida's ocean and coastal resources, adapted from FOCC 2006.

Drivers: Increases of...	Effects
Greenhouse gases	Increases in ocean acidification
Air temperature and water vapor	- Altered rainfall and runoff patterns - Altered frequency and intensity of tropical storms and hurricanes
Ocean temperature	- Increases in coral bleaching and disease - Increases in fish diseases, sponge die-offs and loss of marine life - Changes in the distribution of native and exotic species - Changes in nutrient supply, recycling and food webs - Harmful algal blooms - Hypoxia
Sea level	- Changes in estuaries, tidal wetlands and tidal rivers - Changes in beaches, barrier islands and inlets - Reduced coastal water supplies

POTENTIAL IMPACTS OF SEA LEVEL RISE ON SARASOTA BAY SEAGRASSES

D.A. Tomasko & E.H. Keenan

ABSTRACT

In Sarasota Bay, correlations between levels of chlorophyll-a and water clarity allow for the development of chlorophyll-a targets to protect seagrass as a bio-indicator of system health. However, questions have arisen as to the appropriateness of seagrass maps in determining the depth to which seagrass beds can penetrate. Based on a comparison of techniques, it was found that surveyed offshore edges of seagrass meadows are typically 17 cm deeper than offshore edges delimited from seagrass maps produced via aerial photography and photointerpretation. These surveyed edges typically extend 30 to 40 meters farther offshore than edges delimited from seagrass maps. Based on a comparison between predicted depths from a 1950s-era bathymetric map and direct depth measurements, it appears that the waters of Sarasota Bay are approximately 17 cm deeper in 2007 than was the case in the 1950s, which is consistent with estimates of the degree of sea level rise (SLR) expected over a 50-year timeline (i.e. 1950s bathymetric layer to 2007 measured depths). If future SLR occurs at a similar rate, water clarity would have to improve, to allow the same light level to penetrate to the offshore edges of seagrass meadows growing in areas that would be expected to be deeper due to SLR. Bay-wide, chlorophyll-a levels would have to decrease by 13 to 35% by the year 2020 to allow seagrasses to maintain their present-day coverage in the future, deeper, waters of Sarasota Bay.

INTRODUCTION

In Southwest Florida, a substantial amount of research has focused on the relationships between population growth, changing 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 (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 (Johansson 1991; Johansson and Ries 1997; Lewis et al. 1998). Improvements in water quality have in turn been attributed mostly to improvements in the treatment and disposal of wastewater discharges by the cities of Tampa, St. Petersburg and Clearwater since the mid to late 1980s (Johansson and Greening 1999).

A similar situation exists in Sarasota Bay, where recent increases in seagrass coverage were related to reductions in anthropogenic nitrogen loads to the bay by the City of Sarasota and Manatee and Sarasota counties (Kurz et al. 1999).

Nitrogen loads into Tampa Bay and Sarasota Bay (generated primarily by domestic wastewater treatment facilities) were reduced by 57% and 46%, respectively, during the 1980s and 1990s (Tomasko et al. 2005). In response, both Tampa Bay and Sarasota Bay have lower phytoplankton concentrations, greater water clarity and more extensive seagrass coverage in recent years than in the early 1980s (Johansson and Greening 1999; Kurz et al. 1999; Tomasko et al. 2005). As there is no evidence of a concurrent and explanatory trend in rainfall during this same time period, it was concluded that variation in rainfall could not account for the observed increase in seagrass coverage in these two bays (Tomasko et al. 2005).

While chlorophyll-a can be a useful indicator of impaired conditions in an estuary, interpreting the causes and effects of varying chlorophyll-a concentrations is challenging. Chlorophyll-a is known to be highly variable and responsive to a variety of factors, including nutrient loads,

rainfall, salinity, residence time, season, and even the time of day. In addition, factors other than chlorophyll, such as turbidity and color, may also affect water transparency. If those factors affect water clarity to such a degree that they may also affect phytoplankton growth rates, then issues other than nutrient availability alone may need to be accounted for to develop potential nutrient limits for water quality protection and/or restoration.

For Sarasota Bay, there appear to be two main courses of action that could be pursued, as relates to the issue of potentially biologically relevant values of chlorophyll-a, as summarized below.

Approach 1 - Acceptance of existing standards for chlorophyll-a—This approach would be to accept the state's determination that a chlorophyll-a concentration of 11 µg/liter, and the water clarity that accompanies such conditions, is an appropriate target. Or, that a 50% increase in levels of chlorophyll-a over "historical" levels is an indication of an impaired condition.

Approach 2 - Establish site-specific and resource-linked targets for chlorophyll-a—A second approach would be to develop site-specific and scientifically-defensible water clarity and chlorophyll-a goals that could be linked to a living resource. An approach such as this, using science-based management to protect the health of living resources, is similar to that currently being used by the Tampa Bay Estuary Program.

This paper summarizes efforts conducted in Sarasota Bay related to Approach 2, the development of site-specific and resource-linked targets for chlorophyll-a. Additionally, this paper addresses the issue of whether or not "offshore" or "deep" edges from seagrass mapping efforts are the same as a "deep" edge that would be found from site visits and careful depth measurements. Finally, the potentially important issue of sea level rise, as it relates to the development of resource-based water clarity and chlorophyll-a targets for Sarasota Bay is addressed.

MATERIALS AND METHODS

The techniques used in this paper involved the following steps: 1) identification of 'mapped' deep edges of seagrass meadows, 2) determining the 'surveyed' deep edges of seagrass meadows, 3) assessing the percent coverage of seagrasses at these two depths, 4) determining the depths at these two potentially different locations, 5) determining the light levels at these depths, and 6) determining the levels of chlorophyll-a that would allow for adequate light penetration to target depths, and 7) determining the implications of sea level rise on potential chlorophyll-a targets. For this effort, the original Sarasota Bay segmentation scheme used in the SBNEP's Framework for Action (SBNEP 1992) and the Comprehensive Conservation and Management Plan (SBNEP 1995) was utilized. Table 2 contains information on the relationship between portions of Sarasota Bay and the corresponding SBNEP bay segment numbers used in this paper.

Site Identification

Segments 7, 8, 10 and 11 (Sarasota Bay)—PBS&J, working in concert with Janicki Environmental, verified seagrass coverage and water depth information along the mapped deep-edge of the seagrass beds within Sarasota Bay. As such, Janicki Environmental created a "Universe" of sampling stations along the deep-edge of the consistently mapped seagrass beds performed by the Southwest Florida Water Management District (SWFWMD). Twenty randomly selected stations were provided by Janicki Environmental for each of the segments in Sarasota Bay (bay segments 7, 8, 10 and 11). PBS&J randomly selected eight sites for field verification of both seagrass coverage and water depth in each segment (Figure 1). The sites were visited on two calm weather days, February 20th and 21st, 2007. All sites underwent QA/QC review for site accuracy; four sites were removed from analysis based on spatial inconsistencies with map layers.

Segments 1, 3, 13, 14 and 16—PBS&J identified 7-9 mapped deep-edge sampling points based upon the 2006 SWFMWD seagrass coverage maps. Four sites were randomly selected to verify both seagrass coverage and water depth in bay segments 13, 14 and 16 (Figure 1). These data were collected on May 4, 2007 during calm weather conditions. Five sites were randomly selected to verify both seagrass coverage and water depth in segments 1 and 3 (Figure 1). These data were collected on May 14, 2007 during calm weather conditions.

Seagrass Coverage—At all mapped deep-edge sites, a modified Braun-Blanquet method was used to determine seagrass coverage. Three assessments of seagrass coverage and species diversity were recorded at each site using a 1m² quadrat. Each site was evaluated to determine if it satisfied the Virnstein et al. 2000 method for classification of the deep-edge of a seagrass bed; <10% seagrass coverage (Virnstein et al. 2000). If it was determined that the site had >10% seagrass coverage, divers continued to swim offshore until they eventually located the deep-edge, as described above. The location and water depth of the surveyed deep-edge were recorded.

Depth Measurements—Water depth information was collected at a total of 89 sampling sites throughout all bay segments. A total of 39 depth measurements were recorded at the mapped deep-edge sites and an additional 50 measurements were made at the surveyed deep-edge sites. Eleven of the 50 surveyed deep-edge sites were consistent with the location of the mapped deep-edge. Three depths were recorded at each site using a customized staff gauge with a flat bottom disk designed to reduce errors associated with field measurements. All data were collected during calm bay conditions. The GPS coordinates were recorded for each station using a WAAS (Wide Area Augmentation System) enabled Garmin GPSmap 60CSx.

Depth data were corrected for tidal stage and referenced to Mean Low Water (MLW; Table 2). The 6-minute verified water level data from the St. Petersburg, FL (8726520) tide station was used to model the tide for the Cortez, Sarasota and Venice tide stations (<http://tidesandcurrents.noaa.gov>). The tidal prediction for each station was used to modify the St. Petersburg data. The St. Petersburg tide data were adjusted for high and low tide and a time shift was applied for each station.

NOAA NOS Bathymetry—Bathymetry data was collected in Sarasota Bay by the U.S. Coast and Geodetic Survey between 1953 and 1955 (Figure 2; www.estuarinebathymetry.noaa.gov). The majority of depth soundings were completed in 1954. Six surveys were completed collecting approximately 52,299 depth values using a portable depth recorder. The average distance between readings was 51 meters. The vertical datum was referenced to Mean Low Water. The horizontal datum was referenced to NAD83. This data set provided bathymetry information for all Segments listed in Table 2.

Depth Comparison—Using GIS, the closest NOS water depth point to each field site was selected. The 1950s NOS and 2007 surveyed water depths of all 89 sites were statistically compared using a non-parametric two-sample Mann-Whitney (Wilcoxon) test. Additionally, the water depth at the 2007 mapped deep-edge and the 2007 surveyed deep-edge sites were compared using the Mann-Whitney (Wilcoxon) test, and the distance between the mapped deep-edge and surveyed deep-edge was determined.

Chlorophyll-a and Light Attenuation Data—Chlorophyll-a and the light attenuation coefficient (K) data for each bay segment were downloaded from the Florida STORET database (www.dep.state.fl.us/water/storet). These data were available for all segments during the period of August 2001 to February 2006. Light attenuation coefficients (K/meter) in Segment 1 and 3

were calculated from the raw underwater incident light readings where $z=0.5\text{m}$ (personal communication, Greg Blanchard). Additionally, only uncorrected chlorophyll-a values were available for Segment 1 and 3. Corrected chlorophyll-a was available for all other segments. Statistical significance was determined based on the r^2 value and sample size (Scheffler 1979).

The average K was determined for each segment based on the monthly averages from August 2001 to February 2006. Percent sub-surface irradiance was calculated for each location at four corresponding depths (Mapped Deep-Edge (1950s), Mapped Deep-Edge (2007); Surveyed Deep-Edge (1950s) and Surveyed Deep-Edge (2007); Formula 1), and then the average percent sub-surface irradiance (or PAR -photosynthetically active radiation) for each segment at each depth was determined.

Formula 1: Percent PAR = $+ \exp(-k \cdot z) \cdot 100$

Where k =average light attenuation coefficient and z =water depth in meters at MLW

Chlorophyll-a Target—The chlorophyll-a target for segments 7, 8, 10, 11, 13, 14 and 16 were determined based on the average percent sub-surface irradiance at the Surveyed Deep-Edge (2007). The light attenuation coefficient was calculated based on PAR and the water depth at which light readings were made (Formula 2).

Formula 2: $K = -1/z \cdot \ln(I_z / I_o)$

Where I_z = PAR at depth z (in meters) and I_o = directly sub-surface PAR

The current target chlorophyll-a concentration was calculated based upon the linear model relating K and chlorophyll-a for each segment (Table 3). A target chlorophyll-a concentration was also calculated based upon anticipated sea-level rise in Sarasota Bay for the years 2020 and 2050 (after Clark 1992). Using the measured rate of sea-level rise between 1954 and 2007, sea-level rise was simulated for the next 13 and 43 years. The modified water levels were used to recalculate the new (lower) K value required to keep the same percent PAR at a location with additional water depths (due to sea level rise) and the revised chlorophyll-a target was calculated based on the new estimate of K.

RESULTS

Water Depth Comparison—Tide-corrected water depth measurements from 89 sites were compared to the water depth estimates for these same locations using the bathymetry data compiled in 1953-1955 by the USGCS. The average ($n = 89$) water depth for sites visited in Sarasota Bay, based on the 1950s bathymetry data, was 1.09 m at MLW compared to an average depth of 1.26 m at MLW water depth for those same locations, calculated from directly measured water depths in 2007 (Figure 3). The change in water depth was 0.17 m which is equivalent to 3.2 cm/decade. There was a statistically significant difference between the 2007 water depths and the 1950s data ($N=89$; $p=0.01$). This difference in water depths is within the range of documented rates of sea level rise in St. Petersburg during 1950 to 1990 (2.4 cm/decade), and estimated rates of future sea level rise of approximately 5.2 cm/decade, as outlined in Clark (1992).

Additionally, the mapped deep-edge (2007) sites were compared to the surveyed deep-edge (2007) sites; the average water depth was 1.16 m at MLW and 1.34 m at MLW, respectively (Figure 4), but this difference was not statistically significant ($N=39$; $p= 0.09$). The average distance between the mapped deep-edge (2007) and the surveyed deep-edge (2007) was 42 m.

Chlorophyll-a Targets

Segments 1 and 3: A significant correlation between chlorophyll-a and light attenuation was not found for Segments 1 and 3, which correspond to Anna Maria Sound and Palma Sola Bay. In these parts of Sarasota Bay, prior studies concluded that non-chlorophyll suspended material (NCSM) was the dominant attenuator of light in the water column (Dixon and Kirkpatrick 1995). On average, NCSM explained 77% of the partitioned non-water attenuation for diffuse light in Segment 1 and 58% in Segment 3. Consequently, a chlorophyll-a target was not calculated for these segments.

Segment 7: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficients were 5.4 µg/l and 0.6681 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 7. Based on the Surveyed Deep-Edge water depth and percent PAR, a chlorophyll-a target of 5.3 µg/l was calculated to maintain deep edge light requirements. A potential IWR target was calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968B (Sarasota Bay; 5.5µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise. A projected water depth of 1.63m in 2020 and 1.76m in 2050 was used. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 4.6 µg/l is proposed for 2020. The proposed target chlorophyll-a concentration for 2050 is 2.6 µg/l.

Segment 8: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 5.4 µg/l and 0.6681 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 8. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 5.0 µg/l was calculated to maintain current deep edge light requirements. A potential IWR target was calculated based upon the chlorophyll-a value which exceeds the historical value by 50% for WBID 1968B (Sarasota Bay; 5.5µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise using projected water depths of 1.61m and 1.74m, respectively. To maintain current water clarity conditions with increased water depth, a chlorophyll-a concentration of 4.3 µg/l is proposed for 2020. The proposed target chlorophyll-a concentration for 2050 is 2.2 µg/l.

Segment 10: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 3.8 µg/l and 0.6372 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 10. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 1.9 µg/l was calculated to maintain current deep edge light requirements. A potential IWR target was calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968B (Sarasota Bay; 5.5µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise, therefore, projected water depths of 1.85m and 1.98m were used, respectively. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 1.2 µg/l is proposed for 2020. Due to further increases in water depth, a target chlorophyll-a concentration could not be derived for Segment 10 for 2050 conditions.

Segment 11: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 4.5 µg/l and 0.6837 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 11. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 6.4 µg/l was calculated to maintain current deep edge light requirements. A potential IWR target was

calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968B (Sarasota Bay; 5.5 µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise using projected water depths of 1.19m and 1.32m, respectively. To maintain current water clarity conditions with increase water depth, a target chlorophyll-a concentration of 3.3 µg/l is proposed for 2020. The proposed chlorophyll-a concentration for 2050 is 1.7 µg/l.

Segment 13: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 7.7 µg/l and 0.9993 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 13. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 3.9 µg/l was calculated to maintain current deep edge light requirements. A TMDL target was developed based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968D (Roberts Bay; 7.2 µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise and projected water depths of 1.26m and 1.39m, respectively. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 5.5 µg/l is proposed for 2020. The proposed chlorophyll-a concentration for 2050 is 3.1 µg/l.

Segment 14: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 7.7 µg/l and 0.9105 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 14. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 7.7 µg/l was calculated to maintain current deep edge light requirements. A potential IWR target was calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968E (Little Sarasota Bay; 8.6 µg/l). In comparison, the TMDL target derived by the Florida Department of Environmental Protection and adopted for the improvement of water quality in Segment 14 is 8.6 µg/l. The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise and projected water depths of 1.01m and 1.14m, respectively. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 6.6 µg/l is proposed for 2020. The proposed chlorophyll-a concentration for 2050 is 3.8 µg/l.

Segment 16: Based on the data from August 2001 to February 2006, the average chlorophyll-a concentration and light attenuation coefficient were 5.6 µg/l and 0.9105 K/m, respectively (Table 4). The linear model (Table 3) was used to produce target chlorophyll-a values for Segment 16. Based on the Surveyed Deep-Edge water depth and K, a chlorophyll-a target of 4.7 µg/l was calculated to maintain current deep edge light requirements. An IWR target was calculated based upon the chlorophyll-a value which exceeds the historic value by 50% for WBID 1968F (Blackburn Bay; 5.5µg/l). The 2020 and 2050 target chlorophyll-a concentrations were calculated based on anticipated sea-level rise and projected water depths of 0.94 m and 1.08 m, respectively. To maintain current water clarity conditions with increased water depth, a target chlorophyll-a concentration of 4.0 µg/l is proposed for 2020. The proposed chlorophyll-a concentration for 2050 is 2.2 µg/l.

DISCUSSION

The results of this project can be summarized as follows:

- For much of the bay, correlations between chlorophyll-a and water clarity allow for the development of potential water clarity goals or “targets”
- For much of the bay, chlorophyll-a targets could be developed to protect seagrass resources as a “bio-indicator” of system health

- However, chlorophyll-a and water clarity do not appear to be correlated with each other in either Palma Sola Bay or Anna Maria Sound
- Detailed surveying of the deep edge of seagrass meadows results in a multi-year annual average light requirement of approximately 31 to 46% of sub-surface irradiance (at MLW) for Sarasota Bay
- Surveyed seagrass meadow deep edges are typically 17 cm deeper than “deep edges” delimited by aerial photography and mapping techniques
- Surveyed seagrass meadow deep edges typically extend 30 to 40 meters farther offshore than edges delimited by mapping techniques
- Based on a comparison between predicted depths from a 1950s-era bathymetric map and current depth measurements, it appears that the waters of Sarasota Bay are approximately 17 cm deeper now (2007) than was the case in the 1950s
- The rate of sea level rise (SLR) that appears to have been documented in Sarasota Bay is in line with estimates of the degree of SLR expected over a 50-year timeline
- If future SLR occurs at a similar rate, water clarity would have to be improved to allow the same light level to penetrate to the offshore edges of future seagrass meadows at their current locations
- To maintain the same light penetration to future, deeper waters, chlorophyll-a levels would have to decrease by 51 to 56% by the year 2050, to allow seagrasses to maintain their present coverage (Table 5).

It is not surprising that surveyed offshore edges of seagrass meadows, delimited via snorkeling, extend farther offshore and into deep waters than offshore edges delimited via aerial photography and photointerpretation. It would be difficult for aerial imagery, regardless of the technique used, to be sufficient to pick up all the seagrass resources found in any particular area. However, the more relevant implication of this finding is that “edges” found by snorkeling would have deeper depths, and would have different light levels at those depths than locations depicted by a map as being a deep edge. For this reason, it is important that the type of edge (surveyed or mapped) is consistent with the literature, for developing minimum light requirement targets for seagrasses. For example, estimates of minimum light requirements for seagrasses determined in Tampa Bay (Dixon 1999), Sarasota Bay (Dixon and Kirkpatrick 1995), and Lemon Bay (Tomasko et al. 2001) range as high as 42% of immediately sub-surface irradiance. In all three of these locations, the deep edge used for estimating light requirements was defined by field surveying the site, and not by photointerpretation of aerial photography and mapping techniques. The choice of what target light level to use for maintaining seagrass coverage is an important one for setting water clarity targets, and thus for setting possible chlorophyll-a targets.

An additional consideration for TMDL development in Sarasota Bay is whether or not there is evidence of system-wide improvements or degradation in water quality, based on ambient water quality monitoring programs or other objective assessments. That is, is impairment status derived from the rules and regulations outlined in the Impaired Waters Rule (IWR) consistent with more locally-collected data on trends in appropriate bio-indicators of system health?

In the SBNEP’s document “Framework for Action”, Lowrey et al. (1993) concluded that of the 16 bay-wide segments (plus the nearshore waters of the Gulf of Mexico) in Sarasota Bay, roughly two-thirds of those segments had trends in chlorophyll-a. Of those segments with trends, 93% of them had trends of decreasing values of chlorophyll-a, while only one segment had a trend of increasing values. Likewise, trends for total nitrogen were found in 43% of SBNEP segments, with twice as many having decreasing trends as increasing trends. For total phosphorus, 48% of bay segments had significant trends, with only 10% of those segments showing signs of degradation over time (i.e., increasing values)

For water clarity (measured as Secchi disk depths) 29% of bay segments had trends over time - all of those trends were of improving water clarity (i.e., none showed evidence of degrading water clarity).

In a later assessment of status and trends in water quality, Dixon and Heyl (1999) found that more recent (1989 to 1998) trends of declining concentrations of total nitrogen and total phosphorus (9 of 12 bay segments) were not accompanied by concurrent decreases in phytoplankton abundance, as no bay segments showed trends of decreasing chlorophyll-a concentrations during that same time period. The basis for this potential disconnect between declining nutrient concentrations and further decreases in phytoplankton abundance (between 1989 and 1998) merits further investigation, especially as it relates to the finding (discussed below) of continued increases in seagrass coverage in Sarasota Bay (Kurz et al. 1999; Tomasko et al. 2005).

In the SBNEP's Framework for Action, the technical synthesis chapter (Tomasko et al. 1992) outlined the finding that seagrass biomass and productivity was a better indicator of segment-specific nitrogen loads than either chlorophyll-a or Total Kjeldahl Nitrogen (TKN). A key finding of Tomasko et al. (1996) was that the status and trends of seagrass coverage in Sarasota Bay would be a better indicator of system health than traditional water quality monitoring programs.

As seagrass coverage has been found to be increasing in Sarasota Bay in recent years (Kurz et al. 1999; Tomasko et al. 2005), these results would indicate that findings of impairment for nutrients in various Sarasota Bay segments (e.g., Palma Sola Bay, Roberts Bay, Blackburn Bay) might be more a function of a potentially locally inadequate listing criteria via the IWR than a true indicator of a stressed ecosystem.

However, results of the depth measurements, comparing 1950s bathymetry estimates vs. field survey estimates from 2007 suggest that sea level rise could be an important factor to consider, when developing water quality goals for Sarasota Bay. The average increase in water depth of 17 cm, comparing 1950s depths to 2007 estimates, calculates out to a sea level rise rate of approximately 3.2 cm/decade. This difference in water depths is within the range of rates of measured sea level rise in St. Petersburg during 1950 to 1990 (ca. 2.4 cm/decade), and estimated rates of future sea level rise (ca. 5.2 cm/decade) as outlined in Clark (1992). Should these estimates of future sea level rise prove accurate, chlorophyll-a concentrations in Sarasota Bay would have to be reduced by 51 to 56 percent, to allow for the present-day light levels to penetrate to the same, but deeper, offshore edges of future seagrass meadows in the bay.

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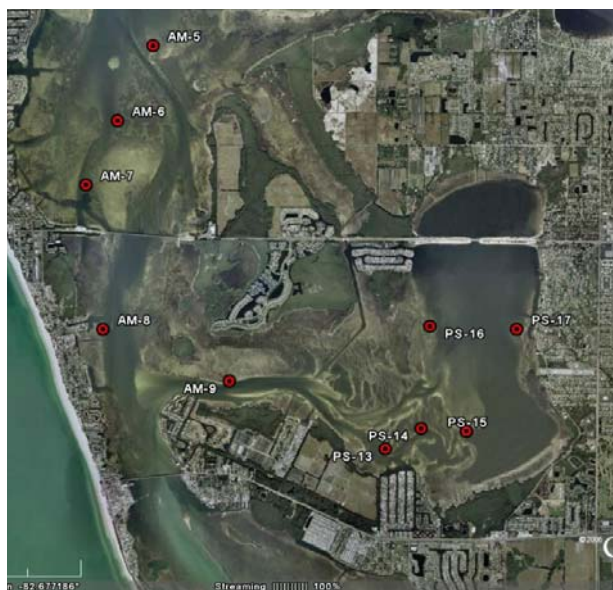
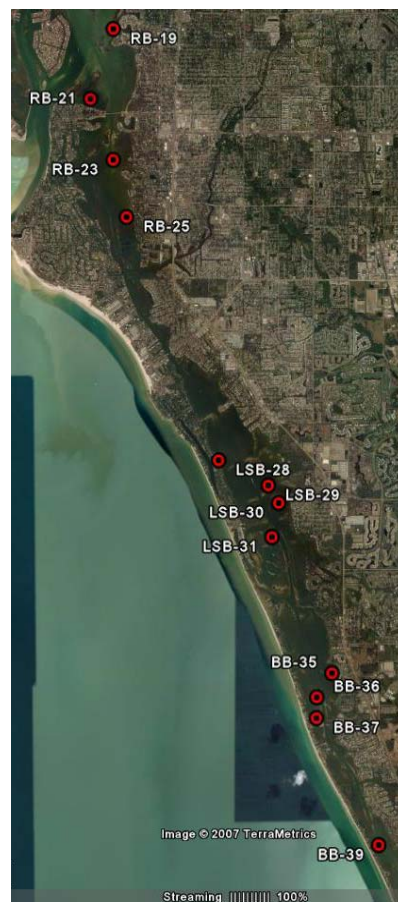
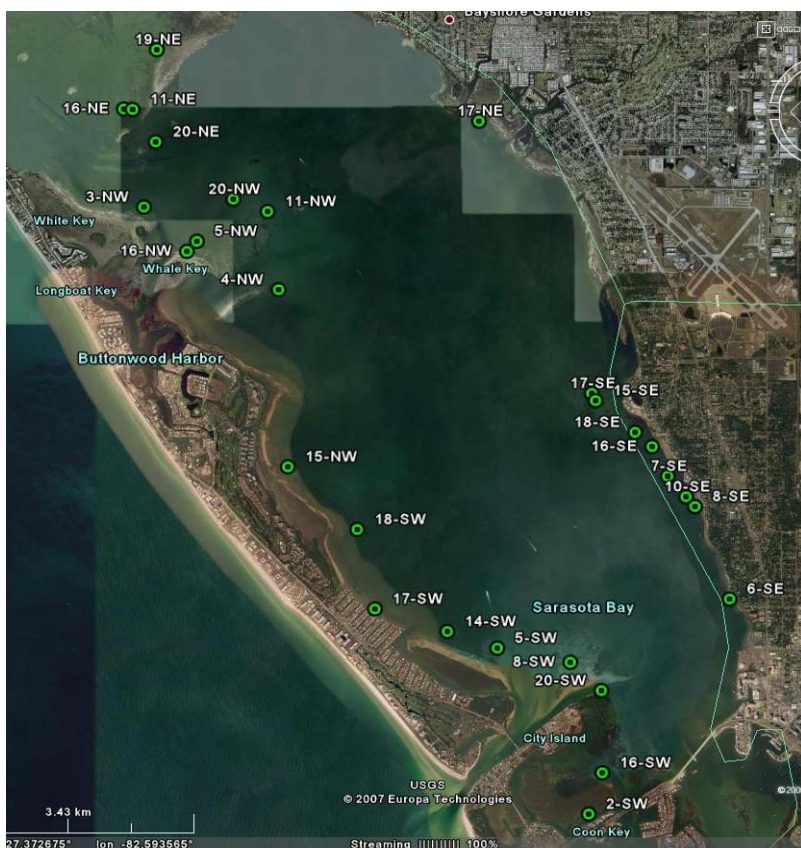


Figure 1. Top left: Site location in Segments 7, 8, 10 and 11. Top right: Site location in Segments 13, 14 and 16. Bottom: Site Location in Segments 1 and 3.

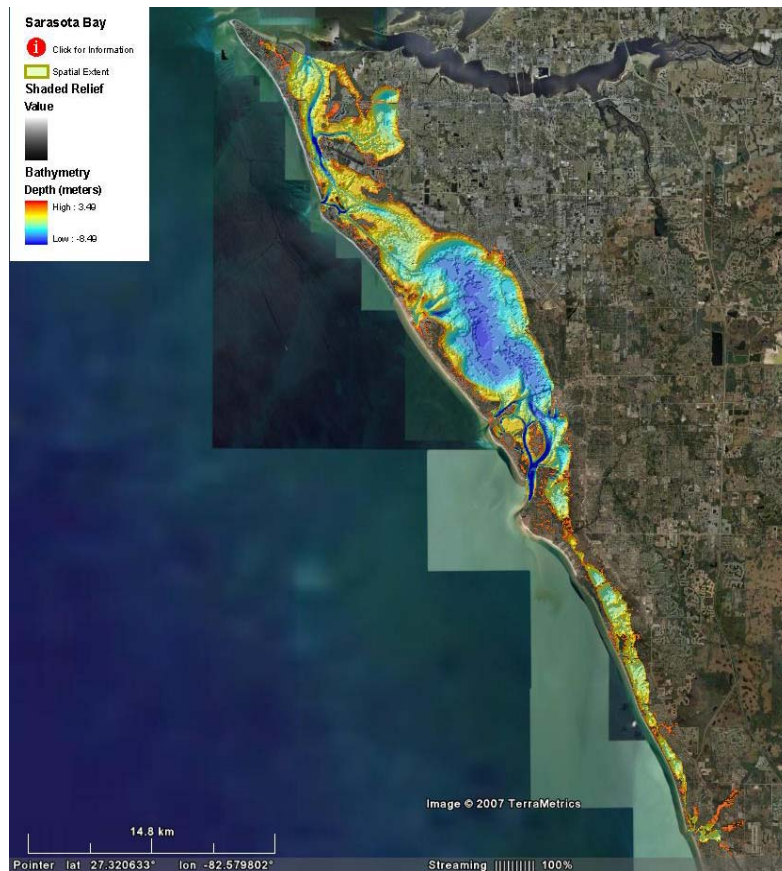


Figure 2. 1950s NOS NOAA bathymetry data for Sarasota Bay.

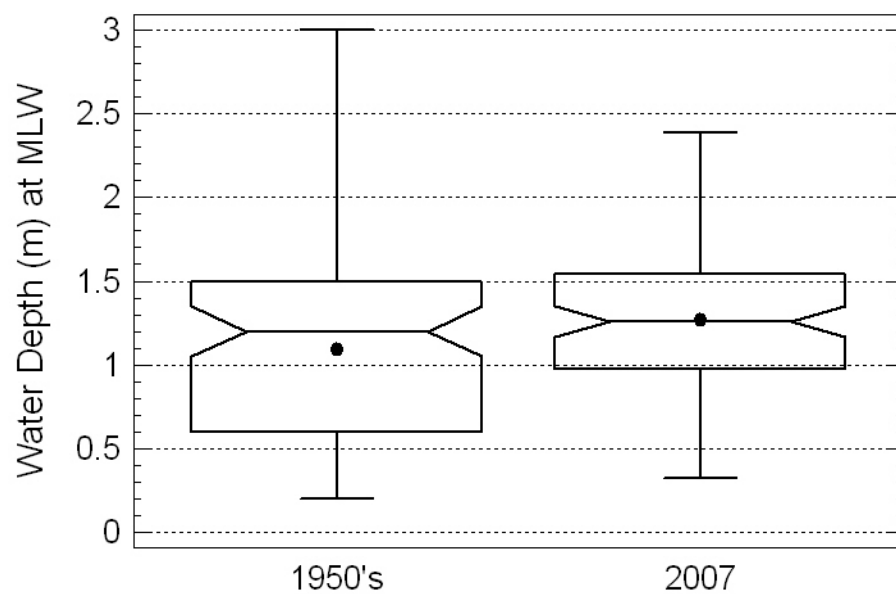


Figure 3. Average Water Depth at the 1950s NOS and 2007 field sites.

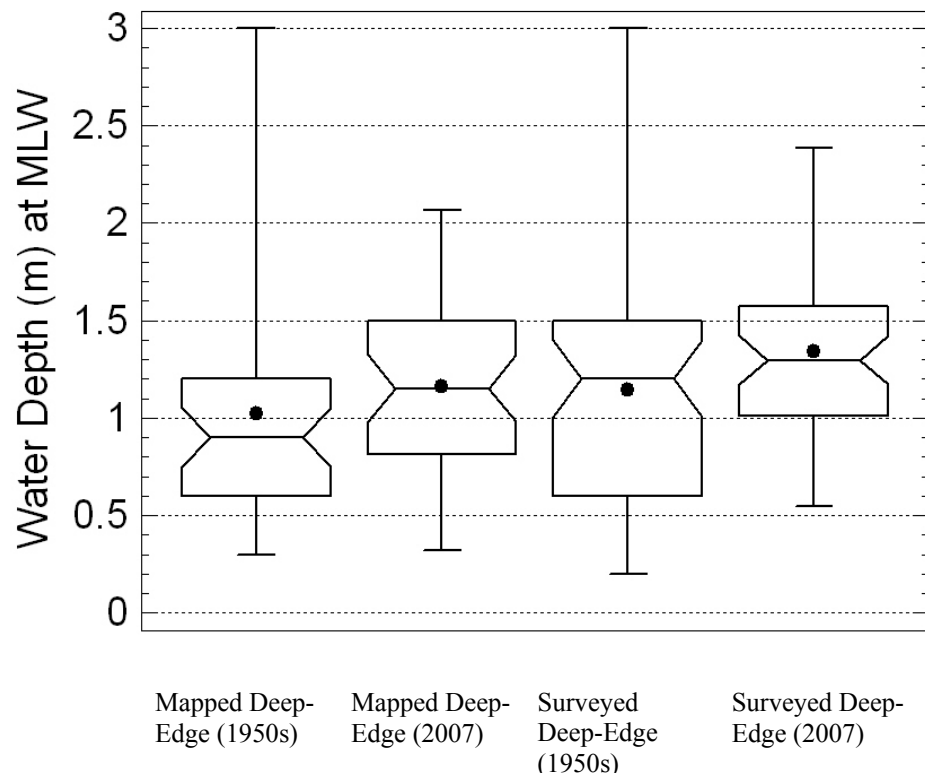


Figure 4. Average water depth using both 1950s bathymetry data and surveyed mapped and deep-edges.

Table 1. Verified Impaired Waterbodies in Sarasota Bay (from FDEP Group 3 Verified Impaired List (6/20/2005)).

Waterbody Name	WBID	SBNEP Segment	Parameter of Concern
Palma Sola Bay	1883	2 and 3	Nutrients (chlorophyll-a > 11 µg / liter)
Palma Sola Bay North	1883 B	3	Bacteria (exceeds DOH thresholds)
Palma Sola South	1883 C	3	Bacteria (exceeds DOH thresholds)
Bowlees Creek	1896	NA	Fecal Coliform Bacteria
Bowlees Creek	1896	NA	Total Coliform Bacteria
Bowlees Creek	1896	NA	Nutrients (chlorophyll-a)
Philippi Creek	1937	NA	Fecal Coliform Bacteria
Philippi Creek	1937	NA	Total Coliform Bacteria
Roberts Bay	1968 D	13	Nutrients (Historic chlorophyll-a)
Blackburn Bay	1968 F	16	Nutrients (Historic chlorophyll-a)
Clark Lake	1971	NA	Nutrients (TSI > 60)
Clowers Creek Tidal	1975 AA	NA	Nutrients (chlorophyll-a)

Table 2. Segment and reference tidal station for field data.

Segment	Bay	Tide Station
1	Anna Maria Sound	Cortez
3	Palma Sola Bay	Cortez
7	NW Sarasota Bay	Cortez
8	NE Sarasota Bay	Cortez
10	SW Sarasota Bay	Sarasota
11	SE Sarasota Bay	Sarasota
13	Roberts Bay	Sarasota
14	Little Sarasota Bay	Sarasota
16	Blackburn Bay	Venice

Table 3. Results of Linear Model Regression between chlorophyll-a and K for each Bay Segment.

Segment No.	Segment Name	Chl A significant?	Linear Model
1	Anna Maria Sound	No	
3	Palma Sola Bay	No	
7	Upper Big SB – West Side	Yes – positive	$K = 0.5342 + 0.0241(\text{chlorophyll-a})$
8	Upper Big SB – East Side	Yes – positive	$K = 0.5408 + 0.0238(\text{chlorophyll-a})$
10	Lower Big SB – West Side	Yes – positive	$K = 0.5628 + 0.0205(\text{chlorophyll-a})$
11	Lower Big SB – East Side	Yes – positive	$K = 0.5025 + 0.0403(\text{chlorophyll-a})$
13	Roberts Bay	Yes – positive	$K = 0.7263 + 0.0364(\text{chlorophyll-a})$
14	Little Sarasota Bay	Yes – positive	$K = 0.7078 + 0.0404(\text{chlorophyll-a})$
16	Blackburn Bay	Yes – positive	$K = 0.592 + 0.0564(\text{chlorophyll-a})$

Table 4. Average Corrected chlorophyll-a, Light Attenuation Coefficient, and PAR for each segment.

Segment	Average Chl-a (μg / liter)	Average Light Attenuation Coefficient (K per m)	Percent PAR at the Surveyed Deep-Edge (2007)
7	5.4	0.6681	35
8	5.4	0.6681	36
10	3.8	0.6372	34
11	4.5	0.6837	47
13	7.7	0.9993	31
14	7.7	1.0180	37
16	5.6	0.9105	46

Table 5. PBSJ Proposed chlorophyll-a targets for all segments and percent reduction required to reach the proposed 2050 target.

Segment	PBSJ Proposed Target	PBSJ 2020 target	PBSJ 2050 target	% decrease (current to 2050)
7	5.3	4.6	2.6	52
8	5.0	4.3	2.2	56
10	1.9	1.2	N/A	N/A
11	3.9	3.3	1.7	56
13	6.4	5.5	3.1	52
14	7.7	6.6	3.8	51
16	4.7	4.0	2.2	53

PREPARING FOR CLIMATE CHANGE AND SEA LEVEL RISE IN TAMPA BAY

H. Greening & E. Sherwood

ABSTRACT

The Tampa Bay Estuary Program, Sarasota Bay Estuary Program, Charlotte Harbor National Estuary Program and the other four Gulf of Mexico NEPs are collaborating on an EPA Climate Ready Estuaries project to develop a Handbook for Gulf Coast coastal communities to assist in incorporating climate change effects into habitat restoration and protection plans. The Handbook would be based on the results from ongoing and upcoming funded projects addressing climate change resiliency in habitat restoration projects in the Gulf NEPs and other relevant case studies. We envision that the Handbook would build on very recent guidelines, but would provide a more specific focus on incorporating resiliency into habitat restoration and protection strategies for Gulf Coast communities using the results from ongoing evaluations in each of our estuaries.

INTRODUCTION AND BACKGROUND

Anthropogenic climate change is now widely regarded as one of the most pressing challenges facing society. Its potential consequences are profound and far-reaching: melting terrestrial and polar ice, rising sea level contributing to coastal flooding and erosion, increased frequency of severe weather, increases in ocean temperature and acidification, and rising incidences of marine diseases and harmful algal blooms that can devastate fisheries (Intergovernmental Panel on Climate Change (IPCC) 2007; Florida Oceans and Coastal Council 2009). With 1,200 miles of coastline and billions of dollars invested in coastal real estate and tourism, a warming climate with higher sea levels places Florida at risk. Higher average sea temperatures and changing precipitation patterns may have dramatic and widespread effects on coastal property and habitats. One possible result is the development of more frequent and intense hurricanes (Elsner 2006) and hurricane-related flooding.

Average air temperatures have risen by about 2°F in parts of Florida since the 1960s, with precipitation decreasing in southern Florida and increasing in central Florida and the Panhandle region (USEPA 1997). By 2100, summer temperatures in Florida could rise an additional 3° - 7°F (Twilley et al. 2001). Warmer temperatures are expected to shift the geographic areas in which freezes occur, enabling subtropical plant species such as mangroves, several of which cannot tolerate freezing temperatures, to expand their ranges northward.

On a statewide basis, Glick and Clough (2006) assessed the potential impacts of sea level rise on coastal habitats and fisheries in nine areas along Florida's coast, including Tampa Bay. The study predicts that many of Florida's shoreline and subtidal habitats will be inundated by 2100 due to sea-level rise from global warming, with potentially grave implications. For the nine sites combined, the report predicts losses of nearly 50 percent of critical salt marsh (22,956 acres) and 84 percent of tidal flat (166,572 acres) habitats during this century. Additionally, dry land would decrease by 14 percent (174,580 acres), and roughly 30 percent (1,000 acres) of ocean beaches and two-thirds (5,879 acres) of estuarine beaches would disappear (Glick and Clough 2006).

In Tampa Bay a 15 in. rise in sea level by 2100 is projected to result in a 96 percent loss of existing tidal flats (42,689 acres) and an 86 percent loss of salt marsh (2,552 acres), with a 10 percent overall loss of dry land (34,676 acres)—an area comparable in size to the city of St. Petersburg. Mangrove extent would more than double under the mean sea-level rise scenario (Glick and Clough 2006). The deeper water would also alter the distribution and composition of seagrasses, which require sunlight to grow. Implications for fisheries, while difficult to forecast,

are likely to be profound. While the prognosis for individual fish species rests on a combination of factors, it is reasonable to deduce that those fish most dependent on vulnerable habitats such as salt marshes are at greatest risk. For Tampa Bay, that would include some of the estuary's most prized game fish – common snook, spotted sea trout, red drum, sheepshead, and tarpon.

Based on assessments of atmospheric and oceanic monitoring data, and information from a suite of global models, the International Panel on Climate Change (IPCC 2007) summarized climate change issues as follows:

- The term “climate change” refers to a change in the state of the climate that can be identified by changes in the mean and/or the variability of its properties, persists for decades or longer, and is caused by natural variability or as a result of human activities;
- Evidence that global climate is warming is unequivocal, based on observations of increases in global average air and ocean temperatures, widespread melting of snow and ice, and rising global average sea-level;
- Since 1750, global atmospheric concentrations of greenhouse gases (such as CO₂, CH₄ and N₂O) have increased substantially as a result of fossil fuel use, deforestation, and other land use changes;
- Most of the observed increase in global average temperatures since the mid-20th century is very likely due to the observed increase in anthropogenic greenhouse gas concentrations;
- Observational evidence from all continents and most oceans shows that many natural systems are being affected by climate change;
- The resilience of many ecosystems is likely to be exceeded during the 21st century by a combination of climate change, associated disturbances (flooding, drought, wildfire, and ocean acidification), and other stressors (land use changes, pollution, fragmentation of natural systems, and over-exploitation of resources).

Using the magnitudes and rates of climate change projected by the IPCC (2007) as a starting point, the Florida Oceans and Coastal Council (FOCC 2009) forecast a range of “probable” and “possible” impacts on Florida's coastal areas that may occur in coming decades. In the Tampa Bay area, the potential habitat-related impacts evaluated by the FOCC (2009) can be grouped into three broad categories based on their underlying causes: those associated with sea-level rise, those associated with increasing air and water temperature, and those associated with increasing acidification of seawater.

Sea-level rise—Global sea-level rose by about 120 m during the several millennia that followed the end of the last ice age (approximately 21,000 years ago). Sea level then stabilized, between 3,000 and 2,000 years ago, at a level that did not change substantially until the mid-19th century (Bindoff et al. 2007). Sea level began rising once again during the 19th century, and during the 20th century it is estimated to have risen at a rate of about 1.7 mm per year. Since 1993 sea-level has been rising at a faster rate — approximately 3 mm per year — and is projected to rise at somewhat higher rates throughout the 21st century (Bindoff et al. 2007). Given a range of plausible greenhouse gas emission scenarios, global simulation models suggest that by 2100 average sea-level may have risen to about 0.6 m above 1980-1990 levels. However, these simulations do not address a number of uncertainties (such as changes in the melting rates of the Greenland and Antarctic ice sheets) that could potentially produce greater increases, and “the upper values of the ranges given are not to be considered upper bounds for sea-level rise” (Bindoff et al. 2007). In addition, sea-level rise is expected to continue for centuries, even if greenhouse gas emissions are stabilized, due to the time scales of climate processes.

The impacts of rising sea-level on habitats in the Tampa Bay area will depend on both the rate and magnitude of the rise. The current rate is about an inch a decade, based upon NOAA tide gauge data (Figure 1). As summarized by the FOCC (2009) and references therein, it appears probable that water depths will continue to increase within the bay's current shoreline, and the shoreline itself will migrate landward in areas where manmade structures such as seawalls and bulkheads are not present to prevent such movement. Depending on the rate of sea-level rise, emergent tidal wetlands and other coastal habitats may be able to persist by accreting vertically, migrating landward, or both. For instance, mangrove forests may be able to keep up with sea-level rise through accretion processes up to a rate of about 25cm of rise per 100 years (Ellison 1994); however, other confounding processes related to climate change may affect their ability to adapt to sea-level rise up to these levels (Gilman et al. 2008). If sea-level rises more rapidly than the biota can respond, however, these adaptive responses may not even be possible. Coastal habitats may also be lost in areas where manmade structures prevent landward migration (Williams et al. 1997; FOCC 2009).

Increasing temperature—Over the next two decades, the IPCC (2007) has projected that global mean air temperature will warm at a rate of about 0.2°C per decade, with the greatest warming occurring over land and at high northern latitudes, and the least occurring over the southern ocean and parts of the North Atlantic Ocean. Sea-surface temperatures have also been increasing in tropical and sub-tropical waters, rising by an average of 0.3°C between the 1950s and 1990 (FOCC 2009). Probable impacts of the ongoing temperature increase that are anticipated to occur during the 21st century include: increased eutrophication (due to enhanced physiological activity by bloom-forming phytoplankton and macroalgae); increased stress, disease, and mortality among corals, sponges, and fishes (due to factors such as temperature stress, more frequent disease outbreaks, more frequent harmful algal blooms, and more frequent hypoxic events); and changes in the geographic distributions of native and introduced species (due to net northward shifts of species in response to changing temperature regimes) (FOCC 2009).

Increasing acidification of coastal waters—Due to the uptake of additional carbon dioxide, the total inorganic carbon concentration of the world's oceans has increased, causing a decrease in the depth at which calcium carbonate dissolves and a decrease in surface ocean pH. On average, pH has fallen (ocean waters have become more acidic) by 0.1 units since 1750. Observations of pH at available monitoring stations for the last 20 years also show trends of decreasing pH at a rate of 0.02 pH units per decade (Bindoff et al. 2007). Because pH is expressed on a logarithmic scale, changes as small as 0.1 pH units can have large impacts on marine organisms such as corals, echinoderms, crustaceans, mollusks, and certain plankton whose shells or cell coverings include calcium carbonate (FOCC 2009). Potential habitat impacts in the Tampa Bay region during the 21st century include variations (upward and then downward) in the productivity of some marine plants as pH declines, reductions in populations of marine organisms that have difficulty tolerating the pH decreases, and increased dissolution of carbonate sediments (FOCC 2009).

Management responses—In addition to long-term reductions in anthropogenic emissions of greenhouse gases, the FOCC (2009) recommended a number of steps that can be taken to respond to climate change. Although some habitat-related impacts may have to be accepted as highly probable, such as loss of coral reefs in southern Florida, others could potentially be mitigated. For example, additional coastal lands may be protected from development, so that tidal wetlands will be able to migrate inland with ongoing sea-level rise. Given the difficult social, political, and environmental tradeoffs that will be necessary when formulating such responses, decision makers will require access to sound technical information and interpretive tools. The FOCC (2009) therefore recommended the following as priority areas for future investigative work:

- Improving the tools available to predict the rate and magnitude of future sea-level rise in Florida, and potential impacts to natural resources, based on existing IPCC emissions scenarios;
- Estimating potential impacts to fisheries productivity due to changes in estuarine habitats caused by climate change; and
- Monitoring, modeling, and mapping the responses of natural systems, such as coral reef communities and other at-risk habitats, in order to improve managers' abilities to predict the future impacts of climate change on these systems.

Ongoing climate change actions and assessments in Tampa Bay—Climatic variability, including the potential for relatively abrupt (e.g., decadal) climate change, also has important implications for coastal habitat restoration projects in which sizeable investments have been made. While there is still significant uncertainty surrounding the timing and magnitude of these changes, habitat restoration teams led by the Southwest Florida Water Management District are now designing and constructing restoration sites in the Tampa Bay region with maximum amounts of high marsh to allow up-slope recruitment as sea level rises. Ongoing planning will require closer consideration of these factors and adaptive management as new information becomes available.

As shown in Figure 1, sea level rise has been ongoing in Tampa Bay at about an inch a decade since records began in the 1940s, and is projected to continue at or above this rate over the next century. The recently-updated Tampa Bay Habitat Masterplan (PBS&J 2009) identified the need to incorporate consideration of climate change into the adopted habitat restoration, protection and acquisition priorities. In 2010 TBEP, in partnership with the six other National Estuary Programs within the Gulf of Mexico and EPA's Climate Ready Estuaries Program will initiate a project to build the concept of resiliency into existing coastal emergent (salt marsh, mangrove, oyster beds) and submerged (seagrass) habitat restoration and protection strategies that consider the effects of climate change.

An important first element of this work will be the completion of a vulnerability assessment for existing habitats under future potential sea level rise and storm surge scenarios. The vulnerability assessment will be used to assist in the identification of habitats most vulnerable to SLR and storm surge, and is expected to provide information to help identify opportunities for acquisition and restoration to allow upslope migration of existing priority vulnerable habitats; and habitats/areas where mitigation activities could slow the loss of habitat or socioeconomic function, and improve upon the important ecosystem services that coastal habitats provide in Tampa Bay.

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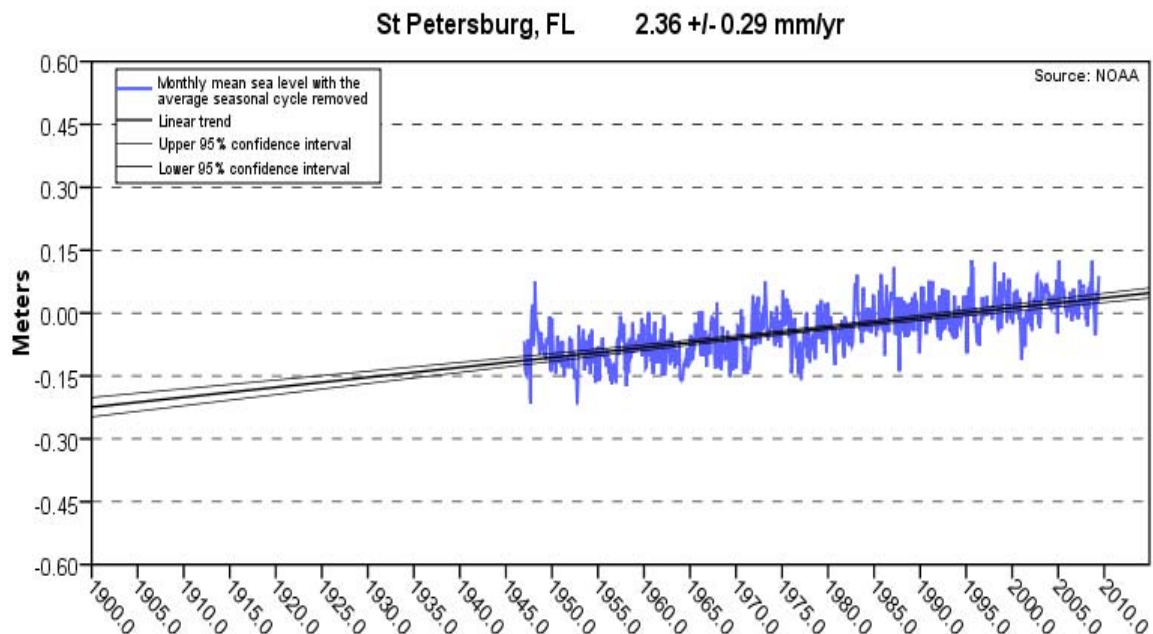


Figure 1. Mean sea level trend for St. Petersburg, Florida, 1948 – 2009. The mean sea level trend is 2.36 mm/yr with a 95% confidence interval of ± 0.29 mm/yr based on monthly mean sea level data from 1947 to 2006, which is equivalent to a change of 0.77 ft in 100 years (from: http://tidesandcurrents.noaa.gov/sltrends_station/).

BASIS 5 AND THE SUSTAINABILITY TRANSITION: PARTNERING IN GLOBAL STEWARDSHIP

M.P. Weinstein & J.M. Krebs

ABSTRACT

A framework is presented for incorporating the concepts of sustainable development, sustainability science, especially for the urban coast, and the human dimensions into the amalgam of Tampa Bay research based on the integration of the biophysical and social sciences into an integrated, transdisciplinary approach to resolving sustainability issues. On a global scale, it is axiomatic that we are not living sustainably and must reconcile human use of natural resources with the planet's ability to provide them. The same trend occurs locally around Tampa Bay but on a place-based scale. The resolution of the paradox of the dual mandate; conflict management; systems model applications addressing human-ecosystem interactions that are at once dynamic, highly complex and non-linear; social learning; and the ecology of cities are discussed in the context of creating new knowledge and better understanding feedbacks and the consequences of our actions.

INTRODUCTION

Confronting the challenge of the [sustainability transition] is a series of great opportunities disguised as insoluble problems John Gardner, Common Cause

John Gardner was referring to the climate-energy issue, but his comment applies equally well to the greatest challenge of the 21st century: reconciling human use of natural resources with the planet's ability to provide them. Because we are a society built on instant gratification, achieving the sustainability transition will be difficult, since the generations that will suffer most from our failures have yet to be born. A tectonic paradigm shift is required in perceptions, attitudes and behavior toward stewardship and doing what parents should do: think about the *long-term*, so their children can have a better future. As Thomas Friedman comments, "green has to become part of America's DNA" (Friedman 2007), yet the road ahead remains fraught with obstacles and challenges.

The Sustainability Transition and Policy Resistance—As the world changes, decision makers and the scientific community increasingly recognize that we are not only failing to resolve the persistent sustainability problems we face, but are in fact *causing them* (Sterman 2002). Our well intentioned efforts with global initiatives, environmental "summits" and dialogue to confront environmental issues not only fail to resolve sustainability issues, but create unanticipated side effects. Today's decisions often provoke reactions (feedbacks) that we could not foresee, and they often become tomorrow's problems. Sterman calls this *policy resistance*, the tendency for interventions to be defeated by the response of the system to the intervention itself. As we invoke "green technology" as the panacea for the world's sustainability ills, these same words become powerful sponsors of advice and caution. We must do a better job understanding the consequences of our actions.

In the remainder of this paper, we present a preliminary summary and road map for the incorporation of sustainability science into resolving issues confronting the Tampa Bay ecosystem, as well as other coastal environs. The intent is not to be prescriptive but simply illustrative of this new approach in tackling the sustainability transition.

Sustainable Development—Sustainable development is a need borne of the recognition that the human condition has placed irreversible stresses on the relationship between the environment and

progress, and the recognition that more needs to be done to reverse unsustainable practices. The often-negative trends observed between population, habitation, wealth and consumption, technology and work, ecological connectedness and biodiversity are likely to persist well into the future (NRC 2002). Yet the very same trends offer a fabric for rigorous analysis and scientific advancement in understanding the threats to, and opportunities for, sustainable development. What will be required for a successful transition are critical advances in *basic knowledge*, in the social and technological capacity to utilize it, and in the political will to turn knowledge into action (NRC 2002).

Humans as a Coastal Species: Is Sustainable Development in the Coastal Zone Possible?—Humans use the coastal zone for living space, extractable resources, shipping and transportation, military activities, as a repository for wastes, and for recreation and aesthetics. The economic value of goods and services derived from the coast are immense. In the United States alone, the coastal economy within the area encompassing the watersheds out to the limits of the Exclusive Economic Zone (EEZ), approaches nearly half the Gross National Product (GNP) or \$4.5 trillion (US Commission on Ocean Policy 2004). The competition for finite resources and the need to buffer “core technologies” against the vagaries of nature (Roe and van Eeten 2001; 2002), however, often leads to conflict among human interests, and between humans and other biota. The resolution of “paradox of the dual mandate” (Roe and Van Eeten 2001; 2002) is a critical requirement on the path toward sustainability. Whereas complexity, interdependence, high levels of uncertainty, unpredictability, and dynamism characterize natural systems - traits that prevent competitive dominance by any one species - human-dominated systems require predictability and stability to ensure uninterrupted provision of resources for human use. The paradox of the dual mandate arises from the need to reconcile society’s desire to preserve, restore, and rehabilitate natural systems while at the same time ensuring the provision of reliable, predictable, and stable supplies of goods and services at a time of escalating demand (Roe and van Eeten 2001).

Rapid urbanization is having profound impacts on local, regional and global environments as well as the human systems that depend on them (Collins et al. 2000). In the case of land use, for example, mankind has diverted nearly 50% of terrestrial production for human uses (Vitousek et al. 1997). Every addition to the population requires approximately one-half acre of land vested in new dwellings, infrastructure, schools, police and fire departments, etc (Greer 1991). The ecological and environmental costs of this conversion can be measured by crowding, pollution, flooding, erosion, local extinctions of animal and plant species and other factors. The same trend towards significant resource exploitation for human use is clearly underway with respect to coastal aquatic resources. To not repeat the harsh lessons of land management, action must be taken and taken soon to prevent irreversible thresholds of change (Palmer et al. 2004; Weinstein et al. 2007).

The Emergence of Sustainability Science—

It has become increasingly clear that much of the workings of the world, and the challenges and opportunities these workings entail for a transition to sustainability lie in the interactions among environmental issues and human activities that have previously been treated as largely separate and distinct...in the next decade we will see research/[education] and problem-solving shift in focus from single issues to multiple interacting stresses.

US National Research Council (2002)

The reconciliation of human dominance of the Earth’s natural resources with the planet’s ability to provide natural capital over the long-term is the central challenge to a sustainable world. We

can no longer afford the luxury of “business as usual” simply because human activities continue to degrade local ecosystems and threaten the very life support systems that sustain us. An integrated and “place-based” (community level) understanding of such threats and the options for dealing with them is a central challenge for promoting the sustainability transition. *Social learning* (NRC 2002) is key, recognizing that the transition will be vested in the slow, interactive accumulation of scientific knowledge, technical capacity, management institutions, and public concern over extended periods (generations); societies must understand the long-term, large scale trends and transitions that have shaped past and present interactions of environment and development.

Any transition toward sustainability, moreover, should consider the dynamics evolution and interplay of social, economic and natural systems in an integrated *transdisciplinary* process (Naveh 2005). This process goes beyond individual stakeholders and themes to the identification of common threads, and drivers of systemic change. Achieving sustainability is not an open and iterative process inclusive of science, policy and public participation, but choosing among alternative development scenarios will require new and/or enhanced skills in conflict resolution and consensus building.

Kates et al. (2001) defined sustainability science as addressing the fundamental character of interactions between nature and society, and society’s capacity to guide those interactions along sustainable trajectories; yet the authors also realized that the resolution of competing interests were at the heart of the sustainability challenge:

... it will also require fundamental advances in our ability to address such issues [competing interests] ... as the responses, some irreversible, of the nature-society system to multiple and interacting stresses.

Thus, it is impossible to maximize all competing interests in a way that will satisfy all stakeholders, nor maintain human dominated ecosystems at some historic, relatively pristine baseline. It is also impossible to ignore the various economic and political powers that enter into the human part of the equation, creating substantial obstacles to social justice and consensus governance of the coastal zone. It is clear that a new approach is required, one that integrates traditional disciplines into the nascent discipline of sustainability science (Clark et al. 2003).

Three fundamental challenges confront society’s ability to acquire useful knowledge through research for sustainability planning (NRC 2002). They take the form of tensions between:

- *Broadly-based* versus *highly-focused* research priorities;
- *Integrative research* that is problem driven versus research firmly grounded in *specific disciplines*; and
- The quest for *generalizable* scientific understanding of sustainability issues versus the need for *place-based* understanding of the environment-society interaction that result in those issues.

To address these challenges the National Research Council (NRC 2002) posed three priorities for sustainability science research:

- Promoting research that integrates global and local perspectives in a *place-based* framework for understanding the interactions between environment and society;
- Focusing, at the outset, on a limited set of *understudied questions*, those that underpin the understanding of those interactions, and
- Promoting more efficient use of existing tools and processes that *link knowledge* and *action*.

The process should be one that integrates across styles of knowledge creation, bridges the gulf between the detached practice of scholarship and the engaged practices of engineering and management, and ultimately broadens knowledge of the workings and interactions of the world's environmental, economic, and social systems. The four core disciplines of sustainability science provide the foundation upon which sustainability research and education are built (Figure 1). The biological system emphasizes the intertwined fates of humanity and the natural resource base - biodiversity, restoration ecology and conservation biology are essential components. The geophysical system addresses climate and biogeochemical cycling, and is grounded in efforts to understand the earth as a system. The social system concerns itself with how human institutions, economic systems, and beliefs shape the interactions between society and the environment. It is couched in the human dimensions that rest on societal values, fundamentally a human endeavor. Lastly, the technological system enhances basic technological knowledge, designs, and processes that produce more social goods with less residual environmental damage.

Sustainability Science, Ecosystem-Based Management and Urban Ecology—

When human beings evolved, the challenge was survival in a world dominated by systems we could barely influence but that determined how we lived and died. Today the challenges we face are the result of systems that we have created [emphasis added]. The hurricane or earthquakes do not pose the greatest danger. It is the unanticipated side effects of our own actions, side effects created by our inability to understand and act in consonance with our long-term goals and deepest aspirations.
Stermann 2002

By 2050, approximately 60% of the world's population will live in urban settings; in the United States that figure is now nearly 80%! This demographic phenomenon, combined with the observation that most megacities (populations exceeding 10^7) are situated on coasts and estuaries, has led to habitat loss and fragmentation, regional and local climate alteration, depletion of water resources, degradation of land and water by contaminants, and have raised concerns among many scientists across multiple disciplines. This makes the ecology of cities an open frontier for sustainability science research, education, and outreach (Pickett et al. 2001; Palmer et al. 2004; Weinstein 2005; 2007; 2008; Weinstein et al. 2007), and with some of the nation's highest coastal population densities existing in the Tampa-St. Petersburg area, there might not be a geographic location better poised for, or in greater need of, a more detailed understanding of urban ecology in the context of the Bay environment.

Perhaps, more than any other human-dominated ecosystem, the urban coast presents a plethora of opportunities to link ecological and social science theory and practice using resource economics concepts (Collins et al. 2000). A realistic understanding of human actions on ecosystems will necessitate conceptual frameworks that explicitly include humans in the landscape. Such an approach is likely a more accurate representation of the ecology of urban systems with the potential to better inform environmental problem solving (Grimm et al. 2000; Weinstein and Reed 2005). The reasons are obvious; it is the human dimensions that drive political, economic, or cultural decisions that lead to or respond to change in ecological systems. As an effective facilitator of much of the area's environmental research, a valuable contributor to resolution of local policy issues, and a trusted steward of the Tampa Bay estuary, the TBEP is a logical and capable leader to guide the way in the emergence of urban coastal research integrated with the social sciences and humanities.

There is growing opportunity for those involved in the conservation and management of Tampa Bay to extend and integrate knowledge of the coastal metropolis combining our ecological lens

with a social and economic understanding at the different scales that govern the flows and cycles of critical resources and which link organizational arrangements and social institutions to the end-product of landscape management (Zonneveld 1989; Paul and Meyer 2001; Weinstein et al. 2007). Urban-rural gradients are complex, multidimensional constructs, but nonetheless, the analysis of such systems has become a powerful tool for understanding ecosystems across a wide range of defining variables, stress factors, disturbances, and other drivers (Pickett et al. 1997). Common themes in urban sustainability science research include questions of hierarchy and scale; how they are related to our ability to understand the dynamics of landscape change, biodiversity, wildlife distribution and vegetation patterns; and the reciprocal relationships between these factors and human activity. Moreover, Pickett and Cadenasso (2006) note that all ecosystems inhabited by humans should be “modeled to include individuals as well as the social aggregations they generate or influence”. They suggest further that “it is perfectly reasonable to incorporate such factors and processes into ecosystem models”.

Both the Baltimore and Phoenix Ecosystem Studies, NSF Long-Term Ecological Research programs have provided fruitful ground for understanding the ecology of cities and provide a useful framework for extending these efforts into coastal aquatic ecosystems. It is beyond the scope of this paper to review the burgeoning new literature on the subject, but simply highlighting that the emerging conceptual framework will, hopefully, capture the attention and imagination of BASIS scientists. Zipperer et al. (2000) suggested that there are two fundamental approaches to unraveling the dynamics and effective management of urban systems; the first is from an ecosystems perspective that considers the magnitude and control of fluxes of energy, matter, species, and information across landscapes; and the second, from a patch-dynamics perspective that focuses on spatial heterogeneity:

1. Complex ecosystems including cities are characterized by emergent properties, multi-scale interactions, nonlinearities, unexpected behaviors, and self-organization (Wu and Marceau 2002). They are thermodynamically open, composed of diverse components that interact nonlinearly, are often characterized by hierarchical organization, delayed response and feed-back loops, and frequently exhibit extensive temporal-spatial heterogeneity. Consequently, new approaches are evolving that address these properties; among them a wide array of system model types: diffusion-reaction, system dynamics, patch (or gap) dynamic, cellular automata, and fractal models (Wu et al. 1990; Wu and Levin 1994; Levin 1999; Sterman 2002). Sterman (2002) comments; “system dynamics helps us expand the boundaries of our mental models... helps people see themselves as part of a larger system, so that we become aware of and take responsibility for the feedbacks created by our decisions...that shape the world in ways large and small, desired and undesired”.
2. Patch dynamics also emphasizes spatial and temporal heterogeneity, non-equilibrium properties, and scale dependence of ecological systems and facilitates the coupling of pattern and process (Wu and Levin 1997). Both natural and anthropogenic disturbances are frequently responsible for these processes (Wu and Loucks 1995). By recognizing that spatial heterogeneity is a central attribute of all ecological systems, that hierarchy is a central structural theme of the architecture of complexity, and that complex ecosystems have both horizontal and vertical structure, hierarchical patch dynamics has become a promising approach to unraveling complexity because it addresses the spatial structure of landscapes, the flow of materials, energy, and information across mosaics/gradients, both as individual spatial components as well as the mosaic as a whole; in short, the patch becomes the fundamental structural and functional unit of ecosystems (Wu and Levin 1994; Wu and Loucks 1995; Pickett et al. 2000; Wu and David 2002; Wu et al. 2003). The landscapes thus become spatially nested hierarchies that can be effectively studied as such (Wu and David 2002).

In this context, the Human Ecosystem Framework refers to the various interactions among structural components and models humans as components of ecosystems (Machlis et al. 1997; Naveh 2005; Pickett and Cadenasso 2006). The concept of urban ecosystems as regions (landscapes) lends credibility to the boundary between planning and ecology. It recognizes that the human populace must have “access and responsibility for the distributed resources and amenities upon which they depend”. (Pickett and Cadenasso 2006). The authors note further, “watersheds, floodplains, large natural areas, connecting corridors, green buffers, green water management infrastructure, and recreational parks and playgrounds are all parts of the kind of patches that must be planned for, appropriately arranged and phased in as urban regions grow and change”.

From a transdisciplinary perspective, the Baltimore Ecosystem Study research goals are illustrative of the need to build sustainability science into coastal ecosystem management. They are presented here as general questions (Pickett and Cadenasso 2006):

1. How do the spatial structures of socioeconomic, ecological, and physical features of an urban area relate to one another and how do they change with time?
2. What are the fluxes of energy, matter, human built capital, and social capital in an urban system; how do they relate to one another; and how do they change over the long term?
3. How can people develop and use an understanding of the metropolis as an ecological system to improve the quality of their environment and to reduce pollution to downstream air and water, and watersheds [as coastal environs]?

In short, what are the institutional arrangements, constraints, and opportunities out there that test our mettle as scientists (natural, social, and economic)? While the knowledge of nature in cities sets the foundation for addressing ecological processes along the urban coast, it is not sufficient for understanding how those processes ultimately become a function of the feedback dynamics associated with interactions among social, ecological, and economic drivers. Like it or not, humans are here to stay (Weinstein and Reed 2005) and we need to “wedge sustainability” into our own consciousness, as well as engaging the public and decision makers to this reality (Coontz 2007).

Sustainability Science in the Tampa Bay Region—Decades of interdisciplinary collaborations by scientists and managers from the Tampa Bay region have laid the groundwork for the establishment and implementation of sustainability science as a tool to enhance our understanding of interactions between the Bay’s human inhabitants and its surrounding ecosystems. Biological and geophysical systems within Tampa Bay are studied at local, state and federal levels by such agencies and institutions as Pinellas County’s Department of Environmental Management, Florida’s Fish and Wildlife Research Institute, the University of South Florida, the U.S. Geological Survey, and the National Oceanic and Atmospheric Administration. Technological systems are being developed by these agencies and by more specialized groups such as the Center for Ocean Technology and Stanford Research Institute (SRI). What became clear at BASIS 5 was that the program could benefit from more frequent interactions among current investigators and specialists from the social sciences and other human dimensions disciplines whose expertise rests in understanding the economic and social underpinnings of the Bay’s economy, and its human needs. Continued involvement by the Tampa Bay Estuary Program to bring these partners together, facilitate research collaborations and communicate a collective environmental message will ensure the mortar necessary to construct a sound basis for creating sustainable communities around the Bay.

It was apparent from the diverse array of environmental topics presented at BASIS 5 and the interdisciplinary and interagency collaborations exhibited there, that members of the Tampa Bay research and management community are in a strong position to further integrate the natural and social sciences to address relevant challenges facing the Tampa Bay metropolitan area:

1. How do we combine applications of the natural sciences and human dimensions to manage the *urbanizing ecosystems* surrounding us?
2. How do we apply sustainability science to better understand the *interactions of multiple stressors*; e.g., habitat loss and alteration, point and non-point contaminant inputs, nutrient loads, shoreline management (reclamation, stabilization, etc.), cooling water withdrawals, maintenance and new dredging, watershed (land) development, infrastructure, intermodal transportation and human health impacts in the estuary and region?
3. How can independent scholarship (research, observation, assessment, etc.) and decision making *be better integrated* into policy formulation, adaptive management, and social learning in the region? Can the science and management highlighted at BASIS 5 more effectively influence regional policy?
4. How do we strike a *balance* between the capacity for ecosystem services and human needs? Has the carrying capacity of the Tampa Bay ecosystem reached or surpassed its limit? If so, what additional actions are necessary to counter, or at least offset, the ecological stresses of excess population densities on an overburdened coastal environment?
5. How can current operational systems for monitoring and reporting environmental and social conditions of coastal and urban Tampa Bay *be extended to provide more useful guidance* for the sustainability transition?

Where Do We Go From Here?—As noted earlier, a central challenge for the 21st century is to address the question: how can the Earth, its ecosystems, and its people interact toward the mutual benefit and sustenance of all? The urban coast and its watersheds present the Tampa Bay Estuary Program with unique opportunities to grow the sustainability science agenda to encompass the following issues:

- With the anticipated doubling to tripling of coastal urban populations worldwide in this century, how can we accommodate new infrastructure that is at once energy efficient, less material intensive, and with smaller ecological footprints?
- Rapid port and intermodal transportation expansion worldwide may require more land reclamation along estuarine shorelines. If the future tradeoff requires such construction how do we choose where and how much to build; understand the potential environmental offsets in terms of reduced carrying capacity, thresholds, and stresses that may shift urban-industrial estuaries to new, less-desirable steady states?

A critical challenge also concerns the emerging science of restoration ecology and our ability to mitigate (restore/create) system functions as the human populace expands. The NRC (2002) has called for informed dialogue on goals for the sustainability transition, a dialogue that is necessary if societies are to adopt a measure of responsibility for their choices, i.e., where it should be headed rather than passively navigate the currents of demographic, economic, and environmental change. With its broad expertise, and integrated approach, Tampa BASIS 5 scientists and managers can be “hands-on” leaders in coastal sustainability science. We hope that the principles and concepts presented here continue to be embraced by area scientists, managers and decision-maker, and that the fruits of this labor will indeed take center stage at Tampa BASIS 6.

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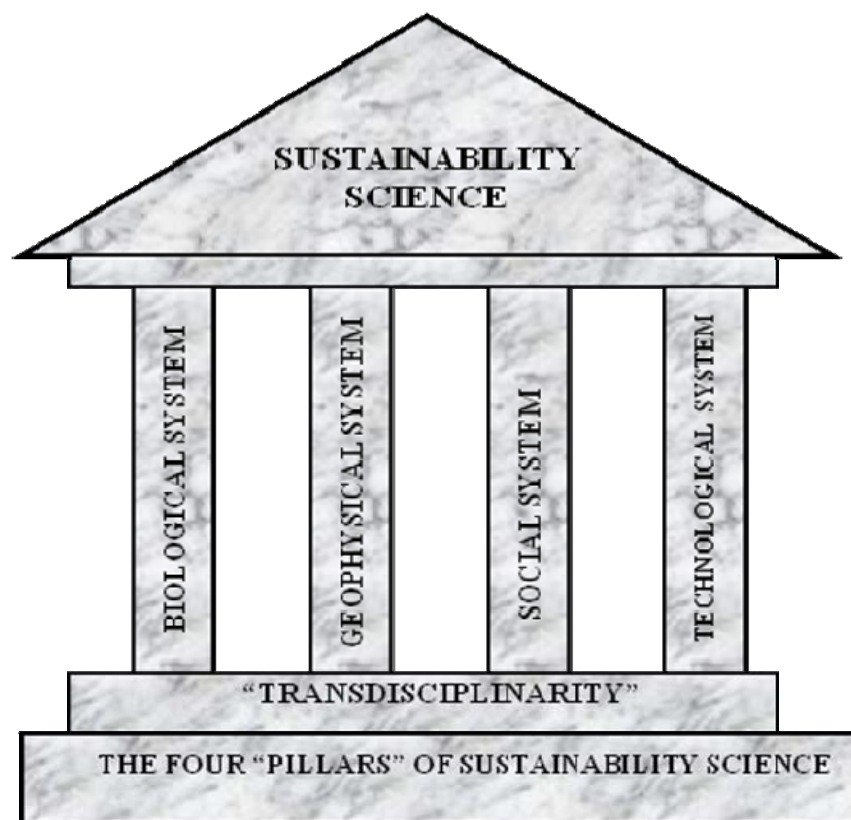


Figure 1. The four “pillars” of sustainability science, integrated through transdisciplinary approaches to problem solving for the sustainability transition.

ABSTRACTS
Session 12: Climate Change & Future Challenges

PRESENTATIONS

Ice Sheets and Sea Level Rise: It's Not Just the Melting That Matters. G.S. Hamilton (University of Maine) and B. Yeager (Clean Air-Cool Planet). The most recent IPCC projection for sea level rise calls for 18-59 cm (7-23 inches) by the end of this century. A large portion of this rise is attributed to reductions in ice sheet volume, but the projections only include the melting of ice (i.e., the transfer of liquid water to the ocean). Research carried out since the IPCC report was compiled shows that ice sheets are losing mass at a much faster rate due to dynamic effects (i.e., the transfer of solid mass - icebergs - to the ocean). This dynamic thinning is being driven by the synchronous and rapid speed-up of several large outlet glaciers draining the Greenland and West Antarctic ice sheets. The causes for the speed-ups are not fully understood, but they most likely represent a non-linear response to climate forcing (either atmospheric warming, or an increase in ocean temperatures). These dynamic effects raise sea levels much faster than melting effects alone, so it is critical that they are included in sea level forecasts. Current best estimates based on recent observations indicate a 1 m rise (40 inches) by 2100 is a more realistic planning scenario.

These are serious predictions that will have severe repercussions on U.S. coastal communities. For the Tampa Bay region, already facing a great threat from sea level rise, these new estimates increase the urgency for us to respond at both national and local levels. In order to slow the rate of warming and melting in the Arctic, we must act quickly to mitigate further pollution. We must also have a serious discussion about which areas of Tampa Bay are most likely to be impacted given these estimates, and what can be done to facilitate regional adaptation efforts moving forward. [gordon.hamilton@maine.edu]

Holocene Climatic and Hydrologic Variability as Recorded in the Benthic Foraminifera *Ammonia beccarii* from Tampa Bay, FL. K.J. Hoover (University of Nevada), D.W. Hastings, G. Brooks (Eckerd College), and B.P. Flower (University of South Florida). We present a Holocene length reconstruction of natural and human induced climate and hydrological variability in Tampa Bay, FL. Reconstructions were developed using benthic foraminifera, *Ammonia beccarii*, collected from a series of sediment cores from Hillsborough Bay. Stable isotope paired with foraminifera Mg data provide records of temperature and salinity changes. The chronology is constructed using a series of AMS ^{14}C dates on samples of *A. beccarii* and ostracods.

The data allow us to assess temperature and salinity changes during the last several thousand years, including the Little Ice Age and the interval of human influence. Our record does not show a strong correlation between Mg/Ca and $\delta^{18}\text{O}_{\text{carbonate}}$, suggesting changes in $\delta^{18}\text{O}_{\text{seawater}}$ of Tampa Bay and thus substantial hydrologic variability on a sub-centennial-scale. Maxima in foraminiferal Mg indicate high temperatures during the Medieval Warm Period; the lowest foraminiferal Mg at 1600 AD suggests a possible association with the Little Ice Age. Current research is underway to develop a temperature calibration for foraminiferal Mg in *A. beccarii*. [hastindw@eckerd.edu]

Estuarine Adaptation: The Incorporation of Climate Change into Estuary Management Planning. A. Hosking (Halcrow Inc.). Climate change and relative sea level rise have the potential to significantly affect the physical, natural and human environments of our estuaries and hence change the way we manage and use these valuable resources. Recognition of this potential is critical to the definition of technically sound management solutions, and in turn to the delivery of future sustainability for estuaries such as Tampa Bay.

In support of the draft U.S. 'Clean Energy and Security Act 2009', ongoing work by the U.S. Government Accountability Office to provide recommendations for climate change adaptation in the U.S. has identified the U.K., as an international leader in this field and are reviewing U.K. practices to inform their recommendations (<http://www.gao.gov/new.items/d09534t.pdf>).

In the U.K. a strategic planning process has evolved that considers potential changes over a 100-year period (typically to 2100) and determines the potential consequences for human and natural resources. Through

this process, the potential for significant changes in landforms, habitats, and flooding and erosion risks are identified. This enables review of the sustainability of current management practices, identification of more sustainable approaches where necessary, and provides opportunity to initiate those responses which necessarily have a long lead-time (such as relocation of buildings). The pioneering 'Foresight Future Flooding' study (2004) demonstrated that by adopting an 'integrated' approach to risk management we can manage risks down to tolerable levels at significantly less cost than approaches relying solely on existing solutions.

Climate change adaptation has evolved as an integral feature of the U.K. coastal and estuarine management process over the last 20 years. This presentation will review key elements of the U.K. framework related to climate change adaptation to identify lessons that could benefit the management of Tampa Bay, at this time when climate adaptation is set to become a far more prominent feature of U.S. processes. [hoskingas@halcrow.com]

THE PEOPLE IN BETWEEN: TAMPA BAY NATIVES FROM CONTACT TO EXTINCTION

B.W. Burger

ABSTRACT

Archival research of primary European documents has resulted in remarkably detailed accounts about some of Florida's last native inhabitants. While 16th century documentation broadly relates to the entire peninsula, more detailed information from the following centuries, while chiefly centering upon north Florida and the panhandle, provides glimpses into native cultures of the Tampa Bay region. Ethnographic data for the central peninsular Gulf coast are quite limited. A review is presented of presently available information and related speculations about the Ucita, Yagua, Tocobaga, and Pohoy, the Tampa Bay peoples (caught) between the Timucua to the north and the Calusa to the south.

With the native depopulation of the Caribbean islands, Spanish slavers hunted along the Florida coasts (Worth 2006). Landing parties and shipwreck survivors became potential disease vectors. While the direct atrocities of the Spanish are legendary, the introduction of European diseases was a greater cause of native mortality and cultural disruption. Among the preliterate native societies, cultural traditions were transmitted by word-of-mouth. With the deaths of both the story tellers and their young audiences, sociocultural systems fragmented. Yet even in the face of depopulation, enslavement, and increasing European colonization, native tribes continued their traditional customs of killing each other.

In April, 1528, Panfilo de Narvaez landed at a village somewhere along the western Pinellas peninsula. His expedition marched northeast to Old Tampa Bay, then north, likely to the village at Safety Harbor later known as the seat of the Tocobaga (Bandelier 1964; Scott 1981).

An unresolvable uncertainty relates to the native names found in the European accounts. These names, with various phonetic spellings, may have reflected the appellations of actual chiefs, temporary headmen, specific villages, geographic areas, specific dialectical subgroups, or larger language stocks, or conflation of various of these. Some native terms may be one language group's own name for another language group, rather than that group's name for itself. These many caveats should always temper the use of such names as have survived.

The De Vaca account describes the Narvaez landing site as consisting of small huts and one very large one capable of holding more than 300 persons (Bandelier 1964). When singular among smaller structures, these larger structures may have been chief's houses or council houses; when multiple, they may indicate communal living facilities, perhaps clan houses. Such features have yet to be discovered and archaeologically excavated. De Vaca's description of what was likely the Safety Harbor village relates nothing as to its size. A village some ten or twelve leagues to its north was described as having fifteen houses and a large, ripe maize field. (An additional difficulty in using the Spanish accounts arises from writers' unspecified use of either the *legua legal* of 2.63 miles or the *legua comun* of 3.46 miles in citing leagues. A single uniform Spanish league measurement was not established until 1801 (Lowery 1959).

Garsilaso de la Vega is the sole authority for the accounts of Narvaez' cruelties to "Hirrihigua," named as the *cacique*, or chief, of the landing site village. Garcilaso's book largely relates the story of the 1539 De Soto expedition and was written in Spain sixty years after that event, relying on three informants and Garcilaso's own imagination (Varner and Varner 1980). The account of

the “Gentleman of Elvas,” a member of the expedition, identifies “Ucita,” not “Hirrihigua,” as the local *cacique* in 1539 (Lewis 1907).

According to Garcilaso, after Narvaez had cut off Hirrihigua’s nose, fed his mother to the war dogs, and headed north to his own fate, an expedition searching for Narvaez arrived from Cuba, was tricked into sending some men ashore, and Juan Ortiz was captured. After three years in captivity, Ortiz escaped to Mocoso, a *cacique* who lived two days journey from the bay, where he spent nine years, until De Soto’s arrival (Smith 1968).

In 1539, De Soto landed somewhere along the southeastern shore of Tampa Bay and occupied the village of Ucita that had been abandoned just before he arrived. The accounts of expedition members Elvas and Biedma supply some details about the sociopolitical organization of the bay area: In about 1531, Mocoso, a *cacique* living some eight leagues from the bay, had raided and burned the town of Ucita. *Cacique* Ucita, captor of Ortiz, had fled to another of his coastal towns. Twenty to thirty leagues distant from Ucita lived “Hirriparacosi,” to whom Mocoso, Ucita, and “all they that dwelt along the coast” paid tribute (Bourne and Smith 1904). This name, which is Timucuan, indicates the bay region was then under the hegemony of the Timucua to the north (Granberry 1993).

Mocoso’s previous raid on Ucita indicates Hirriparacosi had less than full control over his underlings. Both Ucita and Mocoso were independent, localized simple chiefdoms, each consisting of a number of villages. Mocoso complained to De Soto about the chiefs Neguarete, Capaloey, Ucita, and Orriygua. He had been threatened by these other regional *caciques* because he had befriended De Soto and given him Ortiz (Bourne and Smith 1904).

Thus, there were numbers of lesser chiefs - at times fighting each other, who were subservient to a more powerful chief to whom they paid tribute, and these linkages crossed linguistic boundaries: Elvas wrote that Ucita and Mocoso spoke mutually unintelligible languages. “Hirriparacosi” is certainly Timucuan, as is the honorific prefix portion of “Hirri-higua.” So rather than a tightly controlled, centralized paramount chiefdom under a Timucuan overlord, the Tampa Bay region apparently consisted of squabbling independent fiefdoms wherein one gained a certain, temporary ascendancy over the others.

What was the source of Hirriparacosi’s power? Perhaps it was maize. Narvaez had found some maize at Safety Harbor, but more ten to twelve leagues north. De Soto found no great amount until he was over twenty leagues from Tampa Bay. Ortiz said the land of Hirriparacosi, thirty leagues inland, was fertile and “abounding in maize.” Perhaps this was the key to Hirriparacosi’s power. With such a reliable food source capable of supplying a storable surplus, Hirriparacosi would have had the ability to support a larger body politic and a corresponding number of warriors.

In 1549, Tampa Bay was visited by Dominican missionaries led by padre Luis Cancer. After a landing party went inland, Juan Munoz locally held captive since the De Soto expedition, appeared and reported that two of the religious had been killed by the Indians and that a sailor was being held captive. Against the advice of his fellows, Cancer swam ashore, where he was killed in sight of his shipmates, who then cast off to return to Cuba (Lowery 1959).

Also in 1549, three Spanish ships wrecked along the west coast. One of the survivors, Hernando Fontaneda, was held captive by the Calusa of the Charlotte Harbor region for seventeen years. He wrote in his memoir that from north to south along the west coast, the tribes were the Apalachee, Mogoso, Tocobaga, Osiquevede, and Calusa (Smith 1944). In 1567, Governor

Menendez sailed to the Safety Harbor site on Old Tampa Bay, bringing Calos, paramount chief of the Calusa, to make peace with Tocobaga. By this time, the control of the west coast was held by these two, long-warring Indian leaders.

This was a meeting between two paramount chiefs. Calos had fifty subject towns, speaking at least twenty-four different languages (Hann 1991). Tocobaga summoned some twenty-nine *caciques*, 100 headmen, and others to join him - more than 1500 people - for the parlay. The level of sociopolitical organization of the Tampa Bay region had radically changed since the time of Hirriparacosi.

The accounts do not give names or locations for Tocobaga's vassals. However, a general boundary of Tocobaga territory is indicated: When addressing Menendez's desire to find a supposed connection between Tampa Bay and the St. Johns River, Tocobaga told him such an effort would not be possible without more men due to the numerous and hostile Indians on the way; that is, the region to the east/northeast of Tampa Bay was outside his control. By 1564, Calusa influence extended as far north as present Orlando. Further, Fontaneda wrote that the people of Canegacola, living on or near the "river of canes," probably the Suwannee, were subjects of Tocobaga (Smith 1944).

So in less than forty years, the Tocobaga, a group not mentioned in any earlier accounts, had taken control of much of the central west coast. As rivers often served as natural, territorial boundaries, it may be that the Manatee River was then the boundary between Calusa and Tocobaga (c.f. Smith 1968; Shapiro 1986).

The fluidity of native alliances is apparent: After the 1567 execution of Calos I by the Spanish at Estero Bay, the chiefs of two settlements previously subject to the Calusa transferred their allegiance to Tocobaga (Hann 1991:223). The boundary zone between the two politics certainly fluctuated. Complex chiefdoms experienced periods of emergence, expansion, and fragmentation, with regional simple chiefdoms cycling between greater or lesser degrees of centralization (c.f. Anderson 1994).

In 1568, Menendez Marquez sailed to Safety Harbor and found that the garrison left there by his uncle had been slaughtered. He burned the deserted village, including its temple structure. Father Rogel learned that Tocobaga's "captain" had ordered the killings. *Cacique* Tocobaga had been absent "in his own village at the mouth of the bar," perhaps at the Maximo Point site (Hann 1991). There was no native response to the torching of the village and no record of any subsequent Spanish retaliatory expedition.

During the years 1568-1608, Menendez's colonization attempts in south Florida failed. Spanish efforts largely centered on St. Augustine, with religious missions established to the north, and then spreading west into Apalachee. Guale Indian attacks from the north and English incursions into Virginia kept Spanish attention directed northwards. A probable epidemic occurred in 1595. Reports of incursions and attempts to find shipwreck survivors led to occasional Spanish coastal searches and visits to Charlotte Harbor during the early 1600s.

In 1608, the Potano Timucua of north central Florida were being threatened by raids from the Tocobaga and the Pohoy from Tampa Bay. Who were the Pohoy? Swanton thought they were the Ucita from Hillsborough Bay (Swanton 1979). Milanich thought they were equivalent to the Capaloey (Milanich and Hudson 1993). Whatever their origin, the complex chiefdom of Tocobaga had changed. Power was shared in a Tocobaga-Pohoy alliance, independent of the Calusa and the Timucua.

This alliance sent raiding parties to the north in the early 1600s. Raiders canoed a coastal route, then went up the Suwannee and Santa Fe rivers. In 1611, they attacked a Spanish mission on the lower Suwannee, killing seventeen converted Indians. Spanish retaliation was swift: From St. Augustine, Governor Olivera sent a force to Tampa Bay that killed the offenders. Among the beheaded prisoners were the Tocobaga and Pohoy *caciques* (Hann 1996; Worth 1998).

The death of the Tocobaga *cacique* apparently resulted in a power shift in the alliance. In 1612, a force sent on a mission to Calos stopped at Tampa Bay to visit the new *cacique* of Pohoy, then continued south. There is no mention of a new *cacique* of Tocobaga, however (Swanton 1979; 1985; Hann 1991).

A 1614 Spanish report described further tensions in the region. The Calusa paramount had sent 300 war canoes to the province of Mocoso, where some five hundred people were killed. The paramount then sent a dozen captive survivors to St. Augustine to warn both the Spanish and their Indian allies of his opposition to them. Governor Guillamas ordered retaliation, dispatching two armed launches to “exact the greatest punishment possible” (Worth 2005). It appears that this Calusa raid had bypassed Tampa Bay without any interference from what remained of the intervening Tocobaga-Pohoy alliance

During the 1620s, raiding occurred between the Pohoy and the Amacano - the latter apparently located along the coast north of Tampa Bay. In late 1628/early 1629, Governor Borja sent a force to Tampa Bay to bring the “captain” of the Pohoy to St. Augustine to negotiate a peace with the Amacano (Worth 1998).

By 1634, the Pohoy were apparently allied with or subject to the Calusa (Hann 2003). In 1636, further Pohoy raids resulted in a Spanish retaliatory expedition (Worth 1998). Significant for the entire peninsula are reports of epidemics of plague, small pox, and measles for 1612-1617, 1649-1650, 1655, 1659, and 1672 (Hann 1988; 1996). Remnant survivors of increasingly fragmented native societies became wandering disease vectors.

In 1674/75, Bishop Calderon of Cuba made a visitation to the peninsula. Describing points north, he wrote that four leagues beyond the principal village of the Calusa (at Estero Bay) was the bay of Tampa - which would indicate that the original bay of that name was Charlotte Harbor. Six leagues from the beach of Pusale (location unknown) was the Pojoy river, and twelve leagues farther, that of the “Tocopacas.” While Calderon mentioned the Pojoyes, the Pineros, and the Tocopacas north of the Calusa, he did not himself visit the region (Wenhold 1936; Hann 2003).

Prior to 1675, some Tocobagas had moved to a settlement of mixed tribal remnants on the Wacissa River. In 1678, visitador Leturiondo counted 348 people in this village (Hann 1993). Their assigned work was to transport produce by canoe down the Wacissa and Aucilla rivers to the Gulf, south along the coast, then up the Suwannee and Santa Fe rivers to a landing from where it was transported overland to St. Augustine. They also portaged cargo to lighten Spanish ships otherwise unable to cross the shallow Suwannee River mouth (Bushnell 1978).

In 1679, Governor Salazar sent a force including Timucua allies to Calos to ransom some Spanish captives. Traveling by canoe down the Santa Fe and Suwannee rivers to the Gulf, the force followed the coast south to Alcola, a village of up to 300 persons, then traveled overland to Pohoy, described as a comparably sized village. The Pohoy *cacique* reported that Calos had ordered him to prevent the expedition from going any further. The warning was ignored, and the force proceeded south to “Elafay” (likely on the Alafia River), a village that contained up to forty people; then moved on to “Apojola Negra,” containing up to twenty people; then on to

“Tiquijagua,” a village of about 300 people. At this point, their Timucua allies decided to heed the Calusa warning and turned around, soon followed by the Spanish contingent (Hann 1991). This account mentions no Tocobagas at Tampa Bay. It does clearly indicate that the Tampa Bay area was under Calusa control at this time. The Tocobaga-Pohoy alliance was effectively gone.

There is little information about the Tampa Bay region for the 1680s. Natives from Guale were fleeing south from coastal and interior Georgia into peninsular Florida. Yamasees were raiding the northeast Florida missions. In 1688, in Apalachee, Governor Losada met the son of the Calusa *casique* and many of his leading men (Hann 1991). That such a large, important party could travel unmolested all the way from the Charlotte Harbor area clearly indicates Calusa control of the intervening west coast. In 1695, visitador Florencia noted that Tocobaga were still present at the Wacissa River settlement (Hann 1993).

In 1699, acting on a rumor that the British were planning a settlement on Tampa Bay, Governor Ayala sent Ensign Rodrigo, three soldiers, and a group of Timucua allies overland to investigate. In approaching the bay, they camped at an abandoned village called “Elafay” (c.f. Alafia River), then traveled to the southern border of the bay to a river they named the “San Salvador,” called “in the language of the natives Alameva” - likely today’s Manatee River (Childers 2002).

A diary kept by Diego Pena, one of the soldiers, relates that:

“[t]he village of Pojoy remained behind us to the west [sic], twelve leagues distance from that bay by land...From the coast on the south, leaving the edge of the bay in order to go to Pojoy it takes three days by sea according to what these Indian guides say. [It follows from this that the then village of Pojoy was located north of Tampa Bay.] Also, inland, not very far from this bay, there are the following villages: Acassa, Alachepoyo, Amaca, Talafosole, and Ereze.... There are more places in this vicinity which are the following: la Feaba Tesca, Leley, Tequi, Sole, Juyula and Piaya which is a village that has many people. These [latter] are the ones that are close by the said bay. I have not seen these villages but those that know of them have declared it thus to me” (Childers 2002).

The expedition was visited by the *cacique* of Cayuco, who knew of no British incursions. Possibly at Point Pinellas, or perhaps near the mouth of the Manatee River, a large cross was erected with an attached warning sign. Ensign Rodrigo instructed the natives to send word north if any Englishmen appeared. Some of them told him that “[t]he Indians that had lived at the mouth of this bay had been called Tampas and others call them Vantabales, but they had gone north” (Childers 2002). This appears to reference the formerly present Tocobagas and their migration to the Wacissa River area.

The English slave trade was a major factor affecting the entire Southeast during the period 1670-1715 (Gallay 2002). In the early 1700s, slave raiding by Creeks and Yamasees, and British invasions, led to Spanish abandonment of the west Timucua hinterland. The destruction of the north Florida missions gave raiders open access to the southern peninsula. In 1708, a British report stated that Creek slavers had to go as far south as the firm land would permit in order to find and capture the natives (Braund 1993).

In 1701, the Wacissa River Tocobaga apparently moved further inland as a result of raids. After that year, some Pohoy and Alafaes from Tampa Bay had moved to a village nine leagues south of St. Augustine, as well as to a mission site near present Gainesville. In 1726, that mission site was identified as a mixed settlement of Timucua, Pohoy, and Alafaes (Hann 1991).

In 1718, Ribera was sent to the mouth of the St. Marks to build a blockhouse (Kerrigan 1951). Plans for the fort indicate there were two small Tocobaga settlements in the vicinity (Hann 1988). Ribera found no more than two dozen Tocobaga in the area and settled them along the Wakulla River upstream from the blockhouse. Later in 1718, a small Tocobaga community on the Wacissa was attacked by Pohoy coming in from the Gulf (Hann 1991; 2003): Former allies, now enemies. A few Tocobaga were still living in the St. Marks region in 1723 (Hann 1988). By 1755, they had been completely destroyed by the raiding Creeks (Covington 1957).

In 1727, an epidemic killed all but a few of the refugee Pohoy, Alafaes, and Amacapira at a village nine leagues south of St. Augustine. Remnants were present there in 1734, when Governor Sanchez tried to move them closer to St. Augustine. Instead, they fled further south (Hann 1991; 2003).

In 1737, the virtual extermination of a mixed Pohoy/Amacapira village by the Bomto included the deaths of its two *caciques*. The village was apparently located somewhere inland, west of present New Smyrna Beach (Hann 1995).

When Celi surveyed Tampa Bay in 1757, he encountered a few, unidentified Indians, armed with muskets (Ware 1971). The weapons probably indicate a Yamasee or Creek hunting party. The 1769 Jeffreys map shows islands near the mouth of the Withlacoochee River as the "Pogoi Keys," with subsequent maps of the 1770s showing these near the mouth of the Suwannee. These are likely historical, cartographic artifacts transferred from earlier maps, such as the 1711 Crisp map. Evia's 1783 voyage found only Uchises, Talapoosa, Choctaw - all non-Florida Indians - and Cuban fishermen at Tampa Bay (Holmes 1965). Folch's 1793 exploration of the bay similarly found only Creek tribes in the region (Kinnaird 1946). The original natives were gone.

While European accounts relating to the native history of Tampa Bay are limited in detail, this summary presents at least the general story. If the University of Florida's proposed project at the Havana, Cuba, archives should occur, much more may be learned about the last natives of Tampa Bay, the people in between.

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PREHISTORIC COASTAL HABITATION AND EARLY MARITIME TRAVEL ON OLD TAMPA BAY

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ABSTRACT

The vast natural resources of Old Tampa Bay, part of the largest open water estuary in Florida, enabled the development of fairly large prehistoric populations. Recent research by Weedon Island Preserve Cultural and Natural History Center, in partnership with other professionals and volunteers, has led to new information about early occupation sites and subsistence, and an exciting discovery of the first open-water dugout canoe found on the Gulf coast. We present information first on the Shoreline Midden (8Pi11569) public archaeology restoration project and then on the Shoreline Canoe (8Pi11624), a 1100-year old, saltwater dugout canoe documented to be more than 12 meters long.

INTRODUCTION

For several thousands of years, early people inhabited the coastal shores of Old Tampa Bay utilizing the rich resources of the estuarine environment. This led to the development of some fairly large pre-historic populations in this region of Tampa Bay (Milanich 1994), part of the largest open water estuary in Florida. Recent investigation of two sites, Shoreline Midden (8Pi11569) and Shoreline Canoe (8Pi1164) at the Gateway Tract of Weedon Island Preserve has led to new information about this coastal habitation and adds substantial evidence to open water travel. This State conservation land is managed by Pinellas County's Department of Environmental Management, Environmental Lands Division, and is surrounded by dense urban development comprising the St. Petersburg/Tampa Bay area. Weedon Island Preserve is Pinellas County's largest marine preserve, and consists of approximately 1500 hectares of coastal uplands and mangrove forests.

PROJECT AREA DESCRIPTION AND BACKGROUND RESEARCH

The Shoreline Midden (8Pi11569) and Shoreline Canoe (8Pi1164) sites are located within the Gateway Tract, a 275-ha, low-lying coastal fringe of mangrove forests and isolated upland ridges found on the west side of Old Tampa Bay. Habitation sites dating to the early Manasota culture are found on the few upland areas, including a burial mound (8Pi11700). Since the land is owned by the State of Florida and leased to Pinellas County for resource management, there are pertinent federal, state, and local laws that affect its management of those resources. Examples of applicable laws include Florida Administrative Code: Chapters 267 and 872; the federal Native American Graves Protection and Repatriation Act (NAGPRA); Pinellas County Municipal Code: Chapter 146; and Pinellas County Ordinance 08-29.

The Shoreline Midden site, located in portions of Section 36, Township 29S, Range 16E and Section 1, Township 30 South, Range 16 East, is adjacent to the Yat Kitischee or Moog Midden (8Pi1753) site investigated in the 1990s and eligible for listing in the National Register of Historic Places. Yat Kitischee was an exemplary public archaeology project that included hundreds of volunteers and resulted in a publication, *Yat Kitischee, The Archaeology of a Prehistoric Coastal Hamlet*, (Austin 1995) that defines much of what is known about the ancient cultures that lived in the Tampa Bay area and greater Central Gulf Coast region. That project served as the baseline for the present research in this area. The Shoreline Canoe site (8Pi11624) is located within the Gateway Tract at Township 30S, Range 16E, Section 1, northeast and within 400 meters of the Shoreline Midden site.

Located within the Weedon Island Preserve and contiguous with these two sites on Old Tampa Bay is the eminent site Weeden Island (8Pi1), the first identified site of the Weeden Island culture. Weeden Island culture is now known to extend from this Tampa Bay region into North Florida, southern Georgia and Alabama. Over 3,000 Weeden Island sites have been recorded since this discovery in the 1920s by Dr. Jesse Walter Fewkes of the Smithsonian Institute. In Dr. Fewkes' 1924 publication, *Preliminary Archeological Explorations at Weeden Island, Florida*, he described the extraordinary pottery recovered from the burial mound and types of burials found in several strata. He noted that no European contact had taken place at this site (Fewkes 1924). Over a two-year period this excavation was the premier public archaeology project where thousands of visitors were ferried to Weedon Island to observe the excavations under the guidance of Dr. Fewkes' assistant, Dr. Mathew Stirling.

Some of the greatest densities of sites in Pinellas County are located within the Old Tampa Bay area from Grande Bayou to Safety Harbor, including two sites listed on the National Register of Historic Places, Weeden Island and Safety Harbor. Many of these sites encompass evidence of occupation from the Manasota, Weeden Island, or Safety Harbor cultural periods that range from 500 BC to AD 1750 (Austin 1995).

Southeastern Archaeological Research Inc. (SEARCH) conducted a thorough background investigation into previously recorded prehistoric sites within the Gateway Tract, including sites that may have referred to the Shoreline Midden Site (8Pi11569). This research and subsequent survey results are documented in the grant report, Weedon Island Preserve Gateway Tract Archaeological Survey Project Grant Number S0828 (Austin 2008). This work included comparisons of old aerial photographs and modern shorelines. Five prehistoric sites were previously recorded within the Gateway Tract. SEARCH concluded that one of those sites, Dan's Island (8Pi53) as described in the Florida Master Site File (FMSF), correlates to the Shoreline Midden (8Pi11569) site. Another recorded site, Cabbage Patch Point (8Pi57) could not be located and appears on the USGS 7.5 Safety Harbor quadrangle map in the locale of the Shoreline Midden. It is probable that this site also refers to the Shoreline Midden site.

In the SEARCH report, their review of old maps and 1943 aerial photographs indicated the Shoreline Midden site was located on a peninsula. Today this site supports dense mangrove swamp vegetation and disturbance associated with mosquito ditching conducted in the 1960s. In addition to the previously recorded sites, a new site recorded is a prehistoric dugout canoe located approximately 400 meters from the Shoreline Midden site. Shoreline Canoe (8Pi11624) is buried in the intertidal zone of the coastline on Old Tampa Bay. Our recent investigation revealed a pine canoe with a raised bow. The length of the existing canoe is 12.17 meters from bow to a broken stern. Radiocarbon dates in the 2-sigma range are AD 690 – 1010 (Cal BP 1060 to 940). This indicates a temporal association with the Shoreline Midden site during the late Weeden Island period.

PROJECT DESCRIPTION – SHORELINE MIDDEN

The Gateway Tract Shoreline Midden (8Pi11569) Investigation and Restoration Project originated through reports of aboriginal habitation sites in the Gateway Tract at Weedon Island Preserve. Survey of this area confirmed a heavily pot-holed shell mound on state lands located on an upland dune ridge on Old Tampa Bay. The FMSF confirmed reports of sites in the vicinity, but exact locations were unknown. Through a grant funded in part by the Bureau of Historic Preservation, a cultural resource survey was conducted for the Gateway Tract as well as the plan to include the public in the investigation and restoration of the Shoreline Midden site. A 1A-32 Archaeological Research Permit (Number 0708.26) was applied for and received in 2007.

Accessed only by boat, this project involved extensive planning and coordination. Tides and weather were important considerations. Since all transportation was by Pinellas County boats, all volunteers became Pinellas County short-term volunteers with job descriptions. Initial preparation of the mound site, which was covered with Brazilian pepper, also involved seasoned volunteers working side-by-side with County staff. The 167-m mound was cleared of debris and exotic vegetation. Over 250 sandbags were filled, and brought to the site by pontoon boat. Shallow shoals around the area would not allow the overburdened pontoon close to shore, so the sandbags had to be shuttled in with a flats boat.

Three days were devoted to the investigation and restoration went quickly. Over 35 volunteers, many university students, supported this phase of the project and gained valuable hands-on opportunities to conduct real archaeology. Three archaeologists, Robert Austin, Ph.D., Southeastern Archaeological Research Inc., Richard Estabrook, M.A., Florida Public Archaeology Network (FPAN), and Phyllis Kolianos, M.A., Pinellas County, supervised participants and Pinellas County's Communications recorded the event and interviewed several of the volunteers.

RESEARCH METHODS

After an initial reconnaissance survey of the area, it was determined that a comprehensive survey of the entire Gateway Tract was needed. This survey was conducted under contract with Southeastern Archaeological Research, Inc. (SEARCH) of Gainesville, Florida through their Tampa Bay office in Riverview, Florida. The Shoreline Midden site was an opportunity to expand on this survey and accomplish several site management goals while addressing some general research questions. In the event of the discovery of unmarked human remains, investigators were prepared to comply with the provisions of Chapter 872.05, Florida Statutes.

Objectives - Five major objectives were defined for the Shoreline Midden restoration project. These included:

- 1) Restore the pot-holed mound site to a resemblance of its original contours to discourage future looting and vandalism.
- 2) Conduct limited data recovery from selected looter holes to increase knowledge about the Manasota culture in this coastal environment.
- 3) Engage the public in this investigation exposing volunteers to the damage of destructive vandalism and the value of archaeological research.
- 4) Spread this message to the general public through public archaeology events, general programs and communications, and publications.
- 5) Work with local and state partners interested in supporting this project.

FIELD METHODS

Initial site reconnaissance—Prior to the fieldwork, an initial site reconnaissance was conducted by Phyllis Kolianos, Richard Estabrook, and J. Bart McLeod in November 2007. Access to much of the midden was restricted by thick vegetation (mostly Brazilian pepper trees and black mangroves). At least six major looter holes were identified. Surface inspections of all six locations recovered small pieces of prehistoric pottery and some broken shell tools left behind by the looters. The position of the abandoned materials and the “style” of digging used by the looter(s) indicated that most of the six areas likely had been visited recently (within the past year) by a single individual or a small group of individuals working together. The pottery was recovered; the broken shell tools were not.

Site preparation and clearing—Before the project could get started, some of the invasive Brazilian pepper vegetation that covered much of the site was removed by Weedon Island

Preserve personnel and long-term volunteers. Clearing this vegetation provided a safe and open working environment for the volunteers and allowed the participants access to the piles of disturbed shell and soil that was used to re-fill the looter's holes. A path was cleared that allowed access to the main portion of the site from the shoreline/beach landing area. The area on top of the midden and around many of the major looter holes was also hand cleared of vegetation at ground level.

Limited data recovery—As little was known about the site prior to this investigation, a limited data recovery strategy was implemented. Originally three of the large looter holes, designated Units A, B, and C, were selected for the limited data recovery effort. The large number of volunteers allowed a fourth unit, designated Unit D, to be opened in a looter hole between Units A and B. The data recovery plan included cleaning out the backfill and debris that had washed into the hole; creating a vertical surface along one side of a looter trench of which a profile drawing could be made; and screening the previously-disturbed back dirt from within and immediately adjacent to each of the selected holes. Once the loose, previously-disturbed shell matrix was removed for each unit, each profile was drawn and photographed, and the soil colors were determined using Munsell color charts. Each unit was then individually mapped in plan view and mapped as part of the larger site universe.

In Units D and C, excavation continued along the profile face below the loose, previously-disturbed shell and into possibly undisturbed midden matrix in order to gain a better understanding on the various occupations at the site and to determine how deep the midden actually extended. In Unit D, it appeared as if the shell matrix was not very deep, perhaps 30-40 cm, as the looter(s) had begun to undercut the midden by digging horizontally into an area that appeared to have been backfilled and re-contoured. Faunal remains recovered from the disturbed backfill from Unit D revealed possible human remains. Dr. Erin Kimmerle from the University of South Florida, Forensic Anthropologist, confirmed a small finger bone and portion of a vertebra. Dr. Ryan Wheeler, the State Archaeologist was notified, who directed the remains to be re-interred at Unit D.

An intact ash/shell feature was encountered in Unit C at 45 cm below surface. This feature was carefully excavated in natural soil zones by Dr. Robert Austin and a group of volunteers. This roughly circular area of crushed shell and ash appeared to be a hearth or dump area. It was obviously denser and contained much smaller fragments of shell than did the surrounding shell matrix. All of the material from this feature was excavated and bagged separately. A sample of the shell/ash lens matrix was retained for possible C¹⁴ dating. Charcoal (0.2 g) from Feature 1 at 67 cm was sent for AMS dating to Beta Analytical, Inc., with the calibrated date to 2-sigma range of AD 410 – 590 (Cal 1540 to 1360 BP).

The spoil from each looter hole was screened through 0.6-cm hardware cloth screens suspended in portable wooden frames. In some cases, self-supporting shaker screens were used. In other units, frames were suspended with nylon ropes from the overhead branches of the larger trees. All pottery, shell tools, and other modified materials were bagged by zones separately as originating from either disturbed or undisturbed areas. As excavation into undisturbed midden deposits was limited, different anthropogenic strata were not encountered and arbitrary levels were not assigned. Samples of the various sources of food shell (i.e., oyster, clam, gastropods, and the like) were also recovered. All larger bone specimens were recovered (those easily recovered in the mesh screens), but no attempt was made to recover every fragment of bone or shell, especially from the disturbed complexes.

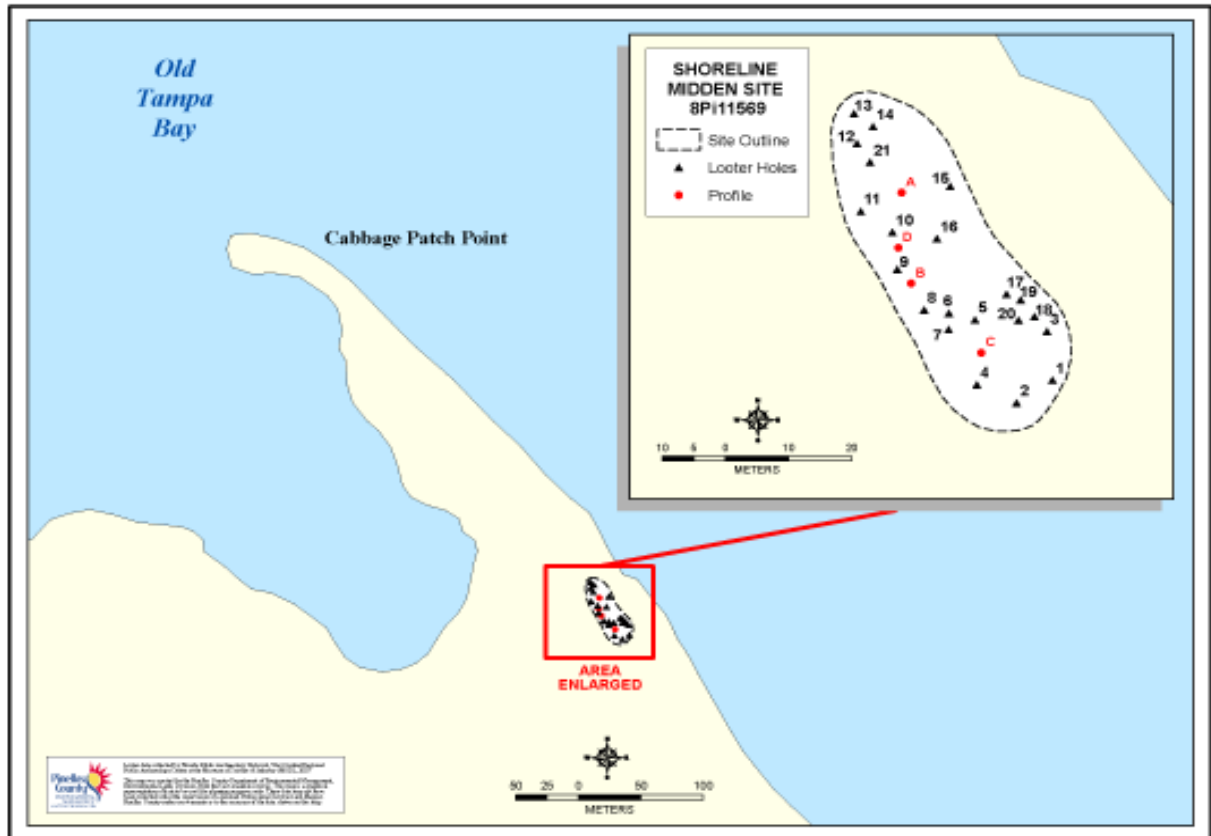


Figure 1. Overview site plan with GPS location of looter holes and looter hole units A, B, C, and D profiled.

GPS location of looter holes—As only a limited number of the looter holes could be investigated, and the vegetation across the site inhibited the development of a map of the entire site, Global Positioning Systems (GPS) hardware and software were used to map the locations of all the obvious damage. All looter holes were flagged and numbered sequentially from 1 to 23. Units A-D were mapped individually. A Trimble GeoExplorer XH GPS receiver with a detachable hurricane antenna was used to collect the locational data. Locations were taken from the center of the looter holes/trenches or as close to that location as was physically possible given vegetation and other obstructions. These data were later post-processed to provide the most accurate locations of each of the looter holes.

Site reconstruction—The looter holes and midden contours were reconstructed using sand bags filled with culturally-sterile sand and disturbed midden matrix recovered from the back-dirt piles found next to the looter trenches. The sandbags had previously been filled with sand by county staff and volunteers and brought to the site by boat. Volunteers carried the bags from the beach landing up onto the midden, where they were distributed out to the various looter holes once the limited data recovery and mapping activities were completed.

Due to the large number of looter holes and the limited number of sand bags, the bags were used as a liner between the surface of the looter holes and the disturbed shell matrix used to level the area. Each hole had at a minimum a single layer of sand bags between the looter ground surface and the refilled spoil used to recreate the land surface. Shovels, rakes, and other hand tools were used to backfill the area and to level out the previous damage. This effort created a surface that will allow any future investigation to determine undisturbed midden from re-deposited fill, but

one that will also discourage future looting as there is now no way to tell by looking at the surface which areas had been previously disturbed by illicit digging.

LABORATORY METHODS

All material recovered from the site was returned to the Weedon Island Preserve Cultural and Natural History Center where it was cataloged and processed. Each provenience was assigned a Field Specimen (FS) number. The paper FS log was then transferred to a computer spreadsheet so that the analysis information could be included.

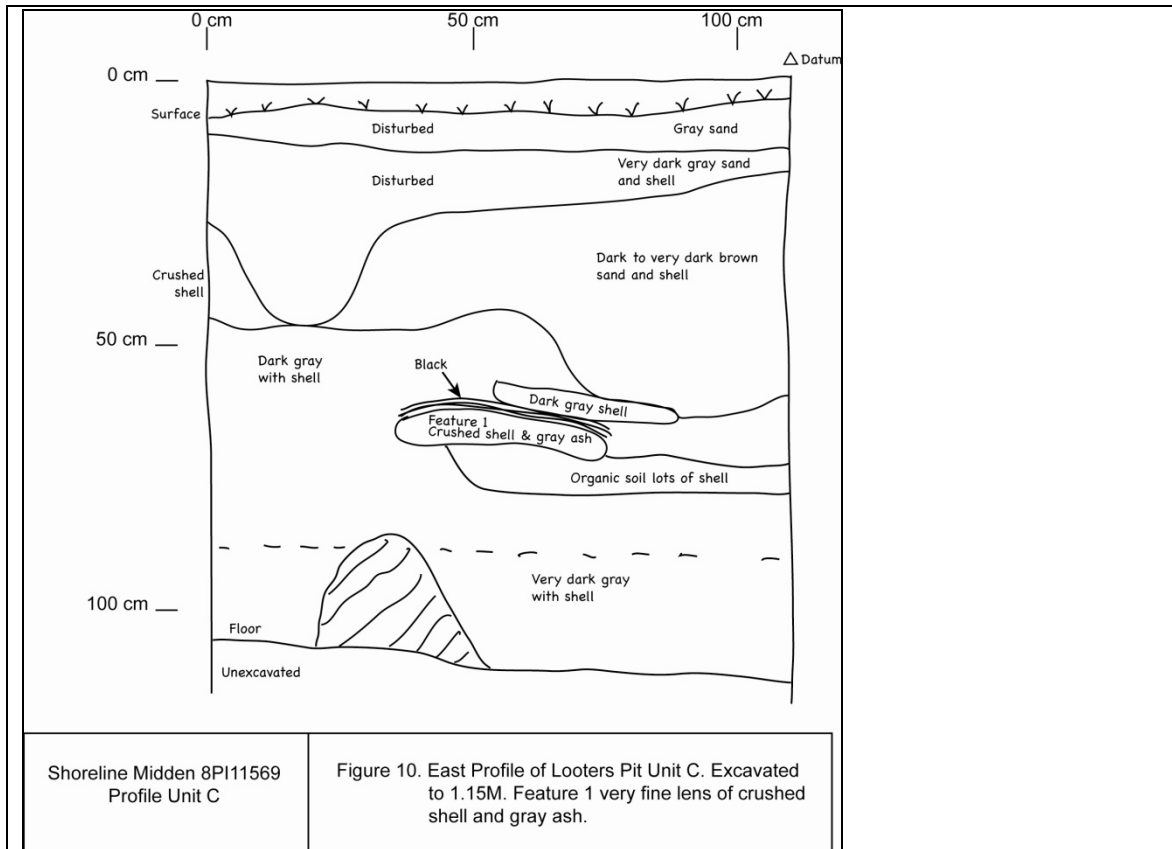


Figure 2. Profile Unit C with Feature 1 (State report: Weedon Island Preserve Gateway Tract Shoreline (8PI11569) Investigation Project).

Materials were separated into various material types. Shell was inspected and any tool use or other modifications were noted. Unworked shell was identified by species. Bone was similarly inspected and then identified as to genus and species (if possible). All material was counted and weighed. As these samples were recovered from previously-disturbed contexts, and the sample sizes were by no means representative, these data were recorded as a simple species list. This list enumerates the variety of resources used by the site's prehistoric inhabitants, but does not attempt to quantify the relative use of any of these food resources.

Pottery was brushed clean of sand, weighed, and the thickness measured. Sherds were separated into standard typological categories after Goggin (1948), Willey (1949), and Cordell (1993). Rim and body sherds were also identified. Given to small number of sherds recovered, vessel

reconstruction was not attempted, but an effort was made to identify the relative size and shape of the vessels represented by the samples.

Historic/modern artifacts were cleaned, re-bagged, and separated into standard categories based on function and material type. The type and provenience of this material were considered of particular importance as they may show the extent of site disturbance and perhaps provide some idea as to how long this site has been the target of intensive looting activities. After analysis, all recovered materials were re-bagged in archivally-stable polyethylene bags and labeled per the Florida Bureau of Archaeological Research's Standards for Curation.

Fauna
Miscellaneous fish bone (<i>Osteichthyes</i> and <i>Sparidae</i>)
Turtle (<i>Testudines</i>)
Deer (<i>Odocoileus virginianus</i>)
Bird (<i>Aves</i>)
Drum (<i>Sciaenidae</i>)
Stone crab (<i>Menippe spp.</i>)
Shark (<i>Chondrichthyes</i>)
Crown conch (<i>Melongena corona</i>)
Lightning whelk (<i>Busycon sinsistrum</i>)
Pear whelk (<i>Busycon spiratus</i>)
Oyster (<i>Crassostrea virginica</i>)
Scallop (<i>Argopecten irradians</i>)
Olive (<i>Olividae spp.</i>)
Southern quahog (<i>Mercenaria campechiensis</i>)

Figure 3. Faunal species collected from Shoreline Midden.

PROJECT DESCRIPTION – SHORELINE CANOE

The Shoreline Canoe (8Pi11624) investigation at Weedon Island Preserve, conducted on December 10, 2007 under the 1A-32 Archaeological Research Permit Number 0708.2, proved to be a significant archaeological find. This one-day investigation was scheduled at a very low tide and all logistical and personnel support was coordinated to accomplish the task in the short window of a few hours. Professional archaeologists assisting with the investigation were Donna Ruhl, University of Florida, Archaeobotanist, Robert Austin, Ph.D. and Justin McNesky, Southeastern Archaeological Research, contractors with Pinellas County for the Gateway Survey, and Phyllis Kolianos, Environmental Education Coordinator from Weedon Island Preserve, Environmental Lands Division.

To reach this site, accessible only by boat, required planning and coordination to ensure that tides and weather were suitable. Logistics of building a coffer dam took several months of research to determine the fastest, cheapest, and best construction to do the job of keeping the tide waters at bay during mean high tide. The actual coffer dam was built of two tiers of sandbags enveloped in heavy vinyl sheeting with occasional wood post supports. It was installed at dead low tide prior to the team's investigation. Two commercial "mud hog" pump systems were in place to evacuate water away from the canoe once exposed through a trench system. The dam system worked

efficiently up until the last minutes of the investigation when the tide breached the dam and allowed the water to enter the area.

The objectives of the canoe investigation were to determine, through precise documentation, the significance of this canoe situated in the intertidal zone soils of Old Tampa Bay. The specific purposes of the investigation were to:

- Measure all dimensions of the canoe: length, width and depth.
- Sample the canoe for wood identification and radiocarbon dating.
- Photograph, video tape, and record all steps of the investigation.
- Research comparative canoes and study possible similarities.
- Publish pertinent information about the canoe for public awareness and stewardship of cultural resources in an urban setting.

FIELD METHODS

Initial Site Reconnaissance—An initial trip to the site with the informant confirmed the location of the canoe and its approximate orientation in October 2006. The informant thought the canoe may have been up to 13 meters in length. Plans were developed to create a coffer dam with two rows of stacked sandbags in rolled heavy plastic vinyl surrounding the intertidal area to the shore. This would prevent tidal water up to a depth of 51 cm. Red and black mangroves dominated the shoreline area, which also supported considerable oyster growth.

Field Investigation—On December 8, 2007, two teams converged on the site during a very low tide. The sandbags had been positioned the previous day to allow for ready construction of the coffer dam. J. Bart McLeod, Management Intern, and two volunteers quickly built the dam, while the second team began to carefully excavate the canoe. Two mud pumps were taken to the site for siphoning off the water as the investigation progressed. The exposure of the entire canoe in a short time period (four hours) was a laborious process. Parts of the canoe were covered in sediment with a heavy oyster growth and red mangrove roots were also heavily encrusted with oysters. Several trenches were dug at 90-degree angles and then parallel at a distance to allow for pooling of water for runoff and siphoning with the pumps. During the excavation, it was observed that remnants of the gunwales left stained areas in the soils.

A diagram was made of the canoe by Robert Austin and Melissa Ayvaz with measurements taken approximately every 25 – 50 cm. The girth of the canoe, 50 cm from the bow, measured at 79 cm and the exterior depth at 17 cm, at 563 cm, the girth measured 75.5 cm and the exterior depth at 17 cm. The length of the canoe from the bow, which oriented west, to the broken jagged end of the stern was 12.17 meters.

Samples for wood identification and radiocarbon dating were taken from various edge areas of the canoe. Sample #1 came from approximately 38 – 42 cm from the bow, Sample #2 came from 73 cm, Sample #3 came from 610 – 623 cm, and Sample #4 came from a broken piece from the edge of canoe that lay in the canoe from 596 – 641 cm. Radiocarbon dating from Beta Analytic Laboratory results were from canoe samples #2 and #3 and yielded the 2-Sigma calibrated result of Cal A D 690 to 1010 (Cal BP 1260 to 940). The wood was identified as pine (*Pinus* spp.) by Donna Ruhl. The wood had some general loss of cell integrity due to previous exposure and natural processes.

Additional Find—Toward the completion of the investigation and while taking girth measurements, an additional associated artifact was discovered. A single pole was lodged under the canoe. The initial consensus of the investigators was that the pole was a tool used by looters to attempt to dislodge the canoe. After further investigation and research of associated poles

found with canoes (McGee 2001), the decision was made to identify the wood and send a sample for radiocarbon dating. The canoe pole was identified also as pine (*Pinus* spp.). Radiocarbon dating results from the canoe pole yielded the 2-Sigma calibrated result of Cal A D 870 to 1030 (Cal BP 1080 to 920).

DISCUSSION

Although existing research from previous investigations and this recent archaeological evidence expands the knowledge of the early people who lived on Old Tampa Bay, there are many unanswered questions. The archaeological record is sparse for most of the sites. The limited data recovery and the investigation of the looter holes at the Shoreline Midden only explored to an estimated one-third of the midden's total depth. Many areas of the site were undisturbed and could give valuable information about the beginning habitation of the site, formation of the land mass, and insight into the climatic prehistorical record of the Old Tampa Bay environments through anthropogenic remains.

The Shoreline Canoe discovery with its raised bow and 12-m length, even with the broken stern end, is the longest prehistoric dugout canoe found in Florida. It is the first open-water canoe found buried in an intertidal salt water environment, giving solid evidence to theories of extensive water transportation, resource procurement, trade and exchange.

These investigations illustrate the types of integrated archaeological/ecological research that are possible in the future. Collaborative research is the goal of a new effort underway at Weedon Island Preserve by the Alliance for Weedon Island Archaeological Research and Education (AWIARE). AWIARE is a recently-established non-profit organization dedicated to furthering integrated research in the Tampa Bay region.

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ABSTRACT

Session 13: Tampa Bay's Archaeology & Anthropology

PRESENTATION

Underwater Tampa Bay and Beyond Submerged Cultural Resources. J. Moates (FL. Public Archaeology Network). Interest in and commitment to Tampa Bay's maritime past has accelerated. A dugout canoe and shoreline middens, interpretation of an historic arrival in 1528, and the first time a local youngster embarks on a near-shore journey aboard a traditional sailing craft helps bring Tampa Bay's maritime heritage to light. With the efforts of regional groups and organizations, more and more of our submerged and locally known maritime-related remains are being documented. *Underwater Tampa Bay and beyond Submerged Cultural Resources* is a sketch of the bay area's maritime connections as seen through its archaeological and historical remains. Since modern water levels emerged five to six thousand years ago, people's continued occupation along the margins of Tampa Bay has generated indelible marks on and off its shorelines. [jmoates@cas.ufl.edu]

ABSTRACT

Session 14: Relaying The Message – Science Communications

PRESENTATION

Can Science and Sound Bites Coexist? Improving Communication Between Scientists, the Public and the Media. N. O'Hara (Tampa Bay Estuary Program). Recent research shows that science literacy is declining in America. In one recent survey, only 59% of those queried knew that dinosaurs and humans did not live at the same time. This problem is compounded by a quantum shift in mass media – the major source of science knowledge for most Americans – brought on by competition for the public's attention in the information-saturated digital age, as well as an economic upheaval in journalism that has seen a dramatic decline in science coverage in general. The good news is that U.S. adults strongly believe that science education is important and, moreover, they hold scientists in very high regard. Capitalizing on this attitude will require the scientific community to embrace new communication tools and techniques, and to strengthen dialogue with the media messengers who deliver information to the public. [nanette@tbep.org]

SYNTHESIS AND NEXT STEPS TOWARDS MAINTAINING PROGRESS

S.T. Cooper, H. Greening, E. Estevez, E. Peebles & G. Henderson

Although researchers, managers and users of Tampa Bay have been about the business of documenting and understanding Tampa Bay's interwoven habitats, species and systems for well over 40 years, each Bay Area Scientific Information Symposium reveals new findings and identifies gaps in our knowledge. BASIS 5 was no exception. The final synthesis session gave us an opportunity to recap what was learned and what we as a Bay research and management community need to focus on for the future.

Cross-cutting topics for further study – i.e., those that emerged in several sessions or that tie multiple research disciplines together – included: Integrating bay circulation and hydrology into ecosystem-based research, understanding transitional tidal tributary ecosystems better, defining functional success for restoration projects, identifying hardbottom habitats in Tampa Bay and determining their importance to the ecosystem, determining the anticipated impacts of future climate change, identifying new funding sources to complete research priorities, and integrating other research disciplines into current assessments (e.g., economists, coastal land use managers/planners, etc.).

The following summary is based on a synthesis of the conference sessions and BASIS 5 participants' discussions and inputs during the final session.

GEOLOGY, SEDIMENTS & CIRCULATION

New information:

- New analyses indicate that Tampa Bay was formed by sinkholes and that it is not a fluvial valley
- A multidisciplinary approach in Safety Harbor, using modern technology, gave a different interpretation than individual investigations had
- Wind patterns and freshwater riverine inputs are major driving forces influencing bay flushing, not just tides
- 3-D bay circulation models offer an improved perspective on storm surge scenarios

Gaps:

- Better understanding the circulation patterns in the upper bays (especially Old Tampa Bay) related to causeway impediments and the net effect on estuarine processes
- Hardbottom areas in the bay are not clearly identified
- Groundwater connections, particularly in Middle Tampa Bay, are not fully understood

Surprises:

- Residential fertilizer is a dominant source of the nitrogen found in recent Safety Harbor fine sediments

WATER QUALITY, PRIMARY PRODUCTIVITY & ECOSYSTEM IMPLICATIONS

New information:

- NO_x concentrations in bay waters decreased substantially from 2002 to 2008
- The quality of light reaching seagrass beds is as important, or even more so, than the quantity
- Hydrologically-normalized TN load estimates take into account annual variation in rainfall when comparing loads between time periods

- Bay phytoplankton primary productivity is half of what it was in 1980, due in large part from nutrient input reductions since that time

Gaps:

- Incorporating light quality considerations into the development of seagrass light requirement targets
- Incorporating CDOM effects into the development of seagrass light requirement targets (e.g., determining CDOM sources, which may be natural, to better define targets)
- Benthic primary production stemming from sources other than seagrasses (e.g., epiphytes, benthic microalgae, etc.)
- Open-water phytoplankton and zooplankton communities of the bay (community dynamics linked to bay circulation and the connection with the Gulf of Mexico). The importance of these communities and their links across the whole estuarine ecosystem are poorly understood
- Ensuring that sound science and research findings drive new regulatory frameworks (e.g., development of numeric nutrient criteria)
- Updating and/or enhancing bay-wide water quality models
- Determining which key water quality parameters are driving serious bay water quality concerns
- Causes of spring/summer phytoplankton blooms in Old Tampa Bay are not understood

Big Points:

- Four studies demonstrated that water quality concerns are centered near the Safety Harbor/Largo area of Old Tampa Bay
- Monitoring of bay water quality needs to better account for periodic, episodic events

SEAGRASSES

New:

- Observation of aerial photographs over time show that longshore sandbars travel across the shallow shelf from shoreward to the bay, and have been sliding off the shallow shelf every three years in the Mariposa Key area of Lower Tampa Bay
- Seagrass mapping continues to improve with new technologies
- 42% of mapped seagrass areas in the bay are persistent
- Annual changes in *Halodule wrightii* coverage are driving bay acreage estimates (Tampa Bay is a *H. wrightii* driven system) – This should be carefully considered in achieving the overall bay seagrass acreage target (38,000 acres)
- A successful *Syringodium filiforme* transplant was documented adjacent to MacDill AFB

Gaps/Questions:

- Do seagrasses stabilize longshore bars? Or do longshore bars provide protection for seagrass establishment and growth?
- Are longshore bars persistent or shifting in other areas of the bay?
- Persistence of seagrass beds needs to be better understood
- What factors are affecting ephemeral seagrass beds in the bay? What is causing *H. wrightii* coverage changes?
- Linking seagrass species information and a better description of seagrass bed quality into aerial surveys and mapping efforts. Are bay segment-specific seagrass species targets needed?
- Has the 1950 baseline seagrass species composition changed in the bay?
- Understanding photosynthetically-useable radiation (i.e. importance of light quality) requirements of seagrass and the effect of epiphytes on light availability/quality to seagrass

- Better understanding of the effects of commercial/port expansion on seagrass resources is needed
- Better understanding of fisheries use/ecosystem functions of seagrass beds broadly classified as patchy/continuous (including bed density/quality, canopy height, and edge effects) is needed
- Standardizing classification of patchy/continuous seagrass beds across disciplines would benefit comparison of results
- Water quality differences between offshore/inshore areas where seagrass occurs should be examined
- Involving the community in bay seagrass assessments (opportunity for public outreach) should be considered

TIDAL TRIBUTARY RESEARCH & MANAGEMENT

New:

- Snook are 2-36 times more abundant in tidal creeks than in adjacent habitats (charismatic species can be most directly translated to policy makers, if these areas are impacted in the future)
- Snook liked creeks regardless of quality
- New technologies were used to track ecosystem processes stemming from tidal tributary systems (N isotopes / Otolith microchemistry)
- Benthic microalgae pathways have been identified as the base of the tidal tributary food web

Gaps/Questions:

- Contribution of tidal tributaries to overall bay fish production. Are tidal tributaries sources or sinks?
- Better understanding the hydrology of small tidal systems. How is flashiness affecting estuarine processes?
- Developing an overarching management strategy that encompasses the entire estuarine system
- Characterizing the more than 100+ tidal tributaries to better identify trends and indices
- Sampling methodologies may be missing other charismatic species which could also be used to campaign for the protection of these systems to policy makers [e.g. tarpon, higher trophic orders (birds, amphibians, reptiles, etc.)]
- Sediment quality of small tidal tributaries relative to the open bay environment
- Developing water quality targets/requirements which support the ecosystem function of these systems. This may lead to a better integration of current research findings with regulations and management affecting these systems. These systems could serve as a model for development of intermediate water quality criteria (i.e. transitional between freshwater and marine criteria).
- Identifying effective tidal creek restoration opportunities. Many options to be considered and researched (e.g., changing geomorphology, hydrology, habitat composition, etc.)
- Identifying opportunities for public involvement/volunteers in the rapid assessment of these systems. However, caution should be taken for degraded systems where human health impacts may be a concern

Surprises:

- Tidal tributaries, regardless of their perceived alteration, appear to provide similar fish production/use

ECOSYSTEM RESTORATION

New:

- The 2010 Tampa Bay Habitat Master Plan Update has been completed, including a Habitat Restoration & Land Acquisition Work Plan
- An easily-accessible Habitat Restoration Database has been developed, and is available on the Tampa Bay Water Atlas
- Emerging 3-D imagery technology for habitat monitoring and assessment is promising

Gaps:

- Publications on the evolution (including trial and error results) of habitat restoration/creation techniques in Tampa Bay and other estuaries is lacking
- Integrating/considering open water column production/larval recruitment/dynamics in the Bay's habitat restoration goals ("Restoring the Balance") should be considered
- Monitoring and reporting on functional success of restoration/creation projects is needed
- Shifting the focus of restoration opportunities towards the upper watershed (including freshwater systems) that will be minimally affected by anticipated future coastal impacts from climate change should be incorporated into future habitat restoration planning processes
- A better understanding of the conversions occurring in the coastal uplands and important coastal habitats in the upper watershed relative to estuarine habitats identified in the "Restoring the Balance" concept is needed

Surprises:

- Unwillingness of regulators to accept advancements in habitat creation techniques should be recognized and addressed

Big Points:

- Acquire and protect habitat now; manage/restore later; use various means (investigating new strategies, public/private partnerships)
- Define future reality, and determine what steps are needed to reach it (100 -200 years from now)
- Incorporate new climate change forecasts into restoration project plans

BIOTA

New:

- Over-wintering manatees feed on seagrass beds at night. Rays have also been found to contribute to seagrass impacts
- New research revealed that the probability of manatee detection was only 58%, possibly resulting in a dramatic undercounting
- Existing pipeline projects in the bay could provide insight on hard-bottom habitats and mitigation strategies employed by those projects
- Benthic invertebrate sampling is limited to sandy and soft sediments due to sampling methods, and may be under-sampling hard-bottom habitats. New research at FWRI/FIM may address monitoring gaps in hardbottom habitats and other live-bottom habitats in the Bay

Gaps/Questions:

- Do surveys of scallop populations in the Bay take into account transplanted/introduced scallops?
- Are existing surveys conducted using methods which are appropriate to monitor changes in bay scallop populations?

- Do fisheries landings data relate at all to overall Bay quality?
- What is the effect of routine, repetitive sampling programs on specific seagrass beds and infauna?
- Identifying ways to better protect freshwater wetlands (e.g., for water birds, etc.) should be considered.
- Hardbottom habitats and biota are still poorly understood. There is a need for development of monitoring programs to address potential future impacts from in-bay activities, especially in the context of mitigating for future impacts.
- There is a need to define the function of natural hardbottom areas for mitigation determinations
- Caution should be taken to advertise hardbottom areas to prevent exploitation.
- Faunal communities associated with rocks at the mouths of large tidal rivers are not fully described or generally included in discussions of biota of Tampa Bay

Surprises:

- Benthic invertebrates: Seven taxa make up 25% of the overall Bay abundance and 240 species, in total, have been identified from the Tampa Bay Benthic Monitoring Program
- Freshwater influences (e.g., salinity ranges and related water quality factors) affect seagrass productivity, and are related to nekton abundance in the estuary

WATERSHED MANAGEMENT & REGULATION

New:

- New collaborative bay management decisions are effective. Examples include the nitrogen management strategy developed and implemented by the Tampa Bay Nitrogen Management Consortium and the adoption of a Basin Management Action Plan (BMAP) for bacteriological impairment in the Hillsborough River
- A Decision Matrix evaluation of bacterial contamination has been developed for the Hillsborough River BMAP assessment
- ACOE/EPA regulators can now direct watershed mitigation towards regional planning goals, as defined in adopted watershed management plans. There is a need to clearly identify the information needed by these regulators to support this process
- SWFWMD environmental resource permitting now requires a demonstration of “Net Improvement” for redeveloped and new development in the Tampa Bay watershed. This will assist in meeting the EPA’s TMDL and FDEP’s Reasonable Assurance allocations for nitrogen in Tampa Bay because additional nitrogen loads will need to be offset

Gaps:

- Addressing sea level rise in land use planning is needed (e.g., improving FEMA floodplain maps, integrating One Bay Initiative findings into local ordinances)
- Permit monitoring and reporting for small/mid-level industrial facilities is severely lacking
- Assessing the effectiveness of restoring freshwater wetland function through mitigation activities
- There is a need to link emerging research and scientific findings with the regulatory process (e.g., wetland mitigation, anticipated sea level rise and its effect on future land use zoning scenarios, etc.)
- There is a need to creatively explore mitigation options that support the whole ecosystem and watershed (e.g. selectively focus land acquisition, restoration, creation opportunities etc.). It should be recognized that functional mitigation projects are the objective and end result of the regulatory process

Surprise:

- The lack of reporting requirements for Multi-Sector General Permits and Regional General Permits

CLIMATE CHANGE**New:**

- New estimates by the International Panel on Climate Change indicate a potential for 1+m rise in sea level by 2100
- Perspective and understanding of Tampa Bay's climate history has vastly improved

Gaps/Questions:

- What has been the effect of sea level rise on the Bay's depth? Is the Bay getting deeper?
- What is the potential extent of seagrass bed migration in response to sea level changes (movement of beds along the deep and shallow edge)?
- There is a need to translate climate change science and potential effects of climate change on coastal habitats and communities to local land use decision-makers
- Potential sea level rise effects on sensitive biota / aggressively colonizing non-indigenous species is not known
- Implications to shallow-water respiration and dissolved oxygen dynamics in the bay due to increasing water temperatures and/or depths
- Changes in rain, storm-intensity, and other general weather conditions and patterns for the region are not well known in the face of climate change

Surprises:

- Rapid sea level rise is possible, if ice sheets continue to break up in upper and lower latitudes
- The United Kingdom is using a 4-m sea level rise for infrastructure planning purposes

ANTHROPOLOGY**New:**

- Improvements in the understanding of the Bay's past history, both in terms of archaeological resources and cultural anthropology have occurred over the past 10 years

OTHER ISSUES**New:**

- Plant species on spoil islands, sand bars, and shell middens occurring in Tampa Bay have been documented
- Lack of funding focused towards environmental monitoring programs in the Bay may lead to a lack of data needed to measure progress towards goals and the ability to assess critical resources over time. Economic conditions have caused severe cut-backs in local natural resource management departments

Gaps:

- A better understanding of alternative nutrient best management practices is needed (e.g., stormwater harvesting, benefits of urban tree canopies, etc.)

CROSS-CUTTING THEMES**Commonalities:**

- Bay circulation and hydrology need to be better understood in terms of the effects on Bay ecology (e.g., water quality, flushing rates/time, algal bloom formation, and larval recruitment to both restored/natural systems, etc.).

- More research is needed on tidal tributaries (e.g., better understanding their hydrology (flashiness), geomorphology, restoration potential, and overall connectivity with in-situ Bay production, etc.).
- Some measure of functional success for restoration projects needs to be defined (both above and below the waterline, if applicable).
- Hardbottom habitats (in rivers and the bay proper) are poorly understood.
- New research findings and scientific knowledge need to be more effectively integrated into bay management and regulatory frameworks. This includes anticipated climate change effects on coastal habitats and communities and the resulting land use policies and regulations that need to be updated to reflect new sea level rise scenarios.
- Understanding potential impacts related to climate change other than sea level rise (e.g., water supply impacts, regional weather pattern changes, impacts on stormwater transport systems, etc.).
- Additional research is needed across several disciplines, as described in each preceding “Gaps” section.
- Identifying new sources of funding to address research “Gaps” will be crucial in the future. For example, research on open-water zooplankton and larvae communities could be sought as part of permit requirements for Bay dredging projects.
- Multi-entity/multidisciplinary research projects have provided beneficial insight and integrated research results for Tampa Bay. Developing a system-wide conceptual framework may add additional value by better prioritizing where future multidisciplinary projects should be focused.
- As such, other disciplines should and can be integrated into future research assessments (e.g., economic and social scientist evaluations, land use policy/planning staff, etc.). This could lead to additional stream-lining and efficiencies for integrating results of research assessments with regulatory policy and frameworks. In addition, multidisciplinary research approaches will be important for potential future funding related to large-scale projects (e.g., Bay channel dredging, large-developments in the watershed, cumulative impact assessments, etc.)
- Tampa Bay should and can be used as a model in developing multidisciplinary assessments that are effectively integrated into bay management, policy decisions, and regulatory frameworks (especially for collaborations among local/ state/federal levels).
- The EPA Tampa Bay Ecosystem Services Demonstration project can be used as a decision-making tool in the future.

Gaps:

- Re-evaluating existing science and management paradigms is a critical element of the adaptive management approach utilized for Tampa Bay. The paradigms should periodically be assessed and reported on a routine basis.
- Improving our understanding on the potential for future impacts to overwhelm or exacerbate the Tampa Bay ecosystem is crucial. Ongoing updates to our understanding of the cumulative impacts to Tampa Bay will be needed to determine the sustainability of the Bay to future impacts and provide a measure of uncertainty towards this ultimate determination.
- Evaluating whether new/innovative status and trends measures should be established for Tampa Bay.

IMPROVEMENTS TO CONSIDER FOR BASIS 6

- Session arrangements for future Symposia should consider cross-cutting issues at the beginning of the conference.
- Encourage more student and next generation researcher involvement with BASIS 6.
- Effectively transferring the lessons learned at Symposia to policy makers, senior agency

- officials, and bay managers is a high priority. (TBEP staff will develop a four-page newsletter of the BASIS 5 meeting synthesis)
- Effectively communicating BASIS findings to the media. Potentially develop a half-day event focused on transferring the science to policy makers and media outlets. An Agency on Bay Management/Regional Planning Council meeting occurring after future Symposia could be the appropriate forum to accomplish this communication from BASIS.
 - Identifying ways to effectively transfer future Symposia research findings and management actions to State policies and regulations.