Tampa Bay Estuary Program
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SEAGRASS MANAGEMENT: IT'S NOT JUST NUTRIENTS!

Proceedings of a Symposium

St. Petersburg, Florida

H.S. Greening, Editor

FINAL REPORT

August 22-24, 2000

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editor

Foreword and Acknowledgments

These Proceedings contain presentations given at a symposium held August 22–24, 2000, in St. Petersburg, Florida, entitled “Seagrass Management: It’s Not Just Nutrients!” The symposium was held to review recent advances in seagrass research and management, and to define priority issues for effective management of seagrasses in Florida and the Gulf of Mexico. More than 150 seagrass scientists and managers attended the three-day symposium. Each manuscript in these Proceedings has been reviewed by three peers and by Holly S. Greening (editor).

In addition to the input and direction provided by the Steering Committee for this Symposium, we gratefully acknowledge the wisdom provided by Rick Batiuk (Chesapeake Bay Program) and Ken Moore (Virginia Institute of Marine Science) for their summary and critique of each day’s session.

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Proceedings

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EXECUTIVE SUMMARY
Seagrass Management: It's Not Just Nutrients!
August 24, 2000

Holly Greening

Background

In Tampa Bay, an increase in seagrass areal extent has been observed since 1988, due to improvements in water quality as a result of significant reductions in nitrogen loads starting in the early 1980s. However, recent aerial photographs and mapping have indicated that, in some areas of the bay, seagrass recovery has not been as rapid as in others. In fact, apparent seagrass loss may be occurring since 1994 in limited areas, even though water quality and clarity appears to be adequate to support seagrass growth in these areas.

The purpose of the three-day symposium was to examine the status and trends of seagrass and water quality in several Florida estuaries, and then to examine factors, including water quality and others, that may be impacting recovery of seagrass in our coastal systems.

The Tampa Bay Seagrass Management Workshop Steering Committee emphasizes that nitrogen management will remain the primary driver in seagrass recovery in Tampa Bay, but that other factors also appear to be affecting seagrass recovery rates in some areas of the bay.

Symposium Overview

Day 1 of the three-day symposium was focused on examining status and trends of seagrass, water quality and management activities in Tampa Bay, Sarasota Bay, Lemon Bay, Charlotte Harbor, Florida Bay, Indian River Lagoon and other coastal areas. The second day was dedicated to examining emerging issues relating to seagrass management in all these systems. The papers in this Proceedings are based on the Symposium presentations.

Tampa Bay Seagrass Management Workshop

The objective of the Tampa Bay Seagrass Management Workshop, held on the third day of the Symposium, was to develop a Plan of Study for further evaluation of issues other than nitrogen management which may be impacting seagrass recovery in Tampa Bay. A list of workshop participants is attached.

Following presentations and discussion throughout the three day workshop, participants were asked to rank the 19 issues which were identified during the Workshop. Each participant and audience member was given five "votes", and the number of votes for each priority issue was tallied.

These priority issues and the recommended steps to address them generally fall into three categories:

- Technical
- Public Involvement
- Management

As of the time of preparation of these Proceedings (December 2001), significant action had been initiated on 15 of the 19 Priority Issues.

OVERALL RECOMMENDATION:

Form a *Seagrass Working Group* to develop the issues and options listed here; review data and information as it becomes available; and promote exchange of information.

Action initiated: The Tampa Bay Seagrass Working Group was formed in September 2000, immediately following the workshop. As of December 2001, more than 30 seagrass researchers and resource managers are active participants in the Working

Group. Actions initiated by this group for each of the Priority Issues are noted below.

Ranked Priority Issues

1. Identify causes of seagrass recovery slowdown or seagrass loss in "problem areas".
Action initiated: The Seagrass Working Group has identified four "target" or problem areas, and two "reference" areas where seagrasses have remained stable or are increasing.

Target areas are:
1. Feather Sound (north and south of the Howard Franklin Bridge causeway in western Old Tampa Bay);
2. Coffeepot Bayou in St. Petersburg, where seagrasses appear to be thinning on the interior edge while remaining stable at the deep edge;
3. The Kitchen, in eastern Hillsborough Bay; and
4. Wolf Branch, along the eastern shore of Middle Tampa Bay south of Apollo Beach.

Reference areas are:
1. Pinellas Point, western shore of Middle Tampa Bay; and
2. Bishop's Harbor, eastern side of Lower Tampa Bay just north of the Skyway Bridge causeway.

The Working Group is initiating intensive monitoring and experimental work in the target and reference areas (focusing in Old Tampa Bay) in January 2002, with funds from a Pinellas County Environmental Foundation grant awarded in October 2001. The US Geological Survey, Southwest Florida Water Management District, Florida Marine Research Institute, Pinellas County, City of Tampa, Environmental Protection Commission of Hillsborough County, and Tampa Bay Estuary Program are all providing support as cash or in-kind work. The intensive work will extend 18 months.

2. Initiate and support work to better understand seagrass ecology and biology.
Action initiated: A proposal to the Gulf of Mexico Program to update the Ecology of Seagrasses on Florida's West Coast was awarded to the Working Group members in fall 2001. The one-year project will be initiated in January 2002. Results of this literature review and synthesis will be produced in hard copy and as a web-based report.

3. Scan and provide easy electronic access for historic and existing seagrass maps and photos. Develop a photo archive to catalog where photos and maps are stored.
Action initiated: This project has been funded by the Gulf of Mexico Program, with partners including US Geological Survey, Florida Marine Research Institute, the Southwest Florida Water Management District, and Tampa Bay Estuary Program. The photo archive will be housed and maintained at the Florida Marine Research Institute.

4. Enhance boater impacts management, including public education/outreach and stakeholder involvement.
Action initiated: The Tampa Bay Regional Planning Council's Agency on Bay Management education subcommittee has adopted this issue for development and implementation of an education/outreach/involvement strategy.

5. Assess seagrass requirements, including epiphyte attenuation, light requirements for all species, seasonal effects, and impacts of macroalgae and microfauna.
Action initiated: The Florida Marine Research Institute conducted epiphyte loading measurements throughout Tampa Bay in fall 2000, working with a college science class at Eckerd College. Samples were collected from permanent transect

locations, and data will be provided for inclusion in the transect database. Seagrass requirements will be measured during the 18-month intensive study outlined in Priority #1.

6. Examine effects of changes in offshore bars, ship wakes and wave energy.
Action initiated: The National Oceanic and Atmospheric Administration is applying an existing wave energy model in Tampa Bay for the Tampa Bay Estuary Program. NOAA's modeling work will be complete in early 2002, and will include wave energy assessment scenarios with and without bars in the model. The model will focus on the "target areas" and reference areas identified by the Working Group. The Seagrass Working Group will act as reviewers for the project.

7. Evaluate seagrass planting techniques.
Actions initiated: The Florida Marine Research Institute has been funded by the State of Florida to evaluate the effectiveness of various seagrass planting techniques, including mechanical planting techniques and hand planting. Work is expected to be complete by summer 2003.

8. Formalize the transect monitoring program.
Actions initiated: The Tampa Bay seagrass transect monitoring program completed the third year of data collection in Fall, 2001. This combined field sampling effort is conducted by Hillsborough, Pinellas and Manatee counties, the City of Tampa, Florida Marine Research Institute, the Southwest Florida Water Management District, and Tampa BayWatch. Data collected are collated and basic analyses conducted by the City of Tampa Bay Study Group.

9. Establish permanent intensive "sentinel sites" within seagrass beds for research and monitoring.
Action initiated: The seagrass transect monitoring could provide a basis for intensive sentinel sites; however, this action has not yet been formally addressed.

10. Examine and monitor the effects of extreme events on seagrass recovery.
Action initiated: None to date.

11. Further investigate *Labyrinthula* impacts and implications for seagrass recovery.
Action initiated: The Florida Marine Research Institute has analyzed seagrass samples collected by the transect monitoring program for the second year. *Labyrinthula* is present in almost all of the transects, but does not appear to be associated with degraded seagrass condition at this time.

12. Obtain more accurate bathymetry.
Action initiated: The US Geological Survey has initiated intensive geological, subsurface and benthic production research in Tampa Bay, as a pilot for their National Estuaries Assessment Project. Shallow water bathymetry using several methods is being conducted as part of this project.

13. Provide more emphasis on shallow water monitoring.
Action initiated: The intensive seagrass and water quality monitoring programs designed for the "problem areas" study (Priority #1) will include shallow water monitoring in Old Tampa Bay.

14. Consider ecological implications of seagrass fragmentation.
Action initiated: The update of the seagrass ecology synthesis (Priority #2) will include literature review of seagrass fragmentation.

15. Develop a structured synthesis/ storyline of information about Tampa Bay.

Action initiated: A formal synthesis is not planned at this time. However, the fourth Bay Area Scientific Information Symposium (BASIS 4), in which science and management results from all disciplines addressing Tampa Bay are presented, is being planned for December 2002.

16. Investigate seagrass "halos" near discharges.
Action initiated: None to date.

17. Map deep edges of seagrasses.
Action initiated: J.O.R. Johansson and W. Avery have developed and applied a method using GPS positioning for accurately mapping the deep edges of seagrasses. See their paper in this volume.

18. Determine the accuracy and precision of historic and current maps.
Action initiated: The US Geological Survey intensive Tampa Bay study will be examining accuracy and precision issues in mapping seagrasses and other habitats of Tampa Bay. Their work is due to be completed in 2003.

19. Develop a definition of the "deep edge" of seagrass beds.
Action initiated: See Virnstein et al. in this volume.

(HG) Tampa Bay Estuary Program, MS I1/NEP, 100 8th Ave. SE, St. Petersburg, FL 33701

Tampa Bay Seagrass Management Workshop Invited Participants

NOTE: All Symposium participants were also invited to attend the Tampa Bay Workshop

Tampa Bay Seagrass Management Workshop Steering Committee	
Walt Avery	City of Tampa Bay Study Group
Kellie Dixon	Mote Marine Laboratory
Holly Greening	Tampa Bay Estuary Program
Penny Hall	FWC Florida Marine Research Institute
Roger Johansson	City of Tampa Bay Study Group
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Jake Stowers	Pinellas County Department of Environmental Management
Steve Wolfe	Florida Department of Environmental Protection
Seagrass Science and Management "Outside Experts"	
Rich Batiuk	EPA Chesapeake Bay Program
Mark Fonseca	NOAA/NOS Center for Coastal Fisheries and Habitat Research
Jimmy Johnson	USGS National Wetlands Research Center
Ken Moore	Virginia Institute of Marine Science
Fred Short	Jackson Estuarine Laboratory
Bob Virnstein	St. Johns River Water Management District

HISTORICAL OVERVIEW OF TAMPA BAY WATER QUALITY AND SEAGRASS: ISSUES AND TRENDS

J.O.R. Johansson

ABSTRACT

Historical (pre-1930s) seagrass meadows in Tampa Bay are believed to have covered 31,000 ha of the shallow bay bottom. Later, impacts to the bay from increasing population and industrial development of the Tampa Bay area have resulted in large seagrass losses. By 1982, approximately 8,800 ha of seagrass remained. Since 1982, Tampa Bay seagrass monitoring programs have recorded a reversal in the trend of seagrass loss, with the baywide seagrass cover increasing from 9,420 ha in 1988 to 10,890 ha in 1997. However, between 1997 and 1999 the trend again reversed with substantial losses recorded, specifically in the Old Tampa Bay segment. The 1999 baywide cover was estimated at 10,050 ha, thus eliminating most of the seagrass coverage gains recorded since the late 1980s.

Similarly, in Hillsborough Bay, the segment of Tampa Bay that historically has had the poorest water quality, seagrass increased from near 0 ha in 1984 to about 56 ha in 1997. Following 1997, the seagrass expansion stagnated in this segment, with a slight reduction in cover recorded between 1998 and 1999.

It is generally agreed that the Tampa Bay seagrass expansion observed since the mid-1980s was triggered by water quality improvements during the late 1970s to the mid 1980s. These improvements included reductions in phytoplankton biomass and water column light attenuation. These improvements also followed a nearly 50% reduction in external nitrogen loading from domestic and industrial point-sources in the early 1980s. The loading reductions primarily affected point-source discharges to Hillsborough Bay.

The reductions in the Tampa Bay seagrass expansion rate and areal cover realized since 1997, are most probably related to a recent period of high rainfall that began in 1995 and extended through the strong 1997–98 El Niño event. During this period, nitrogen loading and discharges of water with high color content increased, and subsequently, phytoplankton biomass, light attenuation, and color content increased in all major bay segments. High levels of these constituents are known to be detrimental to seagrass growth and it is not surprising that Tampa Bay seagrass monitoring programs recorded recent reductions in expansion and losses of seagrass cover, specifically in the upper bay segments. Although other factors may have contributed to the recent reductions in seagrass growth, it is likely that the high rainfall period created the major impacts.

Rainfall in the Tampa Bay area was below normal in 1999 and 2000. With an extended period of relatively low rainfall it could be expected that the seagrass expansion rate again would increase. Baywide information for seagrass coverage in 2000 is not available; however, Hillsborough Bay seagrass coverage increased from about 56 ha in 1999 to about 69 ha in 2000.

INTRODUCTION

Seagrass has been selected as a central component in many estuarine management efforts that aim to restore a natural balance between primary producers (e.g., seagrass and phytoplankton) by reducing excessive nutrient inputs. This management approach has been attempted for Tampa Bay as well. The Tampa Bay Estuary Program (TBEP) has adopted a seagrass restoration and protection goal to be reached through the reduction of external nitrogen loadings to the bay (Johansson and Greening 2000). To place the ongoing Tampa Bay seagrass trends and management efforts in a historical perspective, it is necessary to understand the history of both seagrass coverage and water quality, including the results of efforts to improve bay water quality. Therefore, this report will discuss the major changes that have occurred in seagrass abundance since the earliest estimate of Tampa Bay seagrass coverage. Some of the most likely causes contributing to those changes will also be discussed.

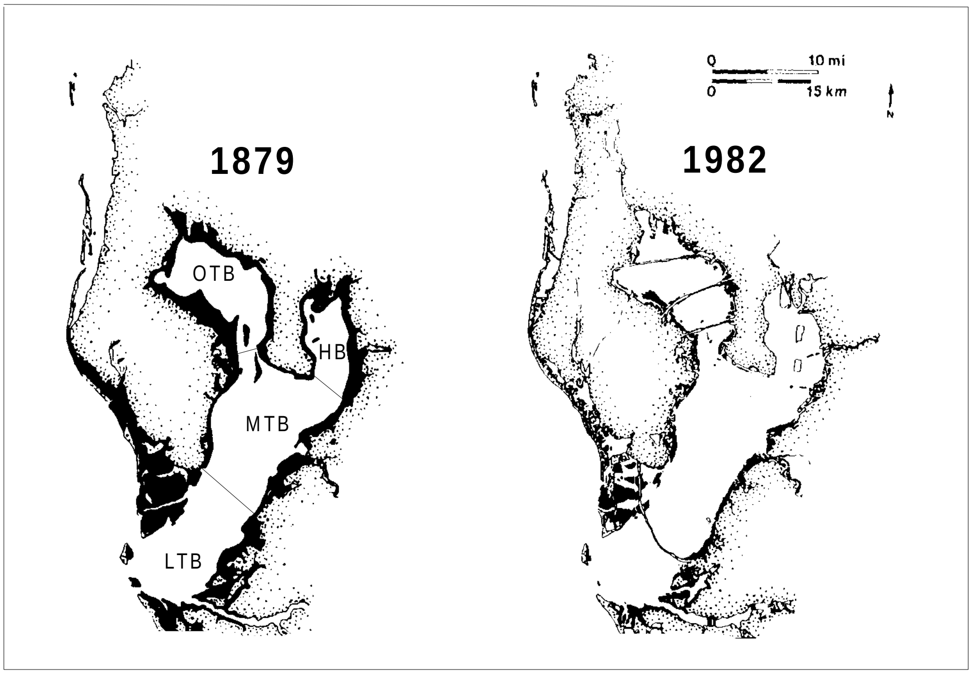

Figure 1. Seagrass coverage in Tampa Bay in 1879 and 1982 (after Lewis et al. 1985). Black areas denote seagrass meadows. OTB=Old Tampa Bay; MTB= Middle Tampa Bay; LTB=Lower Tampa Bay.

In addition to the historical perspective, an up-to-date account of current water quality and seagrass trends in the bay will also be given. Attention will be focused on Hillsborough Bay because this bay segment has the most extensive water quality record and the most detailed information on the recent seagrass recolonization. Long-term trends of selected water quality parameters for all four major Tampa Bay segments will also be examined in addition to long-term trends in the baywide seagrass cover.

HISTORICAL TRENDS

In a comprehensive paper on Tampa Bay seagrass, Lewis et al. (1985) estimated that approximately 31,000 ha of seagrass were present in Tampa Bay during the late 1800s (Fig. 1). Their estimate included areas that were sufficiently shallow (<2m) to support seagrass growth and was not based on measured seagrass coverage. Therefore, this estimate represents the potential Tampa Bay seagrass coverage at a time when man's influence on the bay was very limited.

In 1991, Lewis et al. calculated the distribution of the 1982 Tampa Bay seagrass cover. This estimate was based on state-of-the art analysis of aerial photographs and showed that the 1982 coverage was about 8,800 ha (Fig. 1). When compared against the late 1800s estimate, the seagrass had apparently receded in all segments of the bay over the near 100-year period with major losses evident in the upper bay portions of Old Tampa Bay and Hillsborough Bay. Specifically, all seagrass appears to have been lost in Hillsborough Bay by 1982.

The cause of these large seagrass losses, perhaps as much as 70% of the historical Tampa Bay seagrass cover, is undoubtably related to man's impact on the bay. One major impact was the excessive loading of nutrients from the watershed, or eutrophication. This impact is directly related to the population growth of the bay area and the associated increase in commercial activities. Eutrophication as indicated by water column chlorophyll *a* concentrations (Fig. 2) peaked in the late 1970s and early 1980s (Johansson 1991; Johansson and Lewis 1992; Boler 1999). Another leading cause of large seagrass loss was various dredging operations and shoreline developments. These included in-bay shell dredging, port construction, ship channel expansion, causeway construction, and residential and commercial dredge-and-fill projects. Impacts from these activities culminated during the 1950s, '60s and '70s.

Tampa Bay researchers generally agree that eutrophication and dredging operations were the major reasons for the large seagrass loss, although it is unclear which of these impacts was most serious. Also, questions remain about the process of seagrass loss. For example, did dredging operations cause losses mainly through direct physical destruction of the seagrass meadows, or through more indirect impacts such as increased turbidity of the water column and increased sediment deposition on the meadows? It is also unclear how eutrophication caused seagrass losses. It is generally assumed that eutrophication associated losses resulted from a decrease in light availability, which in turn was caused by an increase in phytoplankton and epiphyte biomass. It is also well known, particularly in Hillsborough Bay (FWPCA 1969; Kelly 1995; Avery 1997) and Old Tampa Bay (J.O.R. Johansson personal observations), that the increased nutrient loading stimulated the growth of large amounts of drift macro-algae. Dense mats of these algae often accumulated in the shallow areas and may have limited seagrass colonization through shading, abrasion, and hypoxia.

A large reduction in nitrogen loading to Tampa Bay occurred between the late

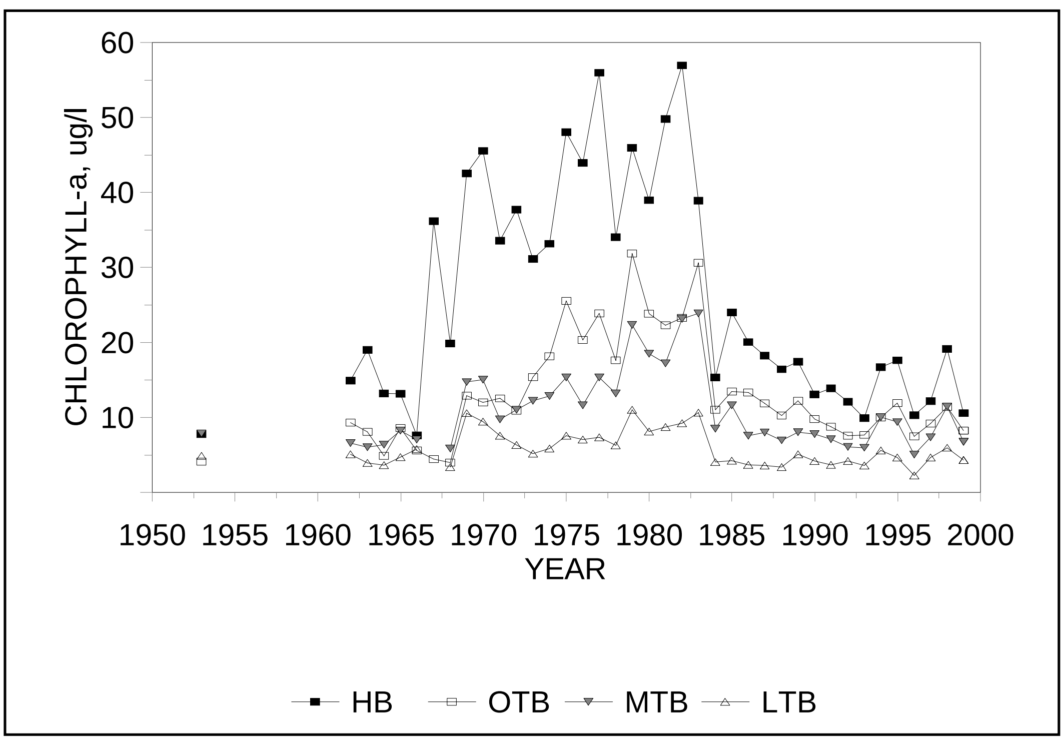

Figure 2. Annual average chlorophyll *a* concentrations for major segments of Tampa Bay, 1953–99 (sources include National Marine Fisheries Service, Hillsborough County Environmental Protection Commission and City of Tampa).

1970s and the early 1980s (Fig. 3). This reduction was primarily caused by improved wastewater treatment from domestic and industrial point-sources that discharged to Hillsborough Bay (Johansson 1991; Johansson and Lewis 1992; Zarbock et al. 1994). A large decrease in phytoplankton biomass (chlorophyll *a*) soon followed the nitrogen reduction. By 1984, chlorophyll *a* concentrations were about half of the levels found only a few years earlier (see Fig. 2). Coincident with declining chlorophyll *a* concentrations in Hillsborough Bay, small isolated patches of *Halodule wrightii* (shoal grass) began appearing in a shallow area of southeastern Hillsborough Bay that previously lacked seagrass vegetation (R.R. Lewis personal communication).

RECENT SEAGRASS TRENDS

Seagrass coverage in Hillsborough Bay, has been estimated by the City of Tampa (COT) since 1984. In 1984, coverage was limited to a few isolated patches of *H. wrightii*, comprising a total of less than 0.02 ha. Since then, each successive survey, until 1998, showed a substantial increase in *H. wrightii* cover (Fig. 4). By 1997 seagrass cover had reached about 55.6 ha. After 1997, the seagrass expansion stagnated in this segment with a slight reduction in cover recorded between 1998 and 1999. The 1999 Hillsborough Bay coverage was estimated at about 56.2 ha. The rate of expansion once again increased between the 1999 and 2000 surveys. The 2000 coverage was estimated at 69 ha.

Similarly, the Surface Water Improvement and Management (SWIM) program of the Southwest Florida Water Management District (SWFWMD) has estimated the baywide Tampa Bay seagrass coverage by interpretation of aerial photography. SWIM has shown that seagrass coverage increased from 9,420 ha in 1988 to 10,890 ha in 1997 (Fig. 5). The trend of expanding coverage reversed in 1999 with substantial losses recorded, especially in the Old Tampa Bay segment. The 1999 baywide cover was estimated at 10,050 ha, thus eliminating most of the gains recorded since 1988. Figure 5 shows the current Tampa Bay seagrass coverage relative to both the historical coverage and the restoration goal adopted by the TBEP.

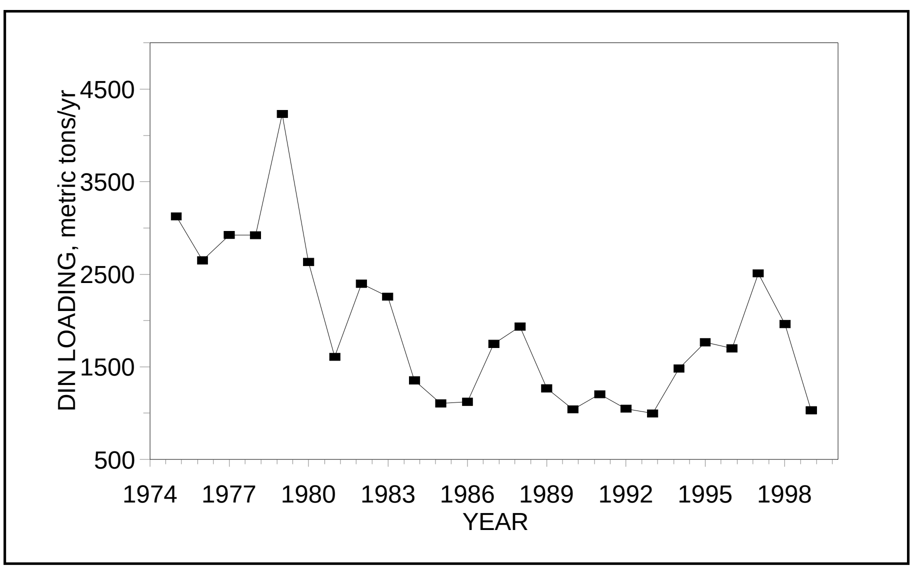

Figure 3. Dissolved inorganic nitrogen loading to Hillsborough Bay from major external sources, 1974–1999. Loadings in 1997, 1998 and 1999 were estimated from rainfall amounts. Flow and nitrogen concentration data were not available for these years.

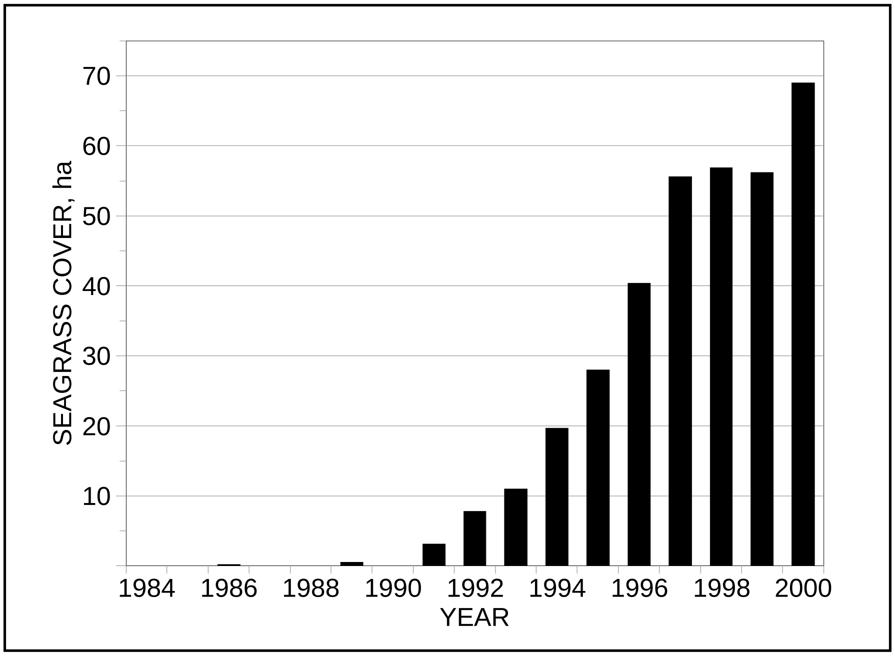

Figure 4. Hillsborough Bay seagrass coverage estimated by the City of Tampa, 1984–2000. The estimate for 1984 is exclusively based on interpretation of aerial photographs. No estimates were performed for 1985, 1987, 1988 and 1990.

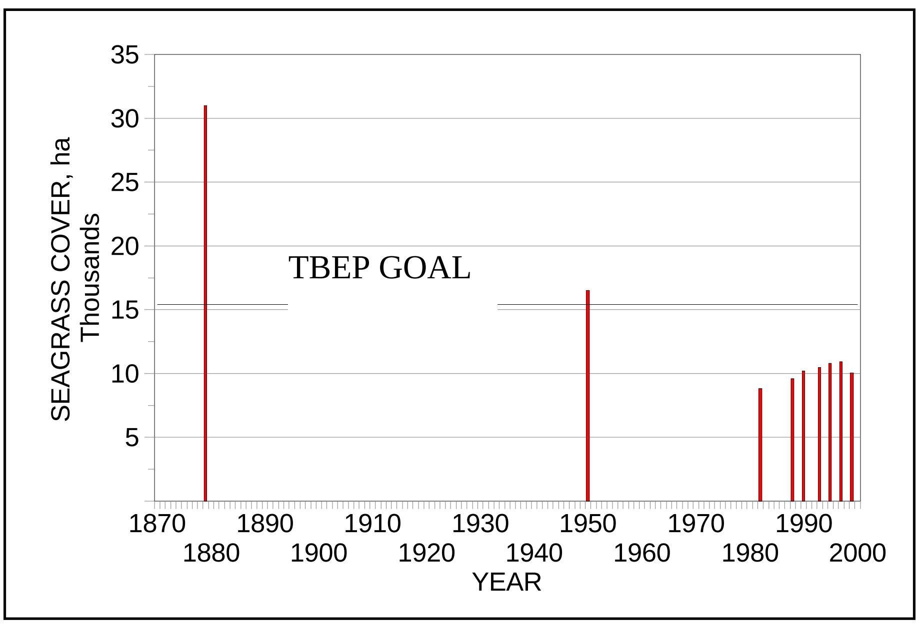

Figure 5. Long-term trend of Tampa Bay seagrass coverage, 1879–1999 (sources include Lewis et al. 1985; Lewis et al. 1991; Johansson and Ries 1997; and Kurz, this volume). The horizontal line marked TBEP GOAL denotes the Tampa Bay Estuary Program's seagrass restoration and protection goal.

SEAGRASS AND WATER QUALITY RELATIONSHIPS

The detailed seagrass information collected by the COT in Hillsborough Bay can be used to search for potential relationships between seagrass expansion and water quality trends. Year-to-year variations in Hillsborough Bay chlorophyll *a* concentrations, which indicate changes in the trophic state of the bay and also the amount of nitrogen being discharged to the bay, are not reflected in the annual trend of Hillsborough Bay seagrass coverage (Fig. 6). Such short-term relationships between the relatively slow process of changes in seagrass coverage trends and more variable water quality parameters should not be expected. Instead, the expansion of seagrass that started in the mid 1980s in Hillsborough Bay, and in other sections of Tampa Bay as well, probably resulted from the large decrease in eutrophic state that occurred in the early 1980s. The trophic state declines are reflected in the Hillsborough Bay chlorophyll *a* and Secchi depths records (Fig. 6 and 7).

Higher than normal rainfall amounts (measured at Tampa International Airport) during the years 1995, 1996, and the 1997–98 El Nino event (Fig. 8) have increased nitrogen loading to the bay. Phytoplankton biomass, light attenuation, and color content increased in Hillsborough Bay and the other major bay segments during this period of high rainfall (see Figs. 2, 7, and 9). High levels of these constituents are known to be detrimental to seagrass growth. Likewise, Tampa Bay seagrass monitoring programs have recorded recent reductions in expansion and also losses of seagrass cover, specifically in the upper bay segments. Other factors, such as accidental spills from industrial and municipal sources (see Cardinale 1998), some caused as a consequence of the high rainfall amounts, may have contributed to the recent seagrass loss and reductions in the seagrass expansion rates.

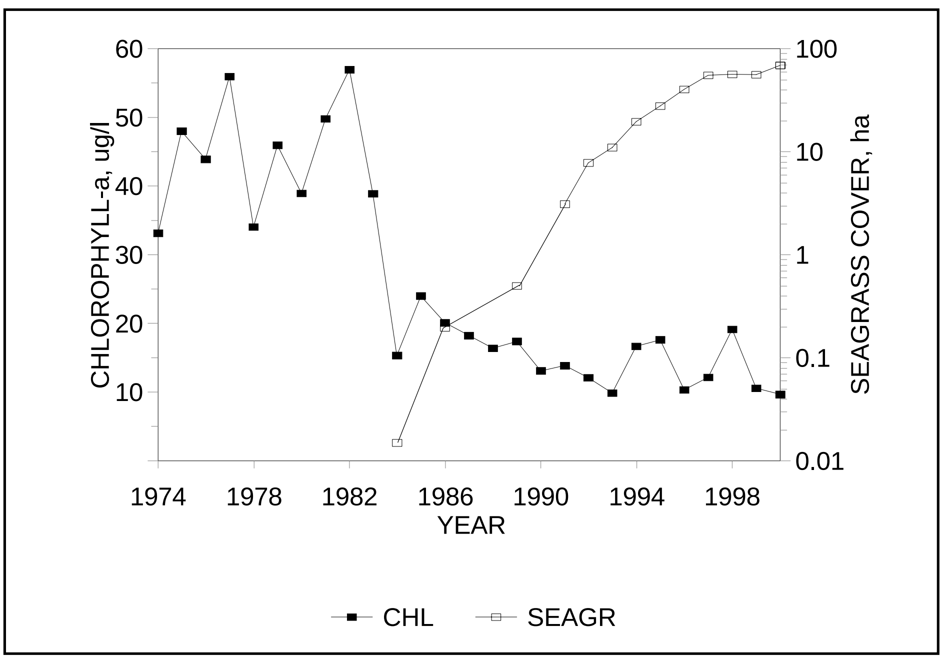

Figure 6. Hillsborough Bay annual average chlorophyll *a* concentrations and seagrass coverage illustrating the start of seagrass recovery in 1984 following the rapid decline in chlorophyll from 1982 to 1984. Chlorophyll *a* concentrations measured by the Hillsborough County Environmental Protection Commission and the City of Tampa. Seagrass coverage measured by the City of Tampa.

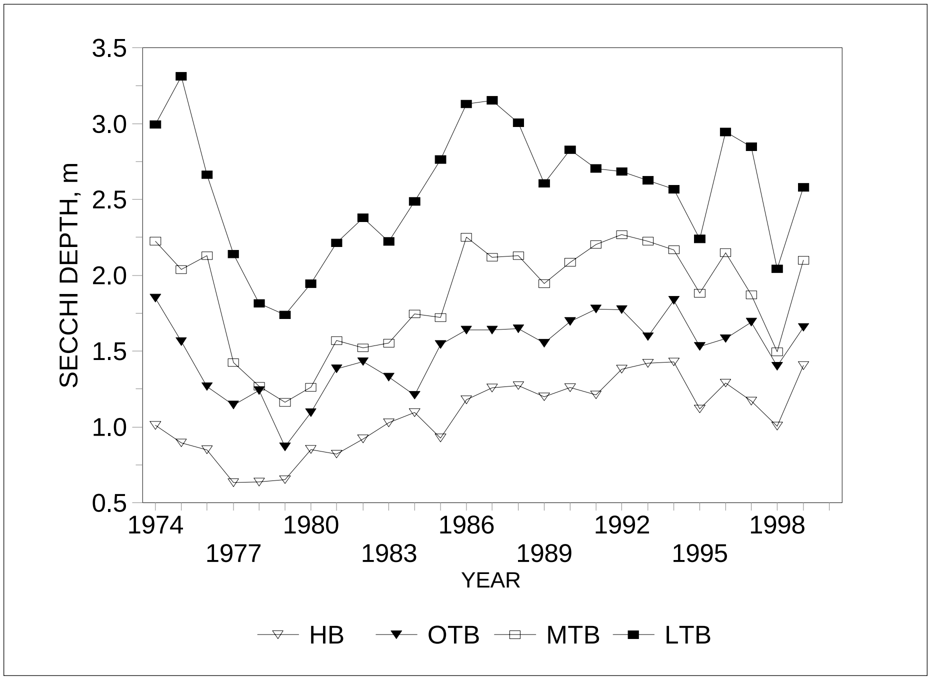

Figure 7. Annual average Secchi depths in the four major Tampa Bay segments measured by the Hillsborough County Environmental Protection Commission.

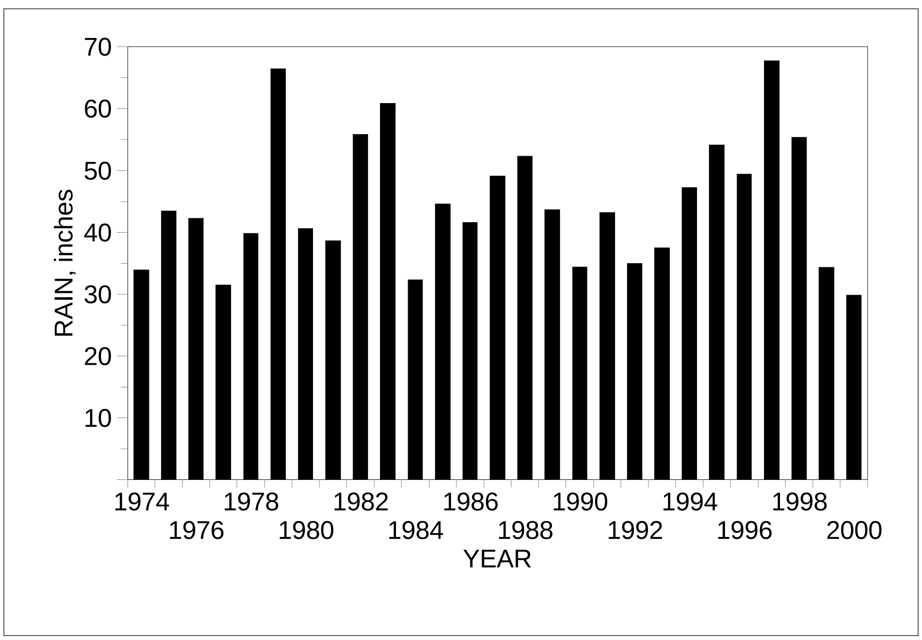

Figure 8. Rainfall at the Tampa International Airport measured by the National Oceanographic and Atmospheric Administration, National Climatic Data Center.

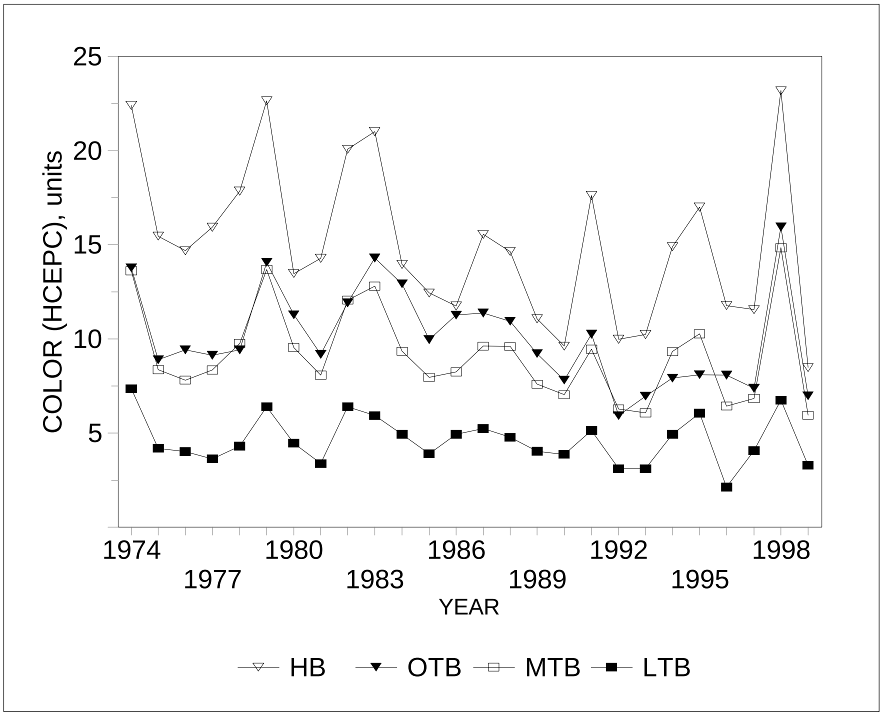

Figure 9. Annual average color content in the four major Tampa Bay segments measured by the Hillsborough County Environmental Protection Commission.

Rainfall in the Tampa Bay area was below normal in 1999 and 2000 (Fig. 8). With an extended period of relatively low rainfall the seagrass expansion rate may once again increase. Baywide information for 2000 seagrass coverage is not yet available, however, Hillsborough Bay seagrass coverage (see Fig. 4) increased substantially from about 56 ha in 1999 to about 69 ha in 2000.

DISCUSSION AND CONCLUSION

The large nitrogen reductions during the late 1970s and early 1980s apparently improved water quality in Tampa Bay after many years of increasing eutrophication. Also, Tampa Bay seagrass meadows started to expand in the mid 1980s, primarily in the upper segments of the bay. Concerned scientists have questioned if the seagrass expansion of the mid 1980s was largely caused by the reduction in anthropogenic nitrogen loading from point sources, or was it more strongly related to a period of relatively low rainfall that generally lasted from the mid 1980s through the early 1990s. It was cautioned that increased inputs of both nutrients and water with a high color content during future periods of high rainfall might reverse the recent water quality improvements and negatively impact the ongoing seagrass expansion by increasing water column light attenuation (Lewis et al. 1991).

An extended period of higher than normal rainfall started in 1995 and lasted through the 1997–98 El Niño event (Fig. 8). During this period, nitrogen loading (Fig. 3) to the bay increased, ambient nitrogen and chlorophyll *a* concentrations (Figs. 10 and 2) increased, and water clarity (Fig.7) decreased. However, both nitrogen and chlorophyll concentrations remained substantially lower during the recent high rainfall period than concentrations found prior to the large anthropogenic nitrogen loading reductions that occurred in the late 1970s and early 1980s. This suggests that the large anthropogenic nitrogen loading reductions have had a long-lasting effect of

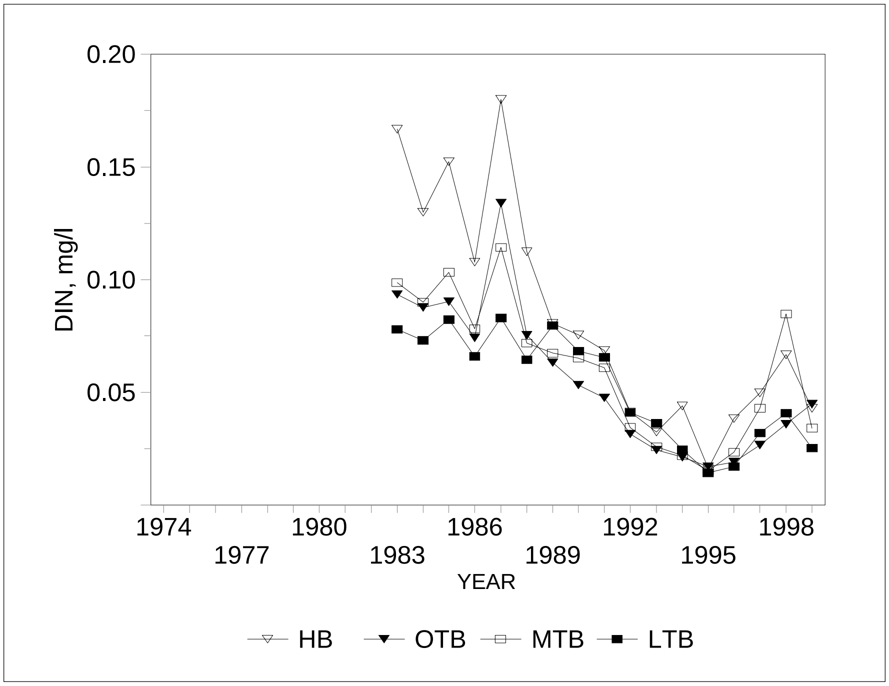

Figure 10. Annual average ambient dissolved organic nitrogen concentrations in the four major Tampa Bay segments measured by the Hillsborough County Environmental Protection Commission.

reducing eutrophication in Tampa Bay. In contrast, potential long-term effects on water clarity (measured as Secchi depth) from the large nitrogen reductions are less apparent. Water clarity reached minimum levels in all bay segments during the late 1970s. The low values were probably caused by a combination of high turbidity associated with the Tampa Bay ship channel deepening and widening project (late 1970s–early 1980s), and high phytoplankton biomass from high nitrogen loadings. After reaching the minimum values, water clarity increased at a constant rate in all four major bay segments during the 1980s. This trend continued into the early 1990s for all segments except Lower Tampa Bay. On the other hand, a distinct long-term improvement is not evident in the Tampa Bay water clarity record, with the possible exception for Hillsborough Bay, because water clarity over the two last decades has generally not exceeded the values recorded in the mid-1970s. It is evident that the large magnitude improvements seen in both ambient nitrogen concentrations and phytoplankton biomass are not strongly reflected in the water clarity record. This suggests that factors, including, events causing increased turbidity and rainfall related discharges of water with high color content (see Fig. 9) may have a substantial impact on water clarity. The caution postulated by Lewis et al. (1991), therefore, appears partially supported by field observations.

Finally, rainfall in the Tampa Bay area was below normal in 1999 and 2000. With an extended period of relatively low rainfall it could be expected, following the scenario developed by Lewis et al. (1991), that the Tampa Bay seagrass expansion rate again would increase. Baywide information for 2000 seagrass coverage is not yet available, however, it is encouraging that Hillsborough Bay seagrass coverage increased from about 56 ha in 1999 to about 69 ha in 2000.

ACKNOWLEDGMENTS

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suggestions during the preparation of the manuscript.

LITERATURE CITED

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STATUS AND TRENDS OF SEAGRASS COVERAGE IN TAMPA BAY, WITH REFERENCE TO OTHER SOUTHWEST FLORIDA ESTUARIES

David A. Tomasko

ABSTRACT

Seagrass coverage in Tampa Bay and other Southwest Florida estuaries has been mapped several times in the past twenty years. In addition, coverage estimates are available from as far back as 1950. Tampa Bay's seagrass coverage is estimated at 24,841 acres in 1999, down 2,074 acres from estimates for 1996, and the first decline in baywide coverage since 1982. The decline in coverage in Tampa Bay from 1996 to 1999 represents an 8% decrease. Sarasota Bay, which has a much more highly urbanized watershed than Tampa Bay, had an 11% decline in seagrass coverage between 1996 and 1999. Charlotte Harbor, which is considered to be a much more pristine system than either Tampa Bay or Sarasota Bay, had a 7% decline in seagrass coverage between 1996 and 1999. A potential cause of the 1996 to 1999 decreases in seagrass coverage in these estuaries is the 1997 to 1998 El Niño, which caused annual rainfall amounts to be 20% to 48% higher than the 1988 to 1999 average. The excessive nutrient and suspended solids loads that accompanied this El Niño would have most likely caused a considerable decrease in water clarity, which could account for the loss of seagrass coverage in portions of Tampa Bay. In contrast, Old Tampa Bay seems to be exhibiting signs of a prolonged decline in seagrass coverage not seen in other parts of the bay.

INTRODUCTION

Southwest Florida's estuaries have been the focus of a significant amount of research on the various interactions among pollutant loads, water quality, and seagrass health. In Charlotte Harbor, Tomasko and Hall (1999) found no direct link between nutrient loads and spatial and temporal variation in the productivity and biomass of seagrass meadows. In Lemon Bay, Tomasko et al. (2001) found that nutrient loads seemed to correlate with chlorophyll a concentrations, and that chlorophyll a concentrations were good indicators of water clarity. In Sarasota Bay, the biomass and productivity of seagrass meadows were inversely correlated with nitrogen load estimates, although water quality measurements did not seem to correlate with nitrogen loads (Tomasko et al. 1996).

In Tampa Bay, recent increases in seagrass coverage have been linked to improved water quality. Improvements in water quality are due mostly to reductions in phytoplankton levels, which, in turn, have been linked to reductions in anthropogenic nitrogen loads (i.e., Johansson 1991, Avery 1997, Johansson and Ries 1997, Johansson and Greening 1999).

As a consequence, seagrass mapping efforts have played an important role in gaging the success, or lack thereof, toward maintaining and expanding on improvements to water quality in Tampa Bay and other estuaries. These mapping efforts are conducted on a roughly biennial basis by the Southwest Florida Water Management District (District).

For the 1999 and previous seagrass mapping efforts, the District puts out a Request for Proposals for consultants to conduct aerial photography, ground-truthing, photointerpretation and GIS-based analysis. In 1999, the successful respondent was Agra-Baymont, Inc. A post map-production classification accuracy assessment (described below) is conducted by District staff.

This paper updates a previous report on the status and trends of seagrass coverage in

Tampa Bay and adjacent estuaries (Kurz et al. 1999).

MATERIALS AND METHODS

Seagrass maps have been produced for the District for the years 1988, 1990, 1992, 1994 and 1996. Previous efforts have been conducted for the years of 1950 (Tampa Bay Regional Planning Council 1986) and 1982 (Haddad 1989). Seagrass maps are produced through a multiple step process. First, aerial photography is obtained, usually in the late fall to early winter. This time of year is associated with both good water clarity and relatively high seagrass biomass. Photography is true color at a scale of 1:24,000. On the day that photography is to be obtained, secchi disk depths must at least 2 meters for each estuary flown, wave height must be less than 2 feet, and wind speed must be less than 10 miles an hour. Other restrictions include sun angle must be greater than 35%, and cloud cover and/or haze must not interfere with the quality of the photography.

Second, photointerpretation efforts are conducted in the field, to allow for the successful evaluation of distinct photographic signatures. Seagrass signatures are divided into two classes: continuous coverage (<25% unvegetated bottom visible within a polygon) and patchy coverage (>25% unvegetated bottom visible within a polygon), with a minimum mapping unit of 0.5 acres.

Third, polygons are integrated into an ARC/Info program. For past efforts, individual polygons were delineated on mylar overlays, cartographically transferred using a Zoom Transfer Scope to USGS quadrangles, and then digitally transferred to an ARC/Info data base for further characterization. These techniques allowed for the seagrass maps to meet USGS National Map Accuracy Standards for 1:24,000 scale maps. For 1999 seagrass maps, a 1:12,000 National Map Accuracy Standard was met. While photography remained at a scale of 1:24,000, the higher positional accuracy standard required the use of tighter ground control and more sophisticated mapping techniques. Analytical stereo plotters were used for photointerpretation, as opposed to stereoscopes. This technique allowed for the production of a georeferenced digital file of the photointerpreted images, without the need for additional photo to map transfer.

Fourth, hard copy plots were made of photointerpreted seagrass coverage, and sixty (60) randomly chosen points were identified for a post map-production classification accuracy assessment. A hand-held Global Positioning System was used, along with the map and the latitude and longitude of the randomly located stations, to develop an unbiased determination of the map's classification accuracy. A 90% classification accuracy standard is required in the consultant's contract, and a 96% accuracy was achieved for 1999 efforts (i.e., 53 of 55 stations that could be visited were accurately described).

RESULTS

Figure 1 shows the location of seagrass coverage in 1999 for Tampa Bay. Most seagrass acreage is in the lower, higher salinity portions of the bay, with the most extensive grass flats being found in the vicinity of Fort DeSoto Park, just west of the northern end of the Sunshine Skyway. Extensive flats are also found on the southeastern shore of Tampa Bay, from Anna Maria Island up to the Little Manatee River. There is significant coverage in Old Tampa Bay and the southern tip of the Interbay Peninsula, while coverage in Hillsborough Bay is much reduced, compared to other portions of the bay.

Using estimates of coverage from 1950 and 1982, Figure 2 shows the overall trend in seagrass coverage for Tampa Bay over the

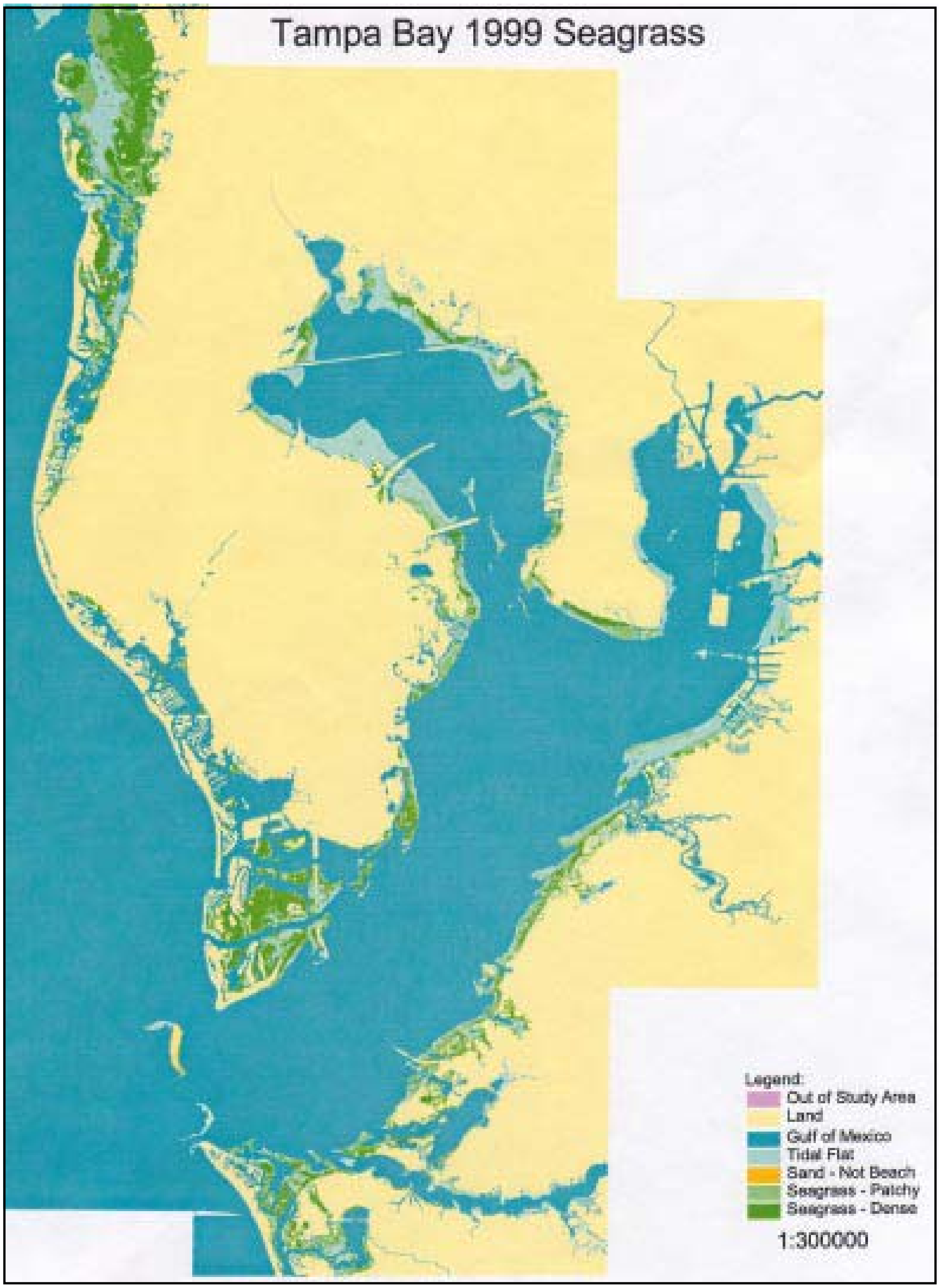

Figure 1. Seagrass coverage in Tampa Bay in 1999.

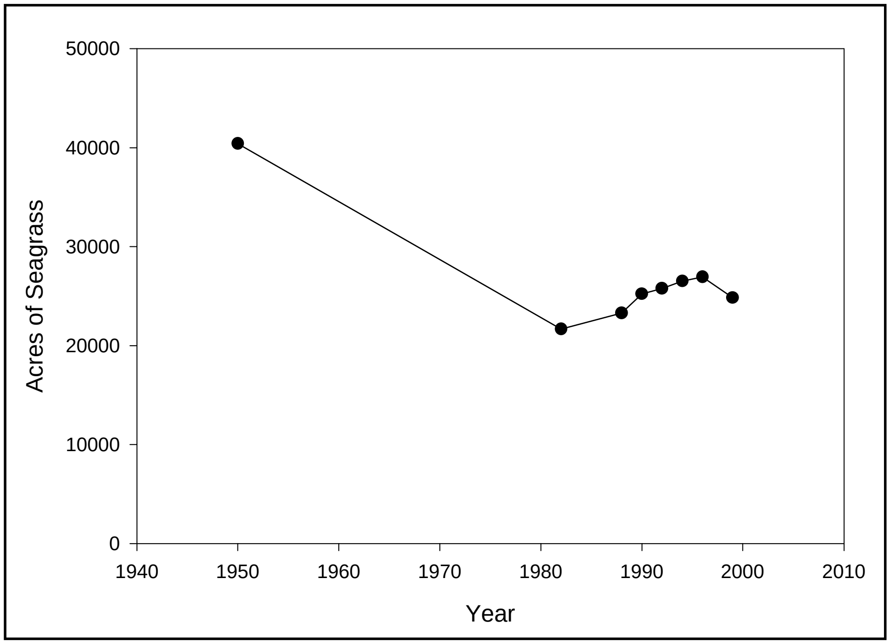

Figure 2. Estimated seagrass coverage from 1950 to 1999 for Tampa Bay.

past 50 years. In 1950, it was estimated that seagrass meadows covered 40,644 acres of bay bottom. By 1982, that number had dropped to 21,653 acres. From 1982 to 1996, acreage increased by 5,262 acres, up to 26,915 acres. The average rate of increase between 1982 and 1996 was 376 acres per year. From 1988 to 1996, the average rate of increase was 454 acres per year, from 23,285 to 26,915. From 1996 to 1999, a decrease in coverage of 2,074 acres was found (8% decline), down to 24,841 acres. Seagrass coverage in Tampa Bay in 1999 is 61% of 1950 estimates.

Trends in seagrass coverage varied among different bay segments. Employing the segmentation scheme used by the Tampa Bay Estuary Program (first proposed by Lewis and Whitman 1985) the bay can be divided into the following segments: Hillsborough Bay, Old Tampa Bay, Middle Tampa Bay, Lower Tampa Bay, Boca Ciega Bay, Terra Ceia Bay, and the Manatee River. Based on this segmentation scheme, various patterns of seagrass coverage can be detected.

In Hillsborough Bay, seagrass coverage is estimated to have dropped from 2,300 acres in 1950 to a complete absence in 1982 (Fig. 3). In 1999, coverage is estimated at 192 acres, a rate of increase of 11 acres per year between 1982 and 1999. Seagrass coverage in Hillsborough Bay in 1999 is 8% of 1950 estimates.

In Old Tampa Bay, seagrass coverage declined from an estimated 10,700 acres in 1950 to 5,006 acres in 1988 (Fig. 4). From 1988 to 1994, acreage increased at a rate of 151 acres per year, to 5,911 acres. From 1994 to 1996, acreage decreased to 5,763 (74 acre per year decline). From 1996 to 1999, acreage decreased to 4,395, a rate of decline of 456 acres per year. Seagrass coverage in Old Tampa Bay in 1999 is 41% of 1950 estimates.

In Middle Tampa Bay, seagrass coverage declined from 9,600 acres in 1950 to 4,042 acres in 1982 (Fig. 5). From 1982 to 1999, seagrass coverage increased by 1,597 acres, a 40% increase. Seagrass coverage in Middle Tampa Bay in 1999 is 59% of 1950 estimates.

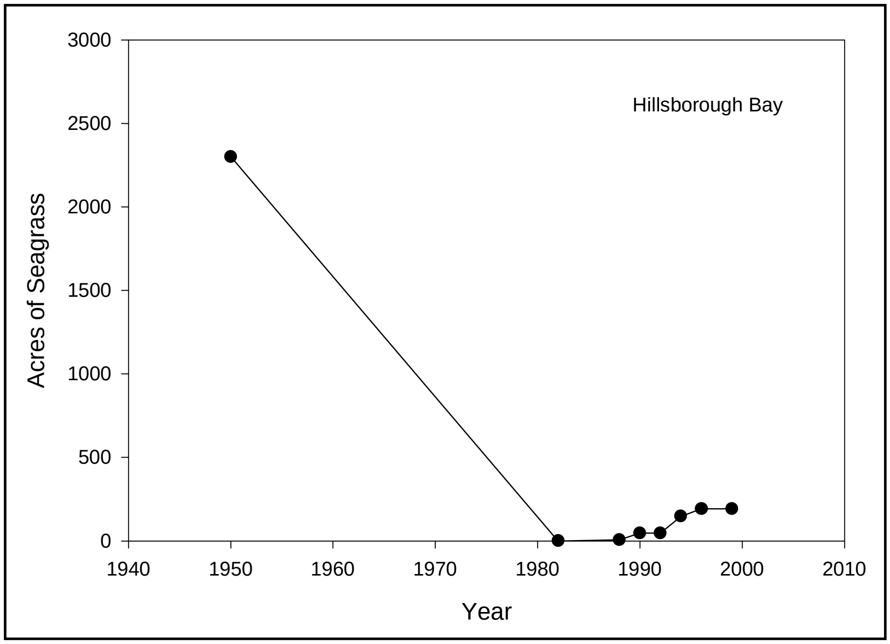

Figure 3. Estimated seagrass coverage from 1950 to 1999 for Hillsborough Bay.

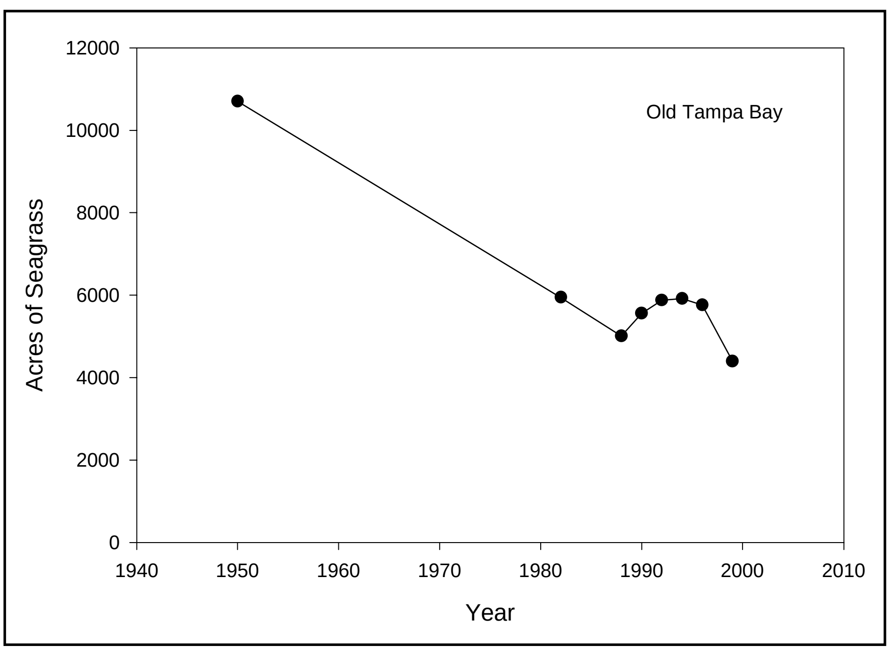

Figure 4. Estimated seagrass coverage from 1950 to 1999 for Old Tampa Bay.

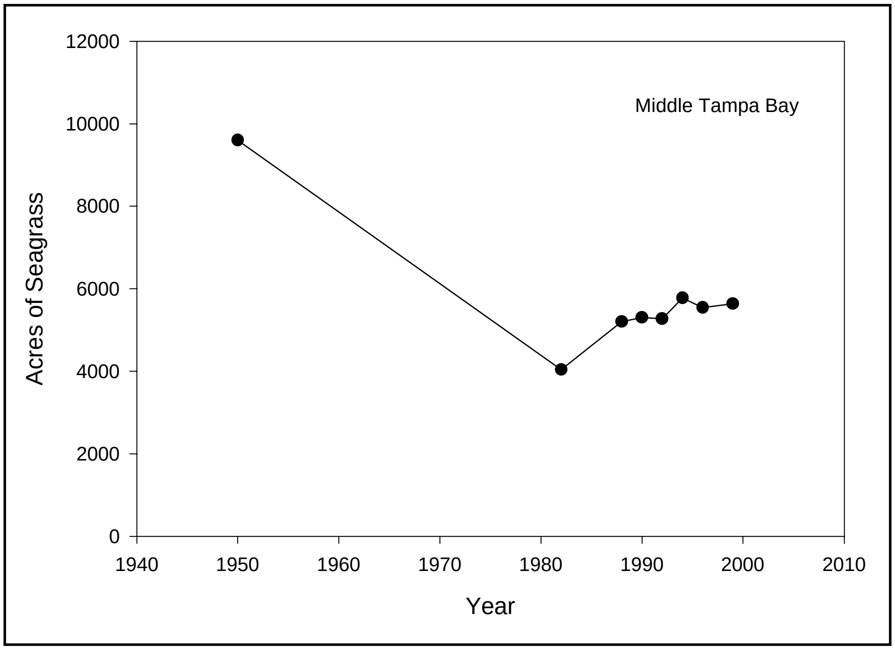

Figure 5. Estimated seagrass coverage from 1950 to 1999 for Middle Tampa Bay.

Portions of the lower bay appear to exhibit a more optimistic picture of seagrass recovery. In Lower Tampa Bay, seagrass coverage in 1950 is estimated at 6,100 acres (Fig. 6). In 1999, seagrass coverage in Lower Tampa Bay is estimated at 5,847 acres, or 96% of 1950 estimates.

In Boca Ciega Bay, seagrass coverage declined from an estimated 10,800 acres in 1950 to 5,770 acres in 1982 (Figure 7). From 1982 to 1999, seagrass coverage increased at a rate of 100 acres per year, to a total of 7,464 acres. Seagrass coverage in Boca Ciega Bay in 1999 is 69% of 1950 estimates.

In Terra Ceia Bay, seagrass coverage remained similar between 1950 and 1982 (700 and 751 acres, respectively) as shown in Figure 8. However, 1988 coverage is estimated to be 947 acres, a 26% increase from 1982. In 1999, seagrass coverage in Terra Ceia Bay is estimated at 929 acres, or 33% higher than in 1950.

In the Manatee River, seagrass coverage declined between 1950 and 1982 (200 and 131 acres, respectively) as shown in Figure 9. However, 1988 seagrass coverage is estimated to be 347 acres, a 165% increase from 1982. In 1999, seagrass coverage in the Manatee River is estimated at 375 acres, or 88% higher than in 1950.

Most bay segments exhibit patterns of periods of losses and gains of seagrass coverage over time. However, Old Tampa Bay is the only bay segment that appears to have less seagrass coverage at present than at any previous time period during 1950 to 1999 (Fig. 4).

DISCUSSION

Tampa Bay's seagrass coverage has been previously shown to be positively responding to the improved water clarity that has accompanied massive reductions in anthropogenic nitrogen loads (i.e., Johansson 1991, Avery 1997, Johansson and Ries 1997, Johansson and Greening 1999). However, the 1996 to 1999 seagrass mapping effort appears to have documented an unexpected decline in coverage of 2,074 acres, an 8% decrease.

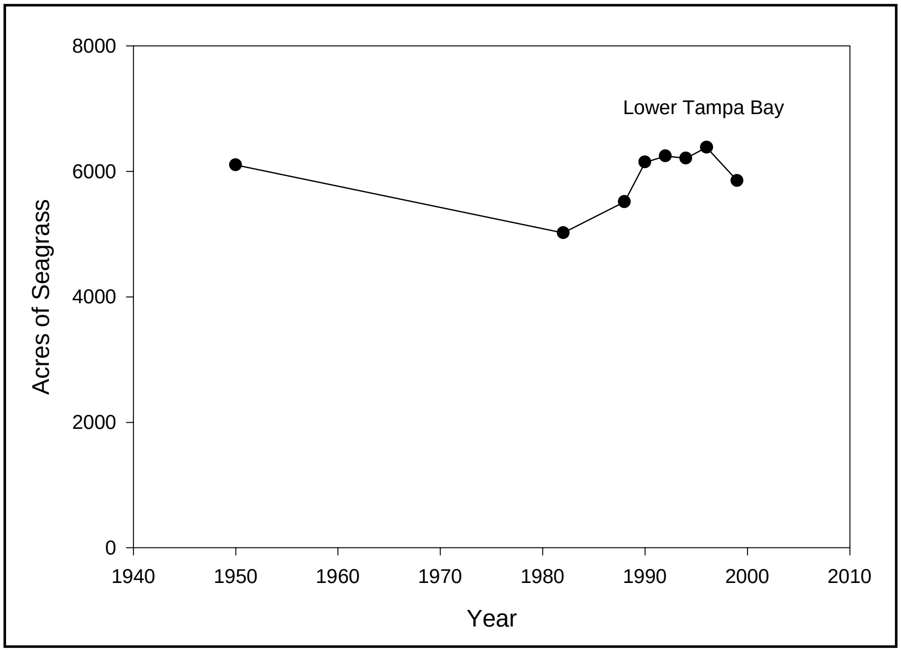

Figure 6. Estimated seagrass coverage from 1950 to 1999 for Lower Tampa Bay.

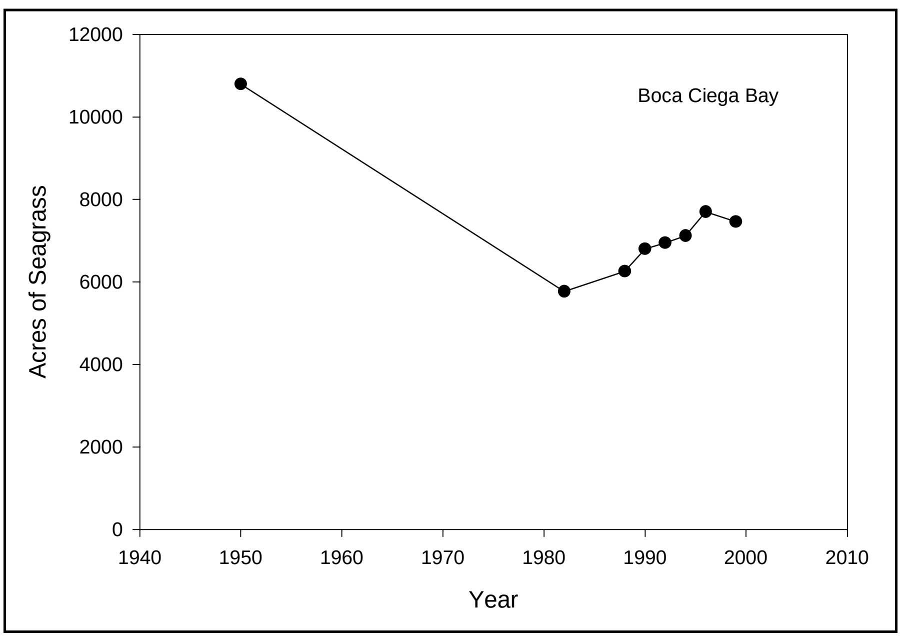

Figure 7. Estimated seagrass coverage from 1950 to 1999 for Boca Ciega Bay.

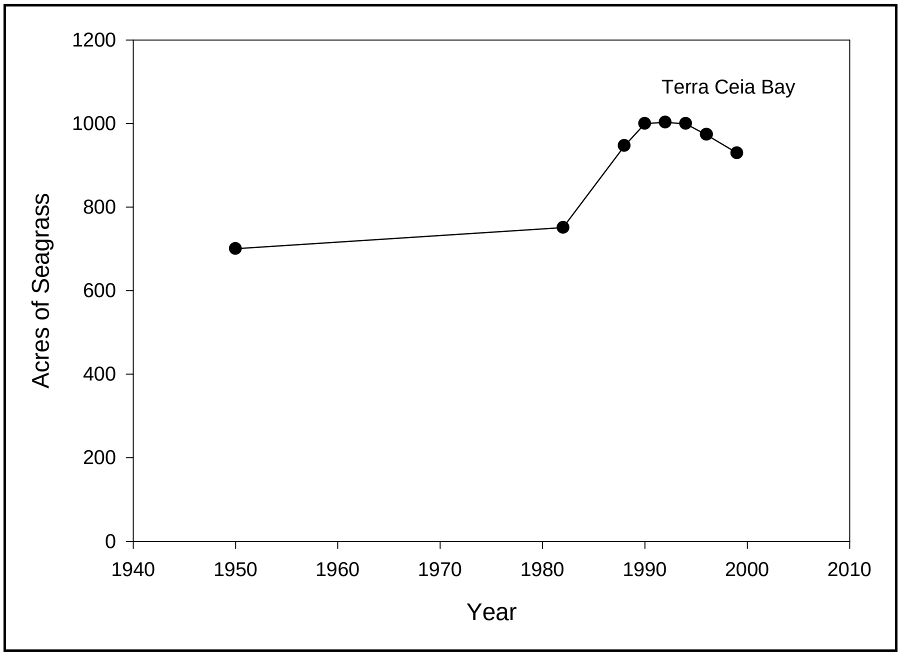

Figure 8. Estimated seagrass coverage from 1950 to 1999 for Terra Ceia Bay.

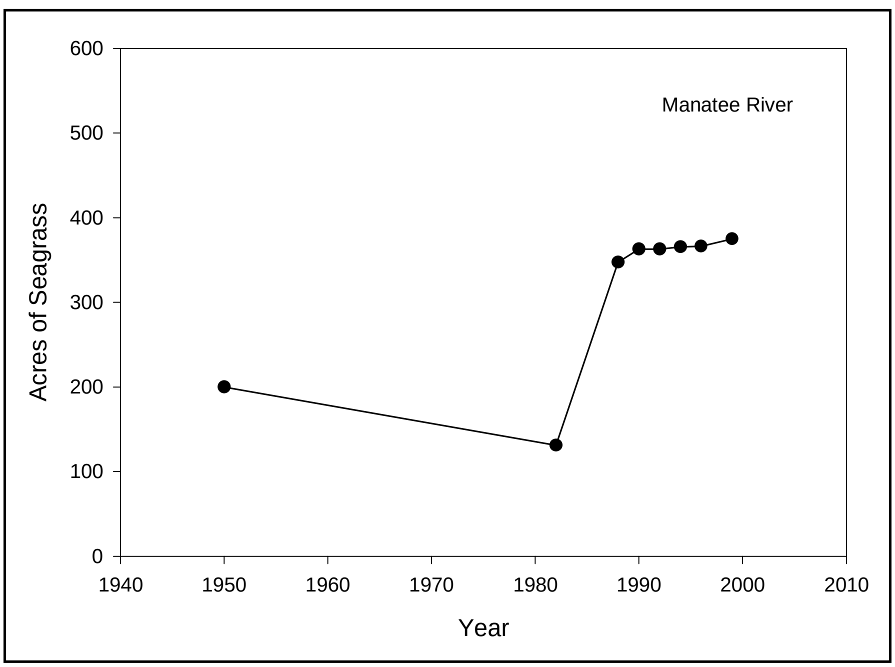

Figure 9. Estimated seagrass coverage from 1950 to 1999 for the Manatee River.

While some of the 1996 to 1999 seagrass coverage decline might be due, in part, to different cartographic techniques, the techniques in question are designed to improve the positional accuracy of the 1999 maps; they should not have had any effect on photointerpretation results. Additionally, permanently marked transects used for monitoring seagrass health in Tampa Bay have shown that seagrass declines have in fact occurred in the bay, particularly in Old Tampa Bay (Avery 1999). Also, examination of aerial photography and extensive groundtruthing support the contention that seagrass coverage has declined in the bay in recent years, particularly in the western portion of Old Tampa Bay, just north and south of the Howard Frankland Bridge.

The potential for rainfall deficits to explain a portion, at least, of seagrass increases between 1982 and 1988 was briefly explored by Lewis et al. (1991). However, Johansson (1991) found no correlation between annual rainfall over the Hillsborough Bay basin and annual average chlorophyll *a* concentrations in Hillsborough Bay.

From 1988 to 1999, annual rainfall measured at Tampa International Airport averaged 46 inches (District rainfall data). During 1997, annual rainfall rose to 68 inches, with a total of 55 inches in 1998. These two years, with rainfall amounts 48 and 20% higher than the 1988 to 1999 average, correspond to the 1997 to 1998 El Niño event, which caused substantial flooding in Southwest Florida. It is likely that the 1997 to 1998 El Niño event caused seagrass coverage declines to occur in Tampa Bay, due to the significant increases in nutrient and suspended solids loads that would have accompanied such an incident.

In other estuaries in Southwest Florida, seagrass coverage also decreased between 1996 and 1999. For example, Sarasota Bay's seagrass coverage declined from 10,333 acres to 9,247 acres, a decrease of 11%. And in Charlotte Harbor, seagrass coverage declined from 19,225 acres to 17,942 acres, a decrease of 7%.

Sarasota Bay's watershed is actually more "polluting" than Tampa Bay's (estimated nitrogen yields of 10.9 vs. 5.9 kg total nitrogen [TN] / hectare / yr), which seems to mostly reflect the degree of urbanization of the watershed (Tomasko 2000). Charlotte Harbor, with only 6% of its watershed urbanized, has an estimated watershed nitrogen yield of only 2.6 kg TN / ha / yr, less than half of Tampa Bay's (Tomasko 2000).

The estimated 8% decrease in Tampa Bay's seagrass coverage between 1996 and 1999 might not be overly concerning, as the rate of decline in coverage is similar to that which occurred in the relatively pristine Charlotte Harbor system, and the time period between 1996 and 1999 included a significant El Niño event. However, the 24% decrease in seagrass coverage in Old Tampa Bay between 1996 and 1999, and the prior decreases between 1994 and 1996, could potentially signify a more serious condition of continuing seagrass decline.

Further mapping efforts are necessary to continue to monitor the success of ongoing efforts to improve water quality and seagrass coverage in Tampa Bay.

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LIGHT REQUIREMENTS OF TAMPA BAY SEAGRASSES: NUTRIENT-RELATED ISSUES STILL PENDING

L. K. Dixon

ABSTRACT

Epiphytic growth on seagrass blades has been demonstrated to respond to nutrient loadings, directly reduces available light to submerged vegetation, and remains largely unquantified in the upper portions of Tampa Bay. In the lower Bay, epiphytes attenuate approximately one third of all light reaching the bottom. Annual average epiphytic attenuation in nearby estuaries can reach 55% and is inversely correlated with salinity (as an inverse surrogate for nutrient load). If epiphytic attenuation is greater in portions of Tampa Bay than where the light targets were determined, then achieving water column targets will still result in insufficient light reaching the blades of the seagrasses. Discrete monitoring, even weekly, can overestimate water column light levels by as much as 5.6%, a large bias when compared to water column light targets of 20.5%. The use of light targets, measured as a percentage of immediately subsurface photosynthetically active radiation, may not be as sensitive a tool for prediction of biomass changes as are total quanta received, but appears to be the only feasible method to conduct as routine monitoring. Additionally, there appear to be seasonal light requirements that are not reflected in the water column light target for *Thalassia testudinum*. The effects of other stressors (such as low salinity) have been seen in nearby systems in which biomass declines have accompanied light levels well in excess of 20.5%.

INTRODUCTION

Reduced water clarity has long been considered one of the dominant contributors to seagrass loss in the Tampa Bay system as losses were concentrated at the deeper edges within each Bay segment. Initial work to achieve seagrass recovery in Tampa Bay demonstrated empirical linkages between nitrogen loading and chlorophyll, and between chlorophyll and water clarity. Determination of light needed by seagrasses would allow the empirical relationships to be applied 'in reverse', i.e. knowing required water clarity, one could calculate maximum chlorophyll values and maximum nitrogen loading that would still permit grasses to exist at established target depths.

ESTABLISHMENT OF WATER COLUMN LIGHT TARGETS

The percentage of immediately subsurface photosynthetically active radiation which remains in the water column is referred to as %PARw. The water column light target for *T. testudinum* was empirically determined as the annual average %PARw required to maintain interannual stability of shoot density and biomass at the deepest edge of a seagrass bed. The %PARw were computed from *continuous water column* PAR data collected between 1000 and 1400 hours. The resulting attenuation coefficients were used to extrapolate canopy level PARw to immediately subsurface values, and thence to %PARw. The individual PAR data were adjusted for the effects of bottom reflectance (which increase the PAR values recorded by a spherical sensor within 30 cm of the bottom by between 5 and 15%) in order to calculate immediately subsurface values of PAR. By removing bottom reflectance, the %PARw light target address only the light attenuation of an infinitely deep water column. Ecologically, *T. testudinum* blades are exposed to the reflected light, undoubtedly make use of the additional photons so available, and plant requirements should retain the reflected fraction. Additionally, the %PARw target value is strictly comparable only to monitoring data which either 1) similarly adjust for bottom reflectance, 2) are taken in the upper portion of the water column or much deeper waters where the bottom-reflected

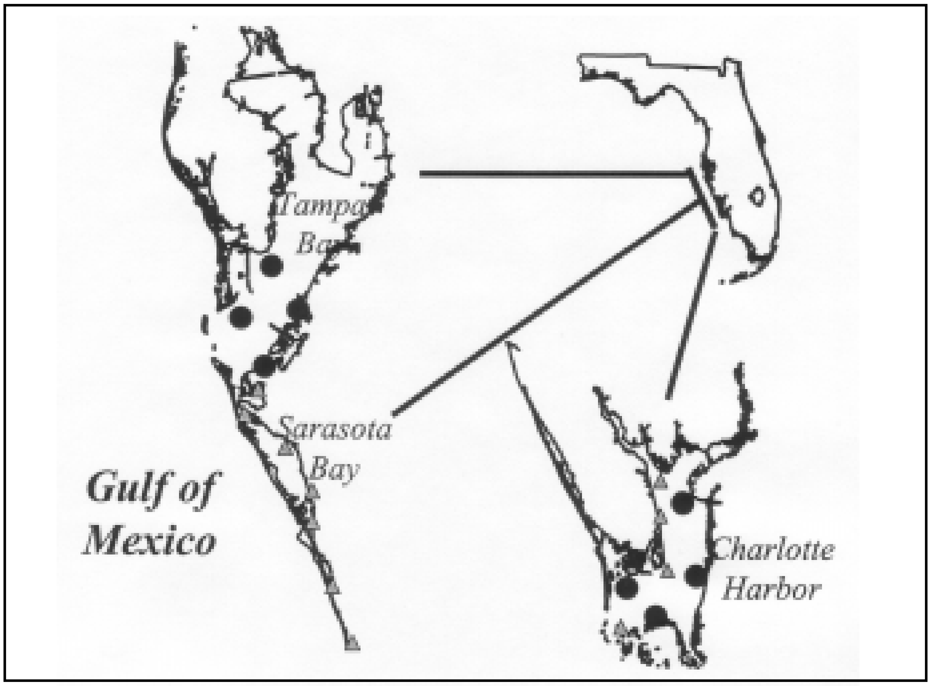

Figure 1. Stations where continuous light data (circles) or discrete samplings (triangles) were obtained. Epiphytic attenuations were determined at all locations.

PAR is minimal, or 3) use flat (2 pi) rather than spherical (4 pi) sensors. (Failing to adjust for bottom reflectance will produce a higher PARw value at depth, an apparently lower attenuation coefficient, which, when applied to the entire water column, will result in a falsely high %PARw computed.)

In Tampa Bay, and for the prevailing conditions of epiphyte coverage, %PARw was established at 20.5% (Dixon and Leverone, 1995). For technical reasons associated with depths of the deep edge of the grasses and equipment size, the data to establish the 20.5% PARw target were collected at four sites in the lower portion of Tampa Bay (Figure 1), where water clarity is among the highest and nutrient and color concentrations among the lowest of the entire Bay system. One of the sites exhibited some apparent shading responses (blade elongation, reduced blade biomass per blade area), but these data were included in the mean calculation of %PARw targets, nevertheless. (The station with shading response received 18.5%PARw, while the remainder received between 20.2% and 21.2%.) The study year was evaluated against historical data, determined to be within the typical water clarity variations experienced, and so targets determined in this period should be representative of ecologically relevant light requirements.

Monitoring programs have since determined that PARw values of approximately 20.5% are present at the depths desired for seagrass restoration for the lower and other segments of the Tampa Bay system, but that seagrass recovery has not expanded to these depths. Other reasons for non-recovery are being proposed, but additional light and nutrient related issues should be considered, based on data collected in Tampa Bay, Sarasota Bay, and Charlotte Harbor.

OTHER STRESSORS, SEASONAL REQUIREMENTS, AND MAINTENANCE vs EXPANSION

Additional physical stresses or seasonal timing of PARw might prevent recovery even when sufficient annual PARw is present. In Charlotte Harbor, biomass declines occurred at one *T. testudinum* bed even though annual PARw received was well over 20.5% (Dixon and Kirkpatrick, 1999). It is difficult to separate the observed declines from either the extreme salinity depression produced by record freshwater flows during the study year or the extreme attenuation from water column color during a seasonal period when growth typically occurs. Others have also found that low temperatures and/or endogenous growth rhythms can effectively halt growth, even when PAR levels are high (Tomasko and Hall, 1999). The continuous PAR data of Tampa Bay (Dixon and Leverone, 1995) indicate that slightly depressed PARw levels during spring and early summer are associated with net biomass declines over the course of a year, even when annual average PARw is very near 20.5%.

In sum, the annual 20.5% PARw target does not incorporate any seasonal light requirements. A conceptual energy model which incorporates temperature related growth and respiration, whole-plant compensation and saturation points, light levels and day lengths would be very helpful in providing a range of seasonal PARw requirements which would maintain net levels of biomass from year to year. Another consideration, from an energy standpoint, is that the 20.5% value is that required to *maintain* biomass, and does not necessarily permit expansion. Depending on the extent and distance of support between shoots, plant material at the deep edge may need to as much as double biomass to achieve significant expansion within a growing season, a growth rate that may not be achievable under maintenance light conditions.

DIFFERENCES IN CONTINUOUS vs DISCRETE MONITORING

The differences between the continuous PARw data (used to set light targets) and the typical monitoring schedules used to determine target achievement should also be considered. While not suggesting that continuous data is an effective (or economical) tool, comparisons between continuous data and the discrete measurements collected during station service visits reveal comparatively large differences. At stations in Charlotte Harbor, ~52 weekly measurements overestimated mean annual %PARw at three of four sites by up to 5.6% in comparison to continuous data (Dixon and Kirkpatrick, 1999). At one station, weekly data overestimated annual PARw as 22.6%; continuous data for the site recorded 17.1% PARw (Figure 2). The fourth station underestimated %PARw by 0.7%. The bias of intermittent sampling towards overestimates can be attributed to lack of discrete sampling during adverse conditions, and continuous sampling occurring over the entire period of 1000 to 1400. Monthly monitoring has the potential for generating even larger differences but with larger uncertainties on the order of ± 2 to 5%. Longer term data sets of discrete sampling may or may not converge on continuous %PARw if sampling biases remain. The suggestion here is that, despite monthly monitoring results, %PARw values may still be as much as 5% short of required levels in the various Bay segments. Recall that shading responses were seen at a site in Tampa Bay with 18.5%PARw or only 2% less than the established light target.

EPIPHYTIC ATTENUATION

An important aspect of the light target used for Tampa Bay is that it does *not* represent the %PAR actually received by the blades of *T. testudinum*. Epiphytic materials growing on the blades were scraped from entire shoots and a shoot integrated attenuation determined (Dixon and Kirkpatrick, 1995). For lower Tampa Bay, epiphytic growths, on average, attenuated approximately one third, or 34%, of all water column light penetrating to the deep

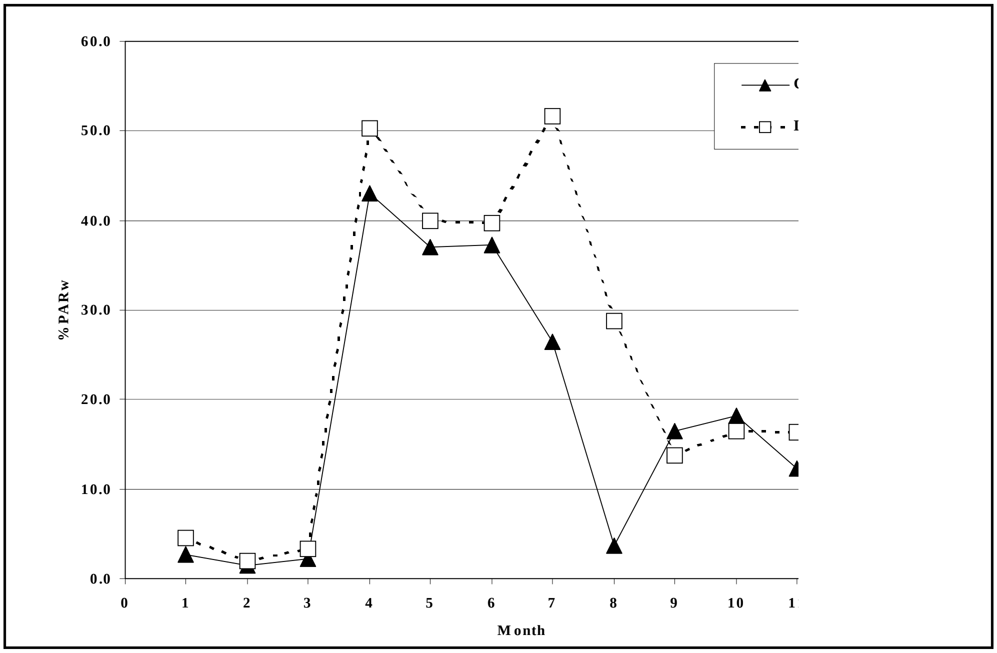

Figure 2. Differences between %PARw computed from weekly discrete samplings and continuous data, both between 1000-1400 hours. Data from a station in Charlotte Harbor.

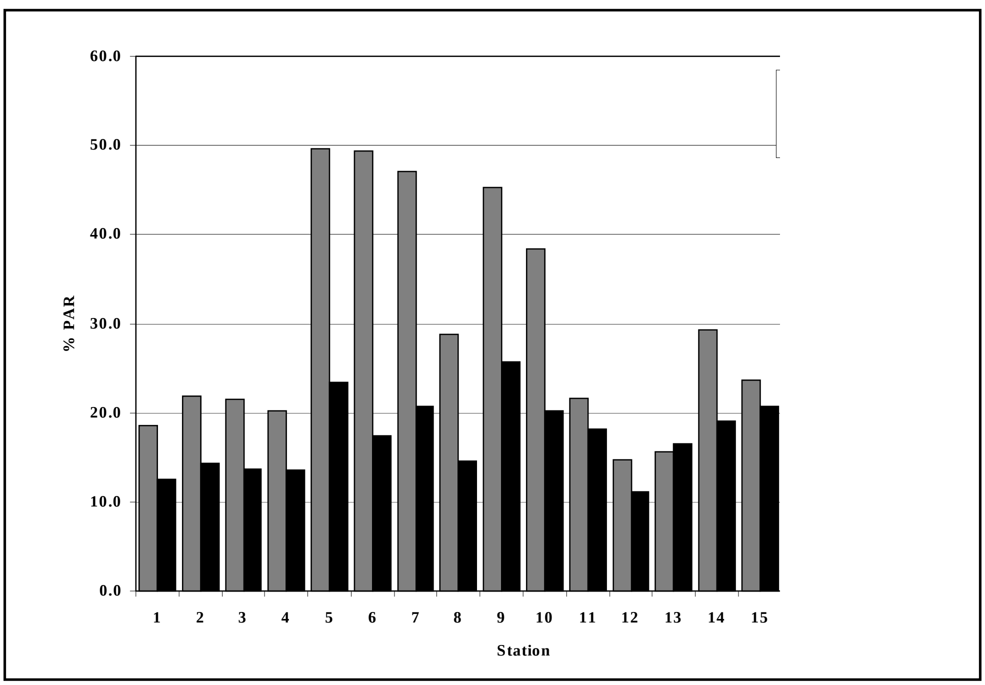

Figure 3. Variation in %PARw (crossed pattern) between stations is reduced when epiphytic attenuation is incorporated to compute %PARp (solid fill). Stations 1-4 in Tampa Bay; Stations 5-10 in Sarasota Bay, and Stations 11-18 in Charlotte Harbor. Stations include both *T. testudinum* and *H. wrightii.*

edge. If 20.5%PARw is reduced by an additional 34%, the %PAR available to the plant (%PARp) is only 13.5%. This is not to suggest that 13.5% is an appropriate water column target, but to point out that 20.5% *is* an appropriate water column target *only if* epiphytic attenuations are 34% or less.

Recall that all four stations used for Tampa Bay light target determination were in the relatively low nutrient conditions of the lower Bay. The riverine flows to the head of the Bay routinely establish nutrient gradients with higher concentrations in Hillsborough and Old Tampa Bays. Many authors have presented increases of epiphytes and macroalgal communities associated with increased or higher nutrient loadings. It is possible that epiphytic attenuations are higher in the upper segments, in which case a water column target of 20.5 %PARw at depth will be insufficient to even maintain, much less permit restoration of grasses.

Additional work has quantified both light targets and epiphytic attenuation in Sarasota Bay, immediately to the south, and in Charlotte Harbor. These data were gathered across greater ranges of salinity (and therefore presumably across a range of nutrient loadings) and demonstrate the potential range in epiphytic attenuation for Tampa Bay. Sarasota Bay is a lagoonal system, with a much smaller watershed than Tampa Bay, although highly urbanized along the coastal sections. Charlotte Harbor has a larger and relatively undeveloped watershed compared to Tampa, and stations were along a very broad salinity gradient.

In a clear illustration of non-transferability of water column targets, annual light targets of %PARw were determined to be 20.5% in Tampa Bay, between 25% and 50% in Sarasota Bay, and between 15% and 30% in Charlotte Harbor if epiphytic attenuations are not considered (Figure 3). These other studies addressed both *T. testudinum* and *Halodule wrightii*, and utilized a variety of continuous, biweekly, and weekly data. Nevertheless, the wide range in apparent %PARw targets argues for considerable caution in application of light targets from even one portion of the Bay to another where epiphytic conditions might be expected to differ.

Much of the variation in annual %PARw at the stable deep edges of seagrass beds can be explained by incorporating epiphytic attenuation and comparing %PARp between sites instead. Epiphytic attenuation ranged between 32.0% and 36.2% (mean of 34%) for Tampa Bay with mean salinities between 28 and 30 (Dixon and Leverone, 1995). Six stations in Sarasota Bay recorded between 40% and 55% annual epiphytic attenuation with salinity between 28 and 32 (Dixon and Kirkpatrick, 1995). Annual epiphytic attenuation at eight Charlotte Harbor stations was between 20% and 45% under salinities ranging from 15 to 28 (Dixon and Kirkpatrick, 1999). For the studies with broader salinity ranges, epiphytic attenuation was inversely correlated with salinity (Figure 4). While the inverse relationships of epiphytic attenuation with salinity differ greatly between estuaries, and undoubtedly reflect system-specific nutrient loadings, it is reasonable to view salinity as an approximate surrogate for nutrient loading within a given system and to expect higher epiphytic attenuations at lower salinity values.

Applying epiphytic attenuations to measured %PARw, generates %PARp values for *T. testudinum* that were near 13.5 % for Tampa Bay, 23%–27% for stable *T. testudinum* sites in Sarasota Bay, and 14%–16% for stable *T. testudinum* sites in Charlotte Harbor (Figure 3). As all studies sampled for at least a one year period, differences between studies might be attributed to biases generated by biweekly

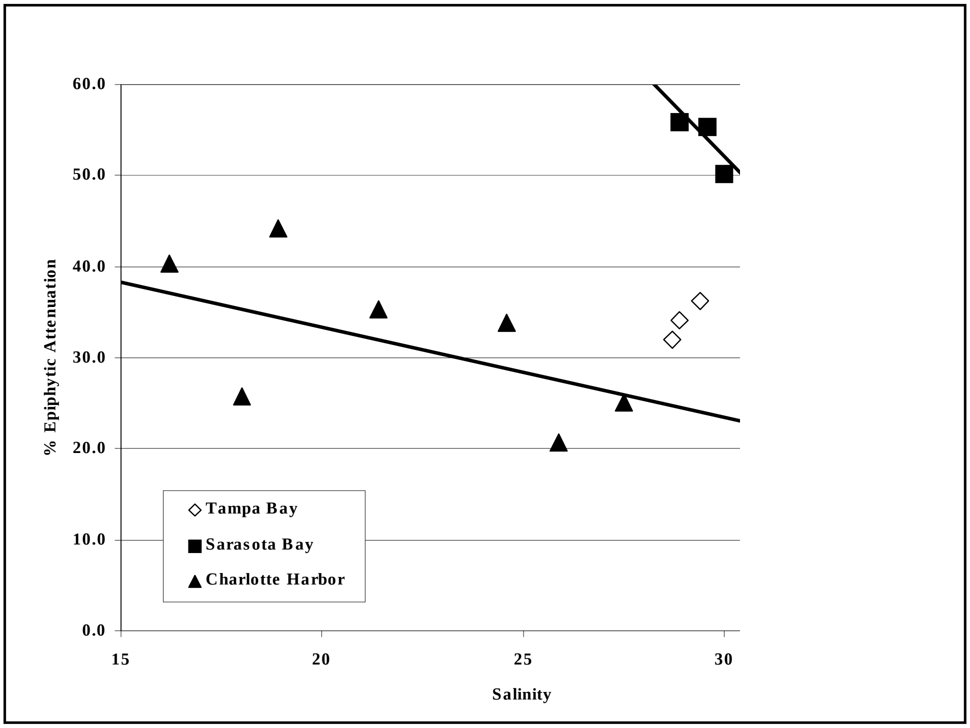

Figure 4. Inverse relationship between annual epiphytic attenuation values and salinity as an inverse surrogate of nutrient loadings. Data from Tampa Bay, Sarasota Bay, and Charlotte Harbor.

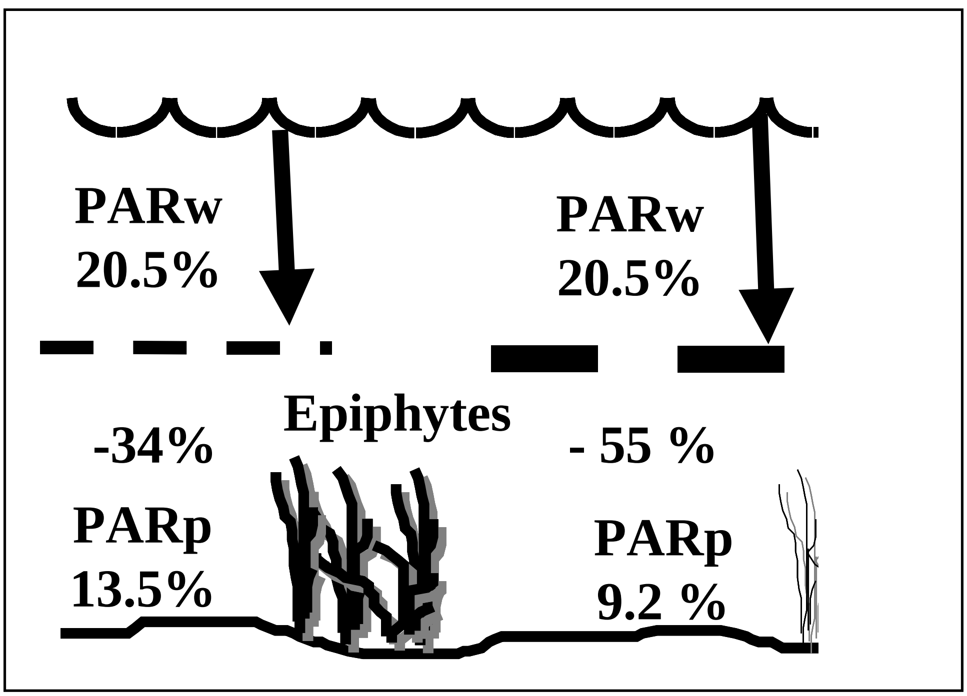

Figure 5. Potential impact of using 20.5%PARw as a water column target under varying epiphytic attenuations. The %PARp required by *T. testudinum* appears to be 13.5%.

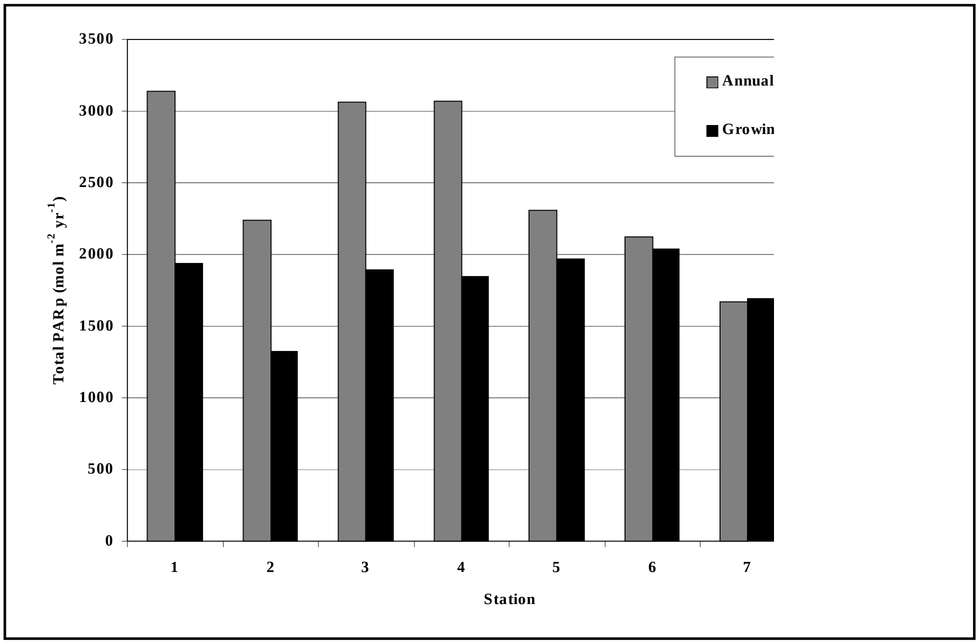

Figure 6. Total quanta of PAR received by seagrass blades (PARp) during a year of continuous sampling and during the April-September growing season.

(Sarasota Bay) versus more frequent samplings.

The potential range in epiphytic attenuation can be applied to water column light levels measured in Tampa Bay to estimate the impacts to %PARp. Light targets of 20.5 %PARw with 34% epiphytic attenuation result in %PARp values near 13.5%. If instead, epiphytic attenuation in the upper Bay is nearer 55%, then %PARp is only 9.2%, less than three quarters of that required to maintain plants in Tampa Bay, and certainly insufficient to permit seagrass expansion to deeper depths (Figure 5). In this instance, while water column light targets may be achieved, the PAR available to the plant may still be insufficient if nutrient-sponsored epiphytic growths exceed those measured in lower Tampa Bay. Ongoing work by the Fish and Wildlife Commission-Florida Marine Research Institute is addressing this critical knowledge gap but these data have not yet been incorporated in the evaluation of existing light levels with regard to targets.

TOTAL LIGHT RECEIVED vs %PARw

The slight differences between annual %PARw received at the site with shading response and the remaining three in Tampa Bay raises another issue with respect to the robustness of %PARw as an ecological management tool. The difference in %PARw between 'effect'–'no effect' is very slight (<2%) and approaches the analytical precision of both the light target and the continuous measurement process itself. (This is why data from all four Tampa Bay sites were included in target establishment.) If PARw is presented as total photons received at the deep edge throughout the year (mol m^{-2} yr^{-1}) rather than as a percentage, then differences between sites are much more apparent. The three stable sites of Tampa Bay received approximately 4,800 mol m^{-2} yr^{-1}, while the site with shading responses received only about three-quarters, or near 3,600 mol m^{-2} yr^{-1}. If epiphytic attenuation is used to compute the total PAR received by the plant, and if growing season totals from April to September are further examined, then the agreement between Tampa Bay and Charlotte Harbor sites improved even more with seagrass blades at most locations receiving near 2000 mol m^{-2} $season^{-1}$ (Figure 6). While continuous light measurements are not a convenient monitoring tool, the concordance between both total light received and biomass trends at individual sites does indicate that total photons measured in both this study and others were an ecologically relevant quantity.

SUMMARY

Discrete monitoring, even weekly, can produce annual %PARw values very different from contemporaneous continuous monitoring. Differences can be greater than 5%, substantive when water column light targets are 20.5%, and when a shading response appeared triggered by a reduction of 2% from target values. Although unlikely to be an effective monitoring tool in the near future, continuous data as total quanta received appears a more sensitive and ecologically relevant tool for predicting biomass outcomes. Additional information is needed on seasonally critical periods and light levels required during these periods. This need would be best addressed by an energy model incorporating light and temperature dependent photosynthesis and metabolic functions, including both normal losses and the biomass increase needed to sponsor seagrass expansion. Most importantly, epiphytic attenuation further reduces water column light and can vary regionally in response to salinity (nutrients). Using a water column light target developed under low epiphytic loads (as the Tampa target was) is not appropriate in regions where epiphytic cover is expected to be higher.

ACKNOWLEDGEMENTS

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IMPLEMENTING THE TAMPA BAY SEAGRASS RESTORATION MANAGEMENT STRATEGY

Holly Greening

ABSTRACT

Participants in the Tampa Bay Estuary Program have agreed to adopt nitrogen loading targets for Tampa Bay based on the water quality and related light requirements of *Thalassia testudinum*. Based on modeling results, it appears that light levels can be maintained at necessary levels by "holding the line" at existing nitrogen loadings. However, this goal may be difficult to achieve given the 20% increase in the watershed's human population and associated 7% increase in nitrogen loading that are projected to occur over the next 20 years.

To address the long-term management of nitrogen sources, a Nitrogen Management Consortium of local electric utilities, industries and agricultural interests, as well as local governments and regulatory agency representatives, has developed a Consortium Action Plan to address the target load reduction needed to "hold the line" at 1992–1994 levels. To date, implemented and planned projects collated in the Consortium Action Plan meet and exceed the agreed-upon nitrogen loading reduction goal.

INTRODUCTION

In 1990, Tampa Bay was accepted into the U.S. Environmental Protection Agency's (USEPA) National Estuary Program. The Tampa Bay National Estuary Program (TBNEP), a partnership that includes three regulatory agencies and six local governments, has built on the resource-based approach initiated by earlier bay management efforts. Further, it has developed water quality models (Zarbock et al. 1994; Janicki and Wade 1996; Martin et al. 1996; Zarbock et al.1996; Morrison et al. 1997; Wang et al. 1999; Janicki, this volume) to quantify linkages between nitrogen loadings and bay water quality, and models that link loadings and water quality to seagrass goals (Janicki and Wade 1996; Greening et al. 1996; Johansson and Greening 2000; Janicki, this volume).

Recent recommendations from the National Academy of Science National Research Council (NRC 2000) include those which regional watershed programs might consider in developing nutrient management strategies. The NRC recommendations are based on the process designed by the Tampa Bay Estuary Program partners to develop and implement a seagrass protection and restoration management program for Tampa Bay. Critical elements of the Tampa Bay process are to:

1. Set specific, quantitative seagrass coverage goals for each bay segment.
2. Determine seagrass water quality requirements and appropriate nitrogen loading targets.
3. Define and implement the nitrogen management strategies needed to achieve the load management targets.

The technical basis of the first two steps is more fully described in Janicki and Wade (1996) and Johansson and Greening (2000). The third step, define and implement the nitrogen management strategies needed to achieve the load management targets, is described in more detail below.

STEP 1. SET QUANTITATIVE RESOURCE MANAGEMENT GOALS

The establishment of clearly defined and measurable goals is crucial for a successful resource management effort. The TBNEP Management Conference adopted the initial goal to increase the current Tampa Bay seagrass cover to 95% of that present in 1950 (TBNEP 1996).

Based on digitized aerial photographic images, it was estimated that approximately 16,500 ha of seagrass existed in Tampa Bay in 1950 (Lewis et al. 1991). At that time, seagrasses grew to depths of 1.5 m to 2 m in most areas of the bay. By 1992, approximately 10,400 ha of seagrass remained in Tampa Bay (Janicki and Wade 1996), a loss of more than 35% since the 1950 benchmark period. Some (about 160 ha) of the observed loss occurred as the result of direct habitat destruction associated with the construction of navigation channels and other dredging and filling projects within existing seagrass meadows, and is assumed to be non-restorable through water quality management actions.

In 1996, the TBNEP adopted a bay-wide minimum seagrass goal of 15,400 ha. This goal represented 95% of the estimated 1950 seagrass cover (minus the non-restorable areas), and includes the protection of the existing 10,400 ha plus the restoration of an additional 5,000 ha (TBNEP 1996).

STEP 2. DETERMINE SEAGRASS WATER QUALITYREQUIREMENTS AND APPROPRIATE NITROGEN LOADING RATES

Once the seagrass restoration and protection goal was established by the participants, the next steps established the environmental requirements necessary to meet the agreed-upon goal and subsequent management actions necessary to meet those requirements. Elements of this process included the following, and are more fully described in Johansson and Greening (2000) and Janicki and Wade (1996).

A. Determine environmental requirements needed to meet the seagrass restoration goal: Recent research indicates that the deep edges of *Thalassia testudinum* meadows, the primary seagrass species for which nitrogen loading targets are being set, correspond to the depth at which 20.5% of subsurface irradiance (the light that penetrates the water surface) reaches the bay bottom on an annual average basis (Dixon and Leverone 1995). The long-term seagrass coverage goal can thus be restated as a water clarity and light penetration target. Therefore, in order to restore seagrass to near 1950 levels in a given bay segment, water clarity in that segment should be restored to the point that allows 20.5% of subsurface irradiance to reach the same depths that were reached in 1950.

B. Determine water clarity necessary to allow adequate light to penetrate to the 1950 seagrass deep edges: Water clarity and light penetration in Tampa Bay are affected by a number of factors, such as phytoplankton biomass, non-phytoplankton turbidity, and water color. Janicki and Wade (1996) used regression analyses, based on long-term data provided by the EPC, to develop an empirical model describing water clarity variations in the four largest bay segments (Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay, and Lower Tampa Bay).

Water color may be an important cause of light attenuation in some bay segments; however, including color in the regression model did not produce a significant improvement in the predictive ability of the model. Results of the modeling effort indicate that, on a baywide basis, variation in chlorophyll *a* concentration is the major factor affecting variation in average annual water clarity.

C. Determine chlorophyll *a* concentration targets necessary to maintain water clarity needed to meet the seagrass light requirement: The empirical regression model was used to estimate chlorophyll *a* concentrations necessary to maintain water clarity needed for seagrass growth for each major bay segment. The adopted segment-specific annual average chlorophyll *a* targets (8.5 g/l for Old Tampa Bay, 13.2 g/l for Hillsborough Bay, 7.4 g/l for Middle Tampa Bay, and 4.6 g/l for Lower Tampa Bay) are easily measured and tracked through time, and are used as intermediate measures for assessing success in maintaining water quality requirements necessary to meet the long-term seagrass goal.

D. Determine nutrient loadings necessary to achieve and maintain the chlorophyll *a* targets: Water quality conditions in 1992–1994 appear to allow an annual average of more than 20.5% of subsurface irradiance to reach target depths (i.e., the depths to which seagrasses grew in 1950) in three of the four largest bay segments (Hillsborough Bay, Old Tampa Bay and Lower Tampa Bay). Water quality in the Middle Tampa Bay segment now allows slightly less than 20.5% to target depth. Thus, a management strategy based on "holding the line" at 1992–1994 nitrogen loading rates should be adequate to achieve the seagrass restoration goals in three of the four segments. This "hold the line" approach, combined with careful monitoring of water quality and seagrass extent, was adopted by the TBNEP partnership in 1996 as its initial nitrogen load management strategy.

As an additional complicating factor, a successful adherence to the "hold the line" nitrogen loading strategy may be hindered by the projected population growth in the watershed. A 20% increase in population, and a 7% increase in annual nitrogen load, are anticipated by the year 2010 (Zarbock et al. 1996). Therefore, if the projected loading increase (a total of 17 U.S. tons per year) is not prevented or precluded by watershed management actions, the "hold the line" load management strategy will not be achieved.

STEP 3.
DEFINE AND IMPLEMENT NITROGEN MANAGEMENT STRATEGIES NEEDED TO ACHIEVE LOAD MANAGEMENT GOALS

Local government and agency partners in the TBNEP signed an Intergovernmental Agreement (IA) in 1998 pledging to carry out specific actions needed to "hold the line" on nitrogen loadings. The IA includes the responsibility of each partner for meeting the nitrogen management goals, and a timetable for achieving them. How those goals are reached will be left up to the individual communities as defined by them in their Action Plans (TBEP 1998a). The Tampa Bay National Estuary Program was also renamed the Tampa Bay Estuary Program as part of progression from the planning phase to implementation of the adopted Comprehensive Conservation and Management Plan.

To maintain nitrogen loadings at 1992–1994 levels, local government Action Plans address that portion of the nitrogen target which relates to non-agricultural stormwater runoff and municipal point sources within their jurisdictions, a total of 6 U.S. tons of nitrogen per year through the year 2010 (Table 1).

To address the remaining 11 U.S. tons of nitrogen of the 17 total per year each year through the year 2010 needed to "hold the line" (attributed to atmospheric deposition, industrial and agricultural sources and springs; Janicki and Wade 1996), a Nitrogen Management Consortium of local electric utilities, industries and agricultural interests, as well as the local governments and regulatory agency representatives in the TBEP, was established (Table 2). The Nitrogen Management Action Plan developed by the public and private partners in the Consortium combines for each bay segment all local government, agency and industry projects that will contribute to

Table 1. Tampa Bay nitrogen management goals; cumulative 1995–1999 goals for nitrogen management/ reduction (tons).

	SOURCE CATEGORY						
	OTB	HB	MTB	LTB	BCB	TOTAL	PERCENT
Pinellas County	0.30	<0.01	<0.01	<0.01	0.85	1.15	1.4
City of Clearwater	0.20	<0.01	<0.01	<0.01	<0.01	0.20	0.2
City of St. Petersburg	0.05	<0.01	0.90	<0.01	1.05	2.00	2.4
Hillsborough County	0.40	4.75	2.50	<0.01	<0.01	7.65	9.1
City of Tampa	0.10	8.45	<0.01	<0.01	<0.01	8.55	10.2
Manatee County	<0.01	<0.01	0.50	8.35	<0.01	8.85	10.6
TBNM Consortium	1.05	28.25	7.15	17.00	2.00	55.45	66.1
TOTAL (reduction in annual load)	2.10	41.50	11.05	23.35	3.90	83.85	100.0

OTB = Old Tampa Bay; HB = Hillsborough Bay; MTB = Middle Tampa Bay; LTB = Lower Tampa Bay; BCB = Boca Ciega Bay

meeting the five year nitrogen management goal (TBEP 1998b).

To ensure that each partner was using similar nitrogen load reduction assumptions for similar projects, guidelines for calculating nitrogen load reduction credits were developed with the partners (Zarbock and Janicki 1997), and were used by each of the partners in the development of their action plans. Three methods for estimating nutrient reduction from a specific project or action are recommended:

- use project-specific estimates of nitrogen load reduction to determine credits, when available;
- use literature values of BMP treatment efficiencies to determine credit; or
- use the TBNEP Optimization Model (Wade et al. 1996) to determine nitrogen load reduction credits.

Project-specific estimates are most frequently generated for stormwater treatment projects by the entity constructing the project. Literature values from Florida and other locations have been collated in Zarbock and Janicki (1997). The Optimization Model is an automated protocol for determining selection of stormwater BMPs for a given locale. For this procedure, design criteria of a proposed BMP would be entered into the model and the model would estimate a potential load reduction based on the input variables. Model construction and application is fully described in Wade et al. (1996).

Table 2. Public and private partners of the Tampa Bay Nitrogen Management Consortium, July 2001.

Public Partners:

- City of Tampa
- City of Clearwater
- City of St. Petersburg
- Manatee County
- Hillsborough County
- Pinellas County
- Manatee County Agricultural Extension Service
- Environmental Protection Commission of Hillsborough County
- Tampa Bay Regional Planning Council
- Florida Department of Environmental Protection
- Florida Fish and Wildlife Commission/Florida Marine Research Institute
- Southwest Florida Water Management District
- U.S. Army Corps of Engineers
- U.S. Environmental Protection Agency
- Tampa Port Authority
- Tampa Bay Estuary Program
- Florida Department of Agriculture and Consumer Services

Private Partners:

- Florida Phosphate Council
- Florida Power & Light
- Tampa Electric Company
- Florida Strawberry Growers Association
- IMC-Phosphate
- Cargill Fertilizer, Inc.
- CF Industries, Inc.
- Pakhoad Dry Bulk Terminals
- Eastern Associated Terminals Company
- CSX Transportation

The types of nutrient reduction projects included in the Consortium's Nitrogen Management Action Plan range from traditional nutrient reduction projects such as stormwater upgrades, industrial retrofits and agricultural best management practices to actions not primarily associated with nutrient reduction, such as land acquisition and habitat restoration projects. A total of 105 projects submitted by local governments, agencies and industries are included in the Plan; 95% of these projects address nonpoint sources and account for 71% of the expected total nitrogen reduction. Half (50%) of the total load reduction will be achieved through public sector projects, and 50% by industry.

Table 3 summarizes expected reductions from those projects which were completed by the end of 1999. A total of 134 tons per year reduction in nitrogen loading to Tampa Bay is expected by the end of 2000, which exceeds the 1995–1999 reduction goal of 84 tons per year by 60%. Old Tampa Bay, Hillsborough Bay and Middle Tampa Bay are expected to meet (and exceed) the Year 2000 Nitrogen Management Goal with completed and ongoing projects alone, and Lower Tampa Bay and Boca Ciega Bay are expected to meet (and exceed) the load reduction goal with ongoing and pending projects. An updated estimate of nitrogen loadings to the bay from all sources was initiated by TBEP in 2000, after which the effectiveness of the proposed projects in maintaining loads to the bay will be will be evaluated.

Examples of specific projects and expected nitrogen loading reductions are described in the Tampa Bay Nitrogen Management

Table 3. Tampa Bay Nitrogen Management Consortium summary of goals and expected reductions; cumulative tons TN reduced or precluded/year by the year 1999.

BAY SEGMENT	1995–1999 NITROGEN REDUCTION GOAL	EXPECTED REDUCTION: COMPLETED OR ONGOING PROJECTS[1]	PENDING PROJECTS[2]	ATMOSPHERIC DEPOSITION[3]
Old Tampa Bay	2.10	3.6	1.5	3.6 - 6.2
Hillsborough Bay	41.5	62.0	3.9	13.9 - 24.0
Middle Tampa Bay	11.1	14.8	6.3	4.6 - 7.9
Lower Tampa Bay	25.4	25.0	11.2	5.7 - 10.0
Boca Ciega Bay	3.9	0.8	4.8	1.2 - 2.1
Total	84.0	106.2	27.7	29.0- 50.2

[1] Projects have been completed or are under construction (1999). These summaries do not include reductions expected from TECO atmospheric deposition reductions.
[2] Projects have funding available and are in the planning or permitting stages (1999). These summaries do not include reductions expected from TECO atmospheric deposition reductions.
[3] Range of atmospheric deposition reductions expected from TECO, as estimated by EPCHC (first estimate) and TECO (second estimate).

Consortium Action Plan 1995–1999 (TBEP 1998b) and include the following:

- **Stormwater facilities and upgrades:** Stormwater improvements or new facilities include both public and private examples. A stormwater retrofit using alum injection in East Lake (Hillsborough County and SWFWMD) is expected to reduce nitrogen loading by almost 2.8 tons per year. Stormwater improvements in Pinellas County and Clearwater (including Lower Sweetwater Creek Regional Stormwater Facility, Alligator Creek, Safety Harbor and Roosevelt Creek stormwater upgrades) will eliminate an estimated 2 tons of TN loading per year. The Lake Maggiore alum injection project is expected to reduce an estimated 3.6 tons per year TN loading. Industrial stormwater improvements at CSX Transportation Rockport Terminal, CF Industries, IMC-Phosphates, Eastern Associates Terminals and Cargill Fertilizer are expected to reduce almost 20 tons TN loading per year by the year 2000.

- **Land acquisition and protection:** Land acquisition and maintenance of natural or low intensity land uses precludes higher density uses and higher rates of TN loading. Land acquisition by Hillsborough County's Environmental Land Acquisition Program (ELAP), Manatee County and the Southwest Florida Water Management District precluded more than 15 tons TN loading per year by the end of 1999. Approved overlay districts requiring additional nutrient control in management areas of Manatee County are expected to preclude an additional 10 tons per year TN loading.

- **Wastewater reuse:** The Plant City Reuse Program in Hillsborough County is expected to result in a 6.4 ton per year reduction on TN loading. Conversion of septic systems to sewer in Largo (Pinellas County) will reduce TN loading by 1.7 tons per year.

- **Emissions reduction:** Tampa Electric Company and the Environmental Protection Commission of Hillsborough County (EPC) used two different methods to estimate emissions reduction from TECO's Big Bend and Gannon Plants between 1995 and 1997, resulting in reductions of NO_x emissions of 11,700 tons (EPC's estimate) or 20,000 tons (TECO's estimate). To estimate the reduction of nitrogen deposition which reaches the bay (either by direct deposition to the bay's surface, or by deposition and transport through the watershed), a 400:1 ratio (NO_x emissions units to nitrogen units entering the bay) is assumed. Expected reductions from atmospheric deposition thus ranged from 29 to50 tons per year by 1999. TECO and EPC agree that ultimately the same assumptions and database should be used to calculate emissions and estimated reductions, and are working toward that end. To date, emissions reductions have not been included in the estimated total TN reduction to the bay, pending agreement on methods.

- **Habitat restoration:** Although typically conducted for reasons other than nutrient reduction, habitat restoration to natural land uses reduces the amount of TN loading per acre in runoff. Habitat restoration projects have been completed or are underway in all segments of Tampa Bay's watershed, and are being conducted by SWFWMD, FDEP and the cities and counties participating in TBEP. Several of the larger projects (and estimated TN load reductions) include: Delaney Creek and East Lake restoration projects (a total reduction of an estimated 5.7 tons TN per year) in the Hillsborough Bay drainage area; Del Oro, Coopers Point and Allens Creek restoration projects in the Old Tampa Bay drainage area, for a total reduction of 0.24 ton per year; Cockroach Bay restoration, a reduction of 0.33 ton TN per year in the Middle Tampa Bay drainage area; and Jungle Lake and Clam Bayou restoration projects in the Boca Ciega Bay drainage area, for a total reduction of 0.84 ton per year.

- **Agricultural BMPs:** Water use restrictions have promoted the use of microjet or drip irrigation on row crops (including winter vegetables and strawberries) and in citrus groves. Micro-irrigation has resulted in potential water savings of approximately 40% or more over conventional systems and an estimated 25% decrease in fertilizer applied. Nitrogen reduction estimates from these actions total 6.4 TN tons per year.

- **Education/public involvement:** For those projects for which nitrogen load reductions have not been calculated or measured, but some reductions are expected, the Consortium Action Plan assumes a 10% reduction estimate until more definitive information is available. Public education programs include Florida Yards & Neighborhoods and the associated Florida SchoolYard Program, Hillsborough County's Adopt-a-Pond, and LAKEWATCH programs. These programs have reduced TN loading by an estimated 2 tons per year.

- **Industrial upgrades**: IMC-Agrico Co. (now IMC-Phosphate) terminated the use of ammonia in flot-plants (an element of the fertilizer manufacturing process), resulting in a reduction of 21 tons per year of nitrogen loading. CF Industries upgraded their product conveyor systems, resulting in a TN reduction of more than an estimated 10 tons per year due to control of fertilizer product loss. The termination of discharge by Tropicana into the Manatee River is expected to result in a reduction of more than 11 tons per year TN loading.

The approach advocated by the TBEP stresses cooperative solutions and flexible strategies to meet nitrogen management goals. This approach does not prescribe the specific types of projects that must be included in the Action Plan; Consortium partners have been encouraged to pursue the most cost-effective options to achieve the agreed-upon goals for nitrogen management. The TBEP will review and revise nitrogen management goals every five years, or more often if significant new information becomes available.

SUMMARY

The Tampa Bay management community has agreed that the protection and restoration of the Tampa Bay living resources is of primary importance and through the TBEP process (initiated in 1991) has adopted nitrogen loading targets for Tampa Bay based on the water quality requirements of *Thalassia testudinum* and other native seagrass species. A long-term goal has been adopted to achieve 15,400 ha of seagrass in Tampa Bay, or 95% of that observed in 1950. To reach the long-term seagrass restoration goal, a 7% increase in nitrogen loading associated with a projected 20% increase in the watershed's human population over the next 20 years must be offset. Government and agency partners in the Tampa Bay Estuary Program and private industries and interests participating in the Nitrogen Management Consortium have identified and committed to specific nitrogen load reduction projects to ensure that the water quality conditions necessary to meet the long-term living resource restoration goals for Tampa Bay are achieved.

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SEAGRASS TRANSPLANTING AND RESTORATION IN TAMPA BAY

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ABSTRACT

Reports of seagrass decline worldwide have been occurring for many years. The decline has been attributed to dredge and fill, pollution, and propeller scarring. Past techniques to restore seagrass beds have not always been successful with many transplanting projects having only a 20% survival rate. New techniques for seagrass recovery are presented that include: a seagrass formula for regrowth into prop scars, an injecting boat that can inject nutrients into prop scars, and a planting boat that can plant short shoots of *Halodule wrightii* in bare root or in peat pots. The planting boat can also plant seedlings of *Thalassia testudinum* grown from seeds. The injection system for regrowth of seagrasses into prop scars yielded results for *Thalassia testudinum* as high as 68% of complete recovery in 15 months and for *Halodule wrightii* at 76% of complete recovery in 15 months.

Seagrass Loss in Florida

Seagrass loss in Florida has occurred for many years and has been extensive. The primary reasons for seagrass loss have been dredge and fill, changes in water quality and propeller damage. Examples of seagrass loss in Florida as reported by Sargent et al. (1994) were as follows:

- Florida has lost about 2 million acres of seagrasses.
- Indian River Lagoon has lost 30%.
- Charlotte Harbor has lost 29%.
- Tampa Bay has lost 80%.
- Ponce Deleon Inlet has lost 100%.
- St. Andrews Bay has lost 30%.
- Florida Bay has lost 60,000 acres since 1987.

The presence of prop scars in seagrass meadows has been a concern for a number of years. Sargent, et al (1994) reported that statewide there are about 63,989 acres of seagrass that have moderate to severe propeller damage. The most heavily scarred areas are in Monroe, Lee, Dade, Pinellas, and Charlotte counties. Monroe County leads the list with over 15,000 acres of moderate to severe damage.

The primary question about prop scar damage is: what can be done about it? The answers are varied and based upon community approaches. Some of the techniques are:

- Some communities have decided just to wait and let the seagrasses regrow on their own.

- Pinellas and Hillsborough Counties have decided to shut down some seagrass beds to boaters. They have erected signs marking seagrass beds that are restricted. Both counties have also set up zones for “seagrass caution.” They have erected markers to warn boaters of the presence of seagrasses and to show caution. Ehringer (2000) compared restricted zones, caution zones and unrestricted zones in seagrass beds for propeller damage in Pinellas County for an eight-year period. Damage to seagrasses from boats was not significantly different from restricted zones to caution zones. The unrestricted zones were more heavily damaged. A conclusion of this report is that if signs are present, whether restricted or caution, less damage occurs in the seagrass beds form boaters.

- In the Florida Keys one approach to regrowth into prop scars was to erect bird roosting posts. The posts were erected in the prop scars. Birds roosted on the posts and their excrement fertilized the seagrasses below.

- Experiments have also been conducted by Ehringer (2000) and others, to plant seagrasses into the prop scars and wait for recovery. This technique produced a recovery of only about 20% to 40% in a 2-year period.

- Another technique is to inject nutrients and plant growth regulators directly into the prop scars to stimulate the growth of seagrasses on the margins of the prop scar. Once stimulated, the seagrasses regrow back into the prop scar. This technique has been tested by Ehringer and Dawes (Dawes, 1997).

Recovery of Seagrasses from Prop Scar Damage

Recovery of seagrasses into prop scars can take many years. Zieman (1976) stated that *Thalassia testudinum* (turtle grass) might require at least two years to begin to recolonize a prop scar. Durako et al. (1992) stated that recovery to normal density would take from 3.6 to 6.4 years. Others have stated that complete seagrass recovery may take as long as ten years (Lewis and Estevez, 1988).

One of the reasons postulated for the slow regrowth of *Thalassia testudinum* into prop scars is the slow appearance of a new apical meristem on the rhizome (Dawes et al. 1997). Formation of a new apical meristem may take as long as 10 months (Kelly et al. 1971). However, this only partially explains the slow regrowth. Ehringer in 1993 speculated that the loss of the anaerobic bacteria in the sediment due to the stirring effect of the prop was part of the problem. Perhaps the bacteria were a source of nutrients to the roots and rhizomes. It may take a period of time to regrow the bacteria colonies and, subsequently, the resulting nutrient load in the sediment.

Seagrass planting programs have not been very successful. Fonseca, Kenworthy and Thayer (1987) reported results of an extensive comparison of seagrass transplanting techniques. In one of their planting projects all manatee grass was gone and only 30% of the shoal grass had survived. In another planting project in Tampa Bay by Fonseca, Kenworthy and Courtney, 14 plots of seagrasses were planted with 6 plots losing 95% to 100% of their plants. Another 5 sites were lost later. Fonseca, Kenworthy, Courtney and Hall reported in 1994 on another transplanting project in which 40% of transplants of *Syringodium filiforme* survived. All of their transplants of *Halodule wrightii* failed to survive. Techniques have been tried using peat pots, bare root, staples, and a variety of other methods (Breedveld 1975; Fonseca 1987). Yet the survival of the transplants remains low (Fonseca 1996).

Injection of Prop Scars

Experiments have been conducted by Ehringer in Tampa Bay since 1993 working with fertilizers and plant growth regulators in an attempt to regrow seagrasses into prop scars. Experiments were conducted in Cockroach Bay, a small bay attached to Tampa Bay, and at the Fort DeSoto Aquatic Preserve (Pinellas County) from 1993 through 2000. From 1997 to 2000 the manner of injection was an injector boat built by James Anderson of Ruskin, Florida.

A seagrass formula was arrived at by a series of small experiments conducted at Cockroach Bay and at the Department of Environmental Protection's fish hatchery from 1994 to 1997. Small tanks were set up at the hatchery with marine sediments and flow-through salt water. Short shoots of *Thalassia testudinum* were grown in the tanks. A series of experiments were conducted using a variety of plant growth regulators supplied by Abbott Laboratories, including auxins, cytokinins and gibberellins to determine if apical meristem growth could be initiated on a cut rhizome. Only one combination of chemicals induced apical meristem growth. A mixture of 2 ml gibberellic acid and 2 ml 6-benzyladenine injected near the cut rhizome caused apical meristems to grow on the rhizomes in 12 of 15 plants within 4 weeks. No apical meristems grew on the controls or any other combination of plant growth regulators. Further experiments were conducted to determine whether urea or nitrate was an optimal source of nitrogen for the rhizomes of *Thalassia testudinum*. Comparisons were made between the sediments of a prop scar and those of a healthy seagrass bed. The largest difference between the two was that of nitrogen. Ammoniac nitrogen was about 60% less in prop scar sediments than in healthy seagrass sediment. Experiments conducted with Clinton Dawes determined that the use of nitrates produced no effect on growth (Dawes 1997). Therefore, a series of small experiments was conducted at Cockroach Bay to determine if urea could be an adequate source of nutrients for *Thalassia testudinum*. Urea was injected around prop-cut rhizomes in concentrations of 2 ml, 4 ml, 6 ml, and 10 ml dissolved in 10 ml of seawater. Injections were made by hand, one injection every two weeks for 4 injections. Seagrasses regrew into the prop scars in all of the concentrations above that of the controls. More growth was evident in the 10 ml concentration. This series of experiments led to the formulation of urea, gibberellic acid and 6-benzyladenine as a formula that could induce growth of *Thalassia testudinum* into damaged seagrass beds.

James Anderson built a special boat and injection system in the winter of 1997. The boat was designed to inject a nutrient formula as mentioned above into the seagrass sediments. The machine has a series of small injectors that introduce about 10 ml of formula into the sediment. The injectors are set in a circular pattern around a wheel (Fig. 1). As the wheel rolls along the bottom, the nutrients are injected into the sediment with a force of about 20 pounds per square inch. Injections only occur when the nozzles are pointed down.

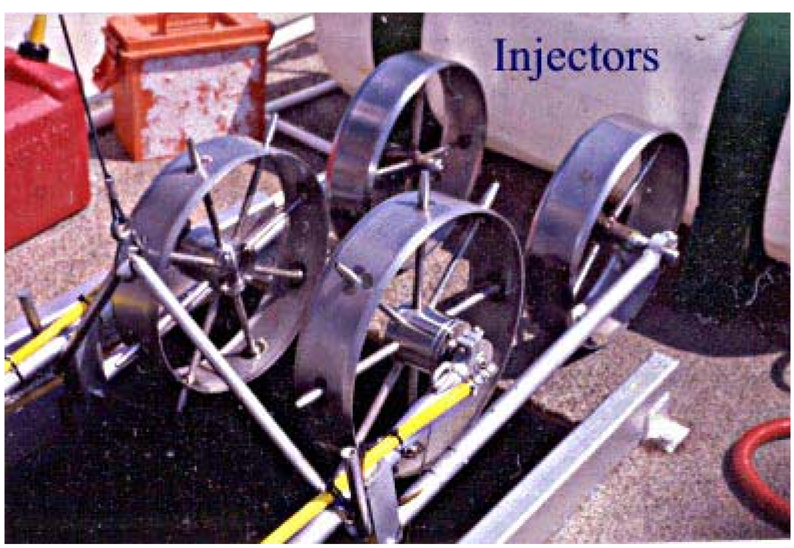

Figure 1. Injectors on Anderson's injector boat.

There are two wheels that roll along the sediment parallel to each other injecting at the same time. The boat contains a 100-gallon tank for mixing of the formula. The formula is pumped from the tank through a series of tubes down to the injectors.

A mixture of 100 pounds of prilled urea 44% (purchased from Scott's) plus 2 ounces of synthetic cytokinin (6-benzlyadenine) and 2 ounces of a gibberellic acid (donated by Abbott Laboratories) was mixed in 100 gallons of seawater. The seawater was obtained from the bay at the site of the injections. Injections were placed into the sediment about every 20 centimeters along designated prop scars. Injections were made at Fort DeSoto beginning in May of 1997 and extended for 5 injections approximately every 7 to 14 days. An additional injection was made in May of 1998. Injections were also made in Cockroach Bay in the same manner.

A counting method was established to determine the number of new shoots that would potentially grow into the prop scars after the injections. A one-meter square PVC frame was made to place over PVC stakes placed in a square pattern over a prop scar (Fig. 2). The center portion of the frame had a 1 m × 22 cm frame inside the larger frame that fit over the prop scar lengthwise.

This frame was set over the stakes at each counting period so that the seagrass shoots could be counted each time in the same manner. Six sites were established for counting in each prop scar, with up to 10 scars per study. Sites were set up as controls in the same manner. The results from the injection sites were as follows:

- In Cockroach Bay experiments were conducted over a 2-year period. Using the injecting boat in this series of experiments with 75 replications *Halodule wrightii* regrew into prop scars at 76% of complete recovery in one season. Complete recovery was determined by setting the frame randomly over established seagrass beds and counting the shoots in the frame. Ten counts were made in *Thalassia testudinum* and 10 counts in *Halodule wrightii*. The numbers were averaged to determine the approximate value of a normal seagrass bed in Fort DeSoto and in Cockroach Bay. Complete recovery

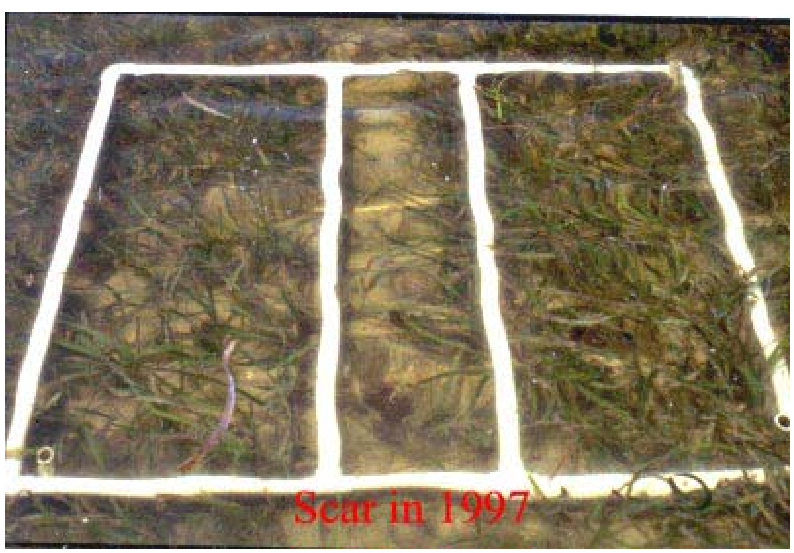

Figure 2. PVC frame set over a prop scar.

of *Thalassia testudinum* would be 70 short shoots per 25 cm by 1 meter. Complete recovery of *Halodule wrightii* would be 200 short shoots. The results of the experiments are listed below. Statistics for all of the experiments show that regrow was significantly greater using the nutrient formula (Table 1).

- In Fort DeSoto at the start of this experiment 9.25 shoots of *Thalassia testudinum* were found in the prop scars (13.2% of complete recovery). The prop scars were injected with the injector boat. After 15 months the average number of shoots was 47.3, or 68% of complete recovery (Table 1). Therefore, prop scar recovery had grown to the point of 68% of a normal seagrass bed in 15 months at Fort DeSoto as a result of the injections.

Further experiments will be conducted in 2001 in Pinellas County on the injection system to verify the recorded growth of seagrasses into prop scars via the injection system.

Mechanical Planting

In early 1998 James Anderson of Ruskin, Florida designed, built and patented a seagrass planting boat (Fig. 3). The pontoon boat floats over the seagrass beds while two workers on the boat "feed" the seagrasses into a planting wheel. The wheel pushes the seagrasses into the sediment at a precise depth so that the rhizomes are about 5 cm into the sediment. Depending upon the number of seagrasses to be planted per acre, the boat has the potential to plant an acre in a day or two using two to three workers. This depends upon the number planted per acre. For example, planting 15,000 shorts per acre would require 2 days, while 8,000 per acre would require one day.

This technique requires fewer workers, is faster than traditional methods, and is less expensive. In addition, since the planter floats over the seagrass beds, there is no damage to the planted seagrasses by workers walking through the beds to plant new seagrasses. The planter has been used on a limited basis in Cockroach Bay, in Fort DeSoto, in the Florida Keys and in Laguna Madre waterway at Corpus Christi, Texas. The system plants *Halodule wrightii* in peat pots or as bare root units. In 1998 and in 1999, 7,850 short shoots of *Halodule wrightii* were planted using the boat in Fort DeSoto. Approximately 48% of the units survived yielding 3,980 square feet of *Halodule wrightii* in a count conducted one year later.

A Seagrass Nursery

A seagrass nursery was established in the field using plants removed from Fred Howard Park. The seagrasses (*Thalassia*

Table 1. Injection results for Cockroach Bay and Fort DeSoto (average number of short shoots per 25 cm × 1 meter).

		MAY 1997		SEP 1997		JULY 1998	
		injected	control	injected	control	injected	control
Cockroach Bay	*Halodule wrightii*[1]	21.5	3.0	45.5	6.5	148	10
	Thalassia testudinum[2]	6.75	4.13	—	—	23.0	9.10
Fort DeSoto	*Thalassia testudinum*[3]	9.25	14.0	40.7	17.3	47.3	19.5

[1]t stat = 23.41, P(T<=) one tail and two-tailed yields less than .0001, t critical one tail = 1.761, SS = 1127.5, $Mean_A$-=108.625, df=0

[2]t stat = 7.00, P(T<=) one tail and two-tailed yields less than .0001, t critical one tail = 1.894, SS = 687.94, $Mean_A$-=11.375, df=14

[3]t stat = 21.14, P(T<=) one tail and two-tailed yields less than .0001, t critical one tail = 1.812, SS = 11023.27, $Mean_A$-=43.63, df=0

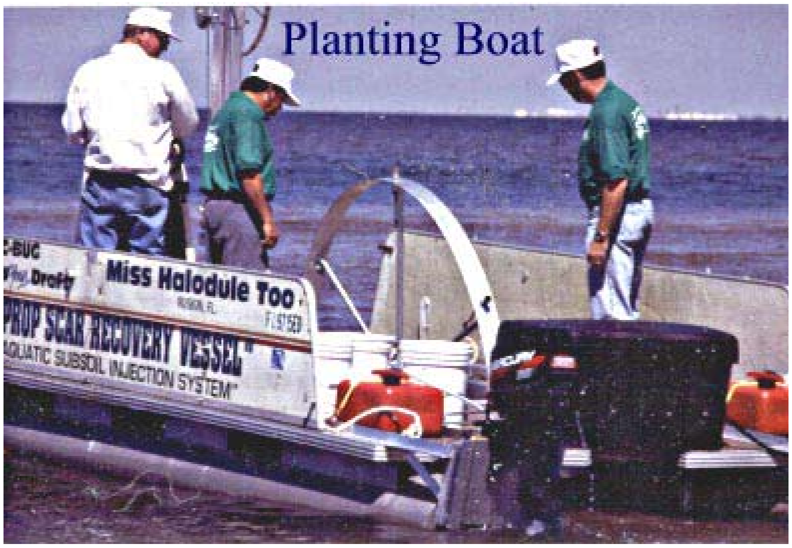

Figure 3. Anderson's planting boat. Note the planting wheel in the center.

testudinum and *Halodule wrightii*) were removed as part of a mitigation plan to widen the beach at the park. Approximately 1,000 square feet of seagrasses were planted at Mangrove Point, near Simmons Park in Tampa Bay. The site was chosen for the following reasons:

1. The site was privately owned. It does not have boat traffic through it.
2. The site had some seagrasses growing in it along an opposite bank; therefore, the water quality was conducive to seagrass planting.
3. The salinity regime was within a normal range for seagrasses.
4. The site has an easy access for boats and the site is shallow.

Beginning on June 18, 1998 and several dates thereafter, seagrasses were planted at the site by hand. Plants were soaked in the seagrass formula of urea, gibberellic acid and 6-benzyladenine prior to planting. Twenty-two boxes were planted on June 18, 1998. About 400 short shoots of *Halodule wrightii* were planted with the planting boat. In August about 650 short shoots of *Thalassia testudinum* and 475 short shoots of *Halodule wrightii* were planted at the site by hand. Two years after planting a visit to the site revealed that the seagrasses had coalesced into a bed of approximately 700 square meters.

Planting Seeds of *Thalassia testudinum*

Seeds of *Thalassia testudinum* typically float inshore in very large amounts in the months of August and September in Monroe and Dade Counties, Florida. About 3,879 seeds were collected in September of 1998. An additional 650 seeds were collected a week later. The seeds were placed in small peat pots with metal disks attached to their base. The units were stored in saltwater upland tanks at a seagrass nursery in Ruskin, Florida and in tanks at Long Key. The seeds were kept at the nursery for about 6 weeks allowing them time to grow into small seedlings. The planting units were transported to the Florida Keys and planted at a site selected by personnel from the Florida Department of Environmental Protection (DEP). The seedlings were planted using the planting boat. The use of peat pots with disks attached proved to be a good choice because the units were set into the sediment in a manner that could not be dislodged by currents.

SUMMARY

Since 1991 a series of experiments has been conducted to attempt to regrow seagrasses into propeller scarred beds. Analyses were made of the sediments of healthy seagrass beds and compared to that of a prop scar. From this series of experiments a formula was derived that replaced the lost nutrients in a prop scar. Following this, another series of experiments was conducted in order to find a formula for a plant growth regulator that would grow an apical meristem on a cut rhizome of *Thalassia testudinum*. The combination of nutrients and plant growth regulators was tested in prop scars in Cockroach Bay and Fort DeSoto. The result showed that about 68% of complete regrowth was possible in one season for *Thalassia testudinum* and 76% for *Halodule wrightii*. The formula was injected into the sediment using an injector boat built by James Anderson. Mr. Anderson also built a seagrass planting boat that was capable of planting up to an acre of seagrasses in one day. The boat has been tested in Tampa Bay, the Florida Keys and in Texas. The planting boat has been used to plant short shoots of *Halodule wrightii* in peat pots and bare root. It has also been used to plant seedlings of *Thalassia testudinum* grown from seeds in the Florida Keys. These new techniques, taken together, offer a wide range of mitigation possibilities for recovery of seagrasses. Seagrasses can be planted in bare areas, prop scars can be recovered, and planting of seagrasses can be accelerated by means of a planting boat.

Future research is needed to refine some of the techniques demonstrated in this study. For example, the technique of using seeds of *Thalassia testudinum*, growing them in peat pots, and planting them with the planting boat needs a rigorous evaluation since no follow-up data is available. The seagrass nursery concept worked very well, but needs to be expanded to other sites, especially an upland nursery. Finally, the use of injections can be further quantified with injections into newly planted sites to determine whether or not growth of newly planted seagrasses will occur more readily with the nutrient additions.

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SEAGRASS SCARRING IN TAMPA BAY: IMPACT ANALYSIS AND MANAGEMENT OPTIONS

Jacob F. Stowers, Eric Fehrmann, Andrew Squires

ABSTRACT

There is little argument when discussing the value of seagrasses. These areas have extremely high productivity and diversity and are frequented by endangered species such as the manatee. Scarring of seagrass beds, mainly by boat propellers, has occurred throughout Tampa Bay and as a result, many groups have taken steps to document the impacts and regulate access within areas of seagrass coverage.

Pinellas County has been active in seagrass protection for over a decade with success in both regulatory and experimental processes. Continued cooperation between a coalition of representatives from government, educational institutions and environmental interest organizations as well as user groups from both the recreational and commercial interests will be required for continued success.

INTRODUCTION

Many studies have been performed documenting the value of seagrasses. These studies have shown the extremely high productivity and diversity of both finfish and shellfish that utilize these areas as both a nursery and refuge. Predator species are naturally drawn to seagrass beds due to the prey species density, which in turn attract sportfishers seeking a challenge. These areas are also frequented by endangered species such as the manatee. It has long been known that scarring of seagrass beds, mainly by boat propellers, has occurred throughout Tampa Bay. As a result, State and local governments, as well as educational institutions have taken steps to document the impacts and regulate access within areas of seagrass coverage.

Documentation of the actual damage incurred can be a costly and labor intensive effort but a combination of aerial photography, photointerpretation, and extensive field verification can result in very accurate estimates of seagrass damage. Seagrass scarring has become more pervasive as more boats are registered and used in the Tampa Bay area. Technical reports by the FDEP-Florida Marine Research Institute indicate that moderate/severe scarring in Tampa Bay averages nearly 30% of the total coverage by seagrass, some of the worst rates in the state (Sargent, et al.1995). Other studies have shown that when scarring becomes severe, the majority of the habitat and water quality functions are lost and the whole bed may lose the ability to regenerate and cease to exist (Sargent, et al. 1995, Ehringer, 1999). Finfish and shellfish production declines, which in turn can severely affect the local commercial harvest economy as well as the recreational fishery.

FORT DESOTO SEAGRASS PROTECTION EFFORTS

Pinellas County became concerned with seagrass scarring and cumulative impacts due to boat propeller scarring in the mid-to-late 1980s. Pinellas County's initiatives began in 1990 and involved a coalition of regulatory and citizen representatives. These included both commercial and recreational fishing interests. Many meetings were held to discuss the issues to build a consensus about a solid action plan that would build support as well as provide the needed resource protection. The group had reached a consensus by the end of 1991 and an ordinance was drafted and adopted in the beginning of 1992 (Ordinance 92-11, since codified under later iterations in the Pinellas County Code).

The Ordinance provided that the management area be divided into zones that:

- eliminated the use of internal combustion engines (exclusion zones)
- allowed use of engines, but imposed penalties for damage to seagrass (caution zones)
- required idle speed (allowed for engine use in exclusion zones to gain access to features such as campsites)
- had no protection (control areas)

The Ordinance provided that the zones be clearly marked (Figure 1) and that the County monitor the zones for 5 years to determine the effectiveness of the management plan. The original ordinance also included a "Sunset Clause" that required it to be renewed each year. This proved to be a non-issue and the sunset clause was removed when the ordinance was renewed in 1993.

The County believed that the best course of action was to take low altitude aerial photographs of the Management Area and then have them digitized and interpreted by a seagrass specialist. Aerials were flown in 1992 prior to installation of signs to provide a "baseline." A second set of aerials was flown in 1992 right after sign installation. Thereafter aerials were flown annually.

Dr. Nicholas Ehringer of Hillsborough Community College (HCC) was retained to digitize the aerials and interpret the results. The scar data were field truthed to provide accuracy. The digitized images were downloaded into the County's Geographic Information System (GIS) (Figure 2).

RESULTS

The scar rate had suffered a large increase prior to the installation of signs in the Caution Zone compared with baseline data. Upon placement of signage in the Management Area, the rate of increase of new scars was considerably reduced in the Caution and Exclusion zones as compared with the control area (Ehringer, 2000). (Note: it was a reduction of the *rate of increase* in the early years, not a reduction in the scar rate.)

The new scar rate remained fairly consistent over the next several years in spite of heavily increased use, apparently due to a proactive public relations campaign and expanded signage at area marinas.

Figure 1. Typical sign located at area boat ramps and marinas.

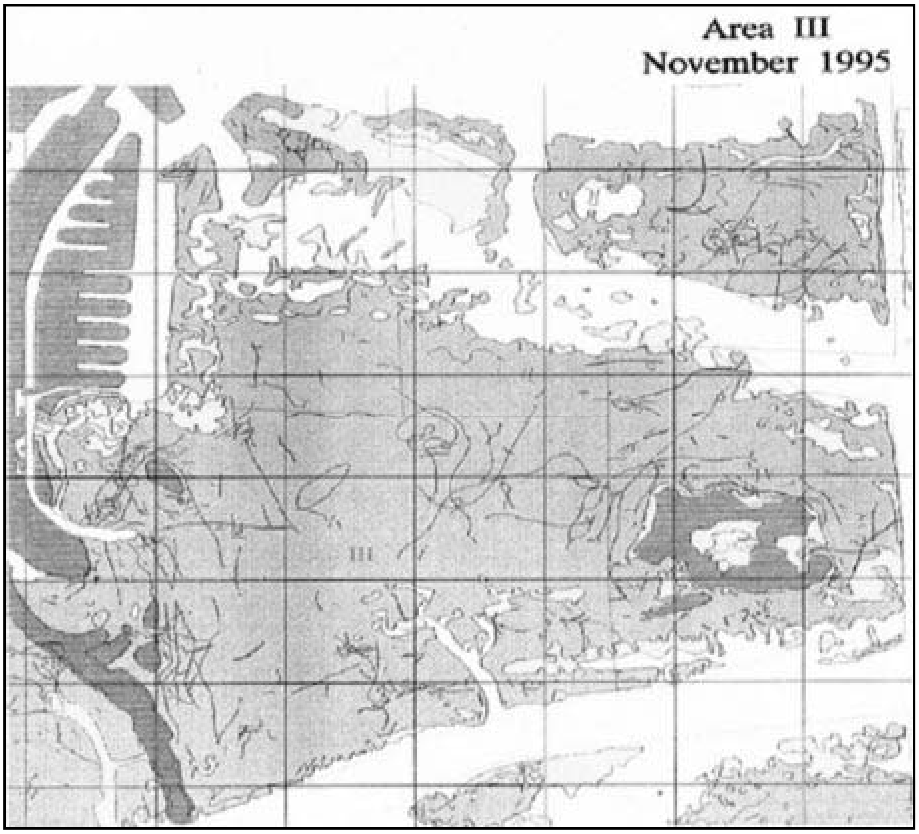

Figure 2. Example map of digitized seagrass scars.

Figure 3. Example of warning sign.

The scar rate in the Caution Zone peaked in 1996 because upwards of 35% of the signs and 50% of the buoys were lost, damaged or relocated due to storms. Buoys disappeared due to anchor failure and many of the signs broke off pilings due to the galvanic reaction between the steel bolts, aluminum signs and bird droppings.

A new sign attachment method and the replacement of buoys with pilings have resulted in a downward shift of the scar rate due to more complete informational coverage. It is believed that the hiring of full time law enforcement officers with shallow draft boats has also reduced the scarring due to ordinance adherence. Both the Caution and Exclusion Zones have experienced similar large reductions in the scar rate. Unfortunately, the scar rate in the unprotected (control) area has continued to rise (Figure 4).

Keys to Ordinance Success

Pinellas County feels that the factors contributing to the success of the program should include efforts to:

- Document the problem thoroughly and highlight the value of the resource. Environmental quality has actual monetary value in addition to its intrinsic value. This can be used to further convince opponents to support the proposed activities.

- Avoid assigning blame and "pointing fingers." Psychological barriers become instantly erected when accusations are leveled at opposing parties. These barriers become increasingly difficult to overcome as discussions progress.

- Get "buy in" from all users. Get public input early and try to incorporate concerns from the users. Fully explain the goals of the program and how these goals will be measured.

- Follow through on "promises" made to users. Failure to perform tasks or agreements will make it nearly impossible to get "buy in" for future projects and could possibly lead to reversal of the ordinance.

- Provide feedback to the users. The public as well as the original members of the team must be kept informed of success or failure of the actions as well as possible future decisions. Use the media to promote effectiveness when possible.

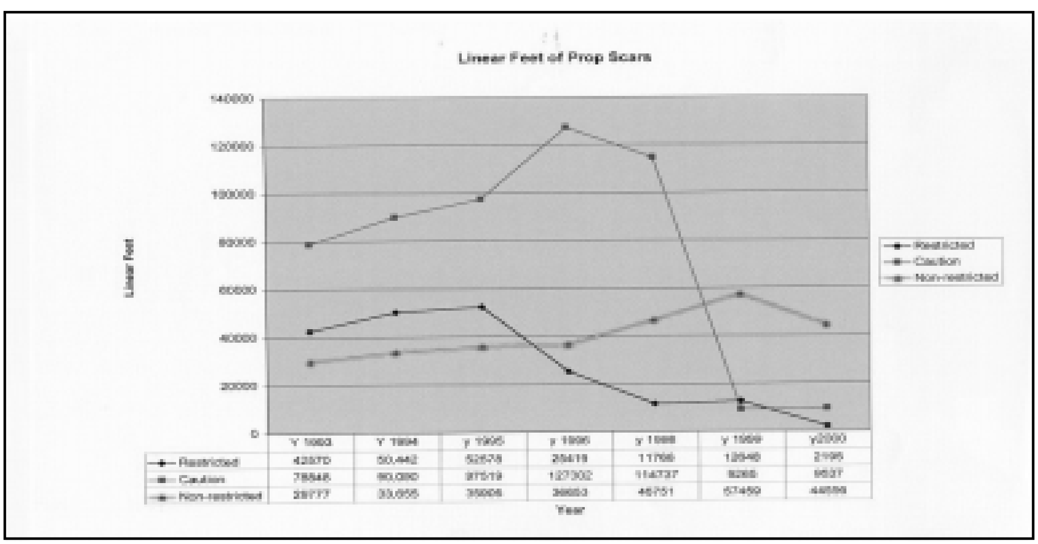

Figure 4. Linear feet of prop scars.

- Adjust the program based on results. Don't be afraid to make changes if the data shows it is the prudent thing to do.

Additional Research

As part of the Howard Park beach renourishment, Pinellas County proposed a replanting/research project as mitigation. The mitigation plan involved the removal of .32 acres of seagrass from Fred Howard Park and the transplanting of the seagrass into the Fort Desoto Management area. The transplanted seagrass was placed in prop scars in order to repair boat propeller damage. The plan had several aspects as follows:

1. Area III of Fort Desoto had 48,365 linear feet of prop scars (.93 acre). In this area nutrients and plant growth regulators were injected into the prop scars to stimulate the growth of new seagrass into existing prop scars without disturbing the grass beds that surrounded the prop scars. Annual photographs of the site taken in the fall of each year were used to ascertain the overall growth of seagrass into the prop scars. In selected sites within the area, small PVC pipes were placed into the prop scars at one-meter intervals. The number of new shoots per meter were compared to linear transects that had not been injected.

2. Approximately 3,000 square feet of seagrass were dug up with sediment from Fred Howard Park and replanted into prop scars at Area V of Fort Desoto. The method of removal involved the digging up of sections of seagrass in squares of 10 inches by 10 inches that included 8 inches of sediment. The seagrass plugs were transported to Fort Desoto in styrofoam boxes and gently placed into prop scars keeping the sediment intact with the rhizomes. For evaluation purposes, transects along the prop scars were set up as in section #1 above.

3. About 10,000 square feet of seagrass at Fred Howard Park were removed by machine. The seagrass was removed from the site with a small backhoe and placed in a strainer to separate the seagrass from the sediment. The seagrass was transported to Fort Desoto in plastic drums that kept the seagrass in fresh marine water. This seagrass was stimulated with plant growth regulators prior to planting by hand in the prop scars. Areas II and VI were the sites for planting the seagrass. The same evaluation system was used for this seagrass as with #1 above.

4. The remaining 939 square feet of seagrass (harvested from floating sprigs) was transplanted into a seagrass nursery that had already been set up in Ruskin. The seagrass in the nursery was stimulated with plant growth regulators to promote new shoots. This seagrass was kept at the nursery and transplanted into sites at Fort Desoto in 1997 and in 1998 into sites where previous plantings had failed.

The results indicated that injection of growth hormone and nutrient into scars where no seagrass was planted was the most effective method of growing seagrass. Seagrass transplanted with sediment was inefficient and had a very low survival rate in this particular situation, and planted sprigs exhibited mixed results (Table 1)(Ehringer, 2000).

Future Directions

It has been recommended in past studies that we expand protection to include areas not currently under protection (the Non-restricted Control Area and the area east of the island of Shell Key). The County Commission approved this additional protection after the presentation at the Seagrass Conference. (Seagrass protection

Table 1. Results of planting methods.

	ORIGINAL AMOUNT	FINAL AMOUNT
Hand transplanted	500 square feet	100 square feet
Sediment transplanted	3,190 square feet	971 square feet
Seagrass planter	3,925 square feet	3,980 square feet
Field nursery	1,000 square feet	500 square feet
Scar injections		26,104 square feet
TOTAL		31,655 square feet

for the Weedon Island Preserve was added with an ordinance amendment in 1996.)

It was also recommended that we reduce the Exclusion Zones and redesignate the areas as Caution Zones based upon the findings that the zones are statistically similar in protecting seagrass. This redesignation was also approved by the County Commission after the presentation at the Seagrass Conference. This action is consistent with our findings that the ordinance success relies on "adjusting the program" and "following through on promises."

Based upon studies, the Board of County Commissioners redesignated some of the zones and added protective zones effective November 2000.

A sign maintenance program and enforcement presence is critical to the long term success of the protection program. Loss of signs was one of the main reasons for non compliance and directly affects the ability of the compliance officers to issue fines for violating the ordinance.

A proactive public information campaign is a key to success. The public in general is much more likely to abide by and support the ordinance if they are well informed of the reasons for the ordinance and can visualize the protection zones.

It is prudent to research and support seagrass planting and restoration efforts to prevent long term problems. It is a goal of Pinellas County to get new seagrass beds established in areas that should support growth based upon favorable growing conditions but where none currently exist.

SUMMARY

To help reduce and avoid seagrass degradation, several local governments have undertaken programs to manage the use of the areas to the benefit of both the citizens and the resource. These programs have generated both controversy and praise. Regulators and political figures are placed in the position of trying to form an alliance of users that are many times at odds with each other. Education and compromise is used as well as persuasive arguments to gain consensus on protecting the resource for the long-term benefit of *all* citizens.

There has been much success in the Tampa Bay area but additional initiatives are required if seagrass beds are to thrive. Recent questions have centered on whether the "exclusion zones" should have been redesignated as "caution zones" and whether the "caution zones" could be expanded to now unprotected areas (redesignation has been approved by the County Commissioners, Figure 5). In addition, the benefits and drawbacks of seagrass scar repair (injections) and the

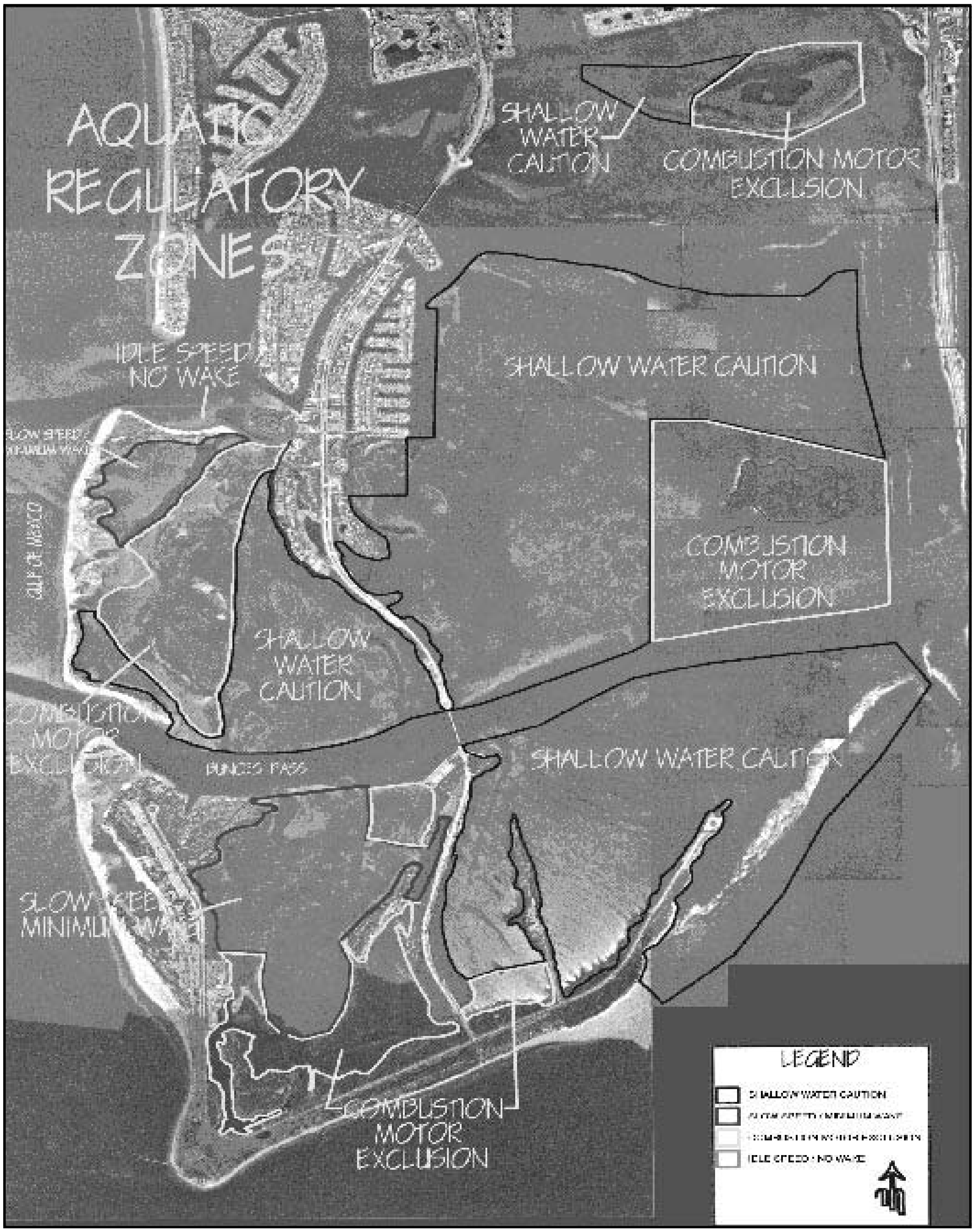

Figure 5. Aquatic regulatory zones.

initiation of new seagrass beds (transplanting) must be addressed. The future approaches to seagrass protection and restoration must be formed by a strong coalition of representatives from government, educational institutions and environmental interest organizations as well as user groups from both recreational and commercial interests.

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SEAGRASS MONITORING ISSUES IN TAMPA BAY

Walter Avery

INTRODUCTION

In the past two decades, the alleviation of eutrophic conditions has resulted in seagrass recolonization and expansion of meadows in many areas of Tampa Bay. As seagrass began to respond to initial water quality improvements during the 1980s, the City of Tampa, Bay Study Group (BSG) and the Southwest Florida Water Management Surface Water Improvement and Management Program (SWIM) instituted programs to monitor changes in the seagrass community. The BSG seagrass program is part of a multi-disciplinary study of the effects of sewage pollution abatement in Hillsborough Bay.

The monitoring programs incorporated methods adapted to meet each agency's data requirements. For example, the BSG has used low altitude aerial photography, on-site GPS measurements, and extensive groundtruthing to generate detailed data on the annual areal coverage and seasonal structure of seagrass meadows in Hillsborough Bay. In contrast, SWIM has relied on interpretation of 1:24,000 scale aerial photographs to generate a seagrass coverage map for Tampa Bay every two years. Following each overflight, SWIM groundtruthed nearly 60 randomly selected sites around the bay to document the accuracy of photographic interpretation.

As seagrass recolonization continued to accelerate in Hillsborough Bay, the BSG decided to append its program in 1997 by adding transect monitoring. The transect monitoring design incorporated many features used by Virnstein in the Indian River Lagoon and also adopted recommendations made by the Tampa Bay National Estuary Program's (now the Tampa Bay Estuary Program or TBEP) Technical Advisory Committee (TAC) in 1994. The TAC recommended that transect monitoring primary objectives be able to:

1. Determine areal extent of seagrass in each bay segment.
2. Document zonation with depth.
3. Follow changes in seagrass zonation, patterns of zonation, and seagrass distribution over time.

The secondary objective of a transect monitoring program would be to determine the health of seagrass.

In order to meet the primary and secondary objectives, the TAC recommended four levels of monitoring. These monitoring levels were:

1. Map and groundtruth all submerged aquatic vegetation (SAV).
2. Description of SAV coverage, seagrass short shoot density, and collection of hydrographic data.
3. Measurements of photosynthetic active radiation (PAR), seagrass leaf production.
4. Collection of SAV for biomass determinations for seagrass, macroalgae, and epiphytes.
5. Implement each level at a greater frequency.

DEVELOPMENT OF REGIONAL MONITORING

In 1997, the TBEP requested the BSG to present its transect monitoring concept to a consortium of Tampa Bay area agencies including:

- City of Tampa
- Florida Marine Research Institute
- Hillsborough County Environmental Protection Commission
- Hillsborough County Cockroach Bay Aquatic Preserve
- Manatee County
- Southwest Florida Water Management District

The consortium endorsed a program that met the TAC's primary objectives by including monitoring levels 1, 2, and PAR measurements from level 3. Each agency was assigned responsibility for transect monitoring in their respective subsection of Tampa Bay. It was concluded that monitoring would commence each October to coincide with SWIM's time frame for photographic overflights of Tampa Bay. Data collection began in October 1998.

During the development of the seagrass transect monitoring program, the collection of comparable data between the agencies became an issue as there were varying degrees of expertise within the consortium. To address this concern, a class is scheduled several weeks prior to October to train personnel on field protocols. The class requires generating and recording a trial data set by each agency at several seagrass locations. The data set includes assessments of seagrass attributes including seagrass species composition, short shoot density, and canopy height. Subjective ratings of sediment type, epiphyte types and loading, and general seagrass health are also addressed. Finally, after transect monitoring is completed, the agencies reconvene to resolve difficulties encountered during sampling.

TRANSECT MONITORING ISSUES

The TBEP has adopted a nitrogen management plan as a tool to control eutrophication and improve water quality in Tampa Bay. Seagrass has been selected as the biological indicator to gauge the effectiveness of this strategy. The TBEP has adopted a seagrass restoration goal similar to the acreage found in 1950. In order to reach this goal, the water column needs sufficient clarity to allow seagrass to grow to target depths established for each major bay subsection.

Generating accurate depth measurements related to a common datum is imperative when developing a relationship between the "deep edge" of seagrass growth and light availability. Johansson (these proceedings) has developed a GPS-based method that may measure points along a bar contour with an accuracy of at least 10 cm regardless of the tidal stage. Fixed elevation reference points along each transect can be quickly established in order to develop accurate depth measurements along bar profile, edge of seagrass bed, and the transect terminus.

Monitoring protocols for the seagrass transects dictate that water quality samples, hydrographic data, PAR data, and assessments of seagrass attributes are collected at the middle and seaward edge of a meadow and the 2-meter water depth contour. Transects traverse seagrass coverage that may be comprised of discrete patches or one to several discrete meadows. Also, seagrass meadows generally have areas of low-density coverage corresponding with an increasing depth gradient. These various coverage patterns make interpretation of mid or edge bed difficult.

Standards have been proposed to aid in defining seagrass coverage. For example, Virnstein has suggested that coverage comprised of 3% to 10% delineate the edge of a meadow in the Indian River lagoon while the BSG has adopted a standard of less than 25% in Tampa Bay. These definitions fail to include a short shoot threshold of sufficient density to produce a signature visible from a boat or on an areal photograph. Therefore, the areal extent of the actual seagrass coverage may be underestimated. These errors may become important, as seagrass zonation along transects will be compared to photographic interpretation of seagrass areal coverage.

After reviewing transect monitoring results from the initial year (1998), data from about 20% of the transects were not included in the database. Several transect locations were deemed unsuitable due to sediment composition (oyster bars or mud). Further, uncertainty of several transect starting points and vectors made the reiteration of data collection along the same line in subsequent years unlikely. Therefore the baseline data set for these transects start in 1999 or even 2000.

Reference sites along transects may be more easily located in the 2000 monitoring as the United States government recently eliminated selective availability of satellite transmissions critical for precise GPS measurements. This action has allowed the local agencies to utilize inexpensive undifferentiated GPS units to find reference PVC poles used to delineate each transect. Therefore, repetition of data collections at predetermined points along each transect will be ensured in succeeding monitoring efforts.

The ability to generate seagrass information at predetermined points along a fixed transect is essential when attempting to discern changes in seagrass species composition and zonation over a depth gradient. Annual trends in species zonation versus depth may be compared within a transect and among transects within a bay subsection. Also, depths along the bar contour may be examined to determine changes in bathymetry due to sediment accretion or erosion. Lewis (these proceedings) suggests that the loss of offshore transverse sandbars may be critical in influencing seagrass distribution. Data from each monitoring effort should provide "real time" information concerning the status of seagrass meadows in Tampa Bay.

NEXT STEPS

The value of water quality data, including PAR measurements, collected during transect monitoring has been a point of discussion within the consortium. For example, the PAR data is the only light attenuation information collected over seagrass beds on a regular interval. However, it only provides an annual, one-day, "snapshot" measurement of light attenuation which probably is not sufficient to develop an understanding of the light climate over the seagrass beds. Therefore, these inshore data must be compared with the water quality information collected at offshore stations, which are sampled on a monthly interval by the Environmental Protection Commission of Hillsborough County (EPC). Trends in this "deep water" water quality data collected by the EPC have been a critical component in seagrass management decisions. If the water quality data collected along the seagrass transects are significantly different from the offshore stations, then the collection of inshore data may need to be expanded.

The premise of these proceedings was generated through a TBEP-sponsored seagrass symposium entitled "Seagrass Management: It's Not Just Nutrients!". One of the recommendations from the symposium was to generate a better understanding of the biology of Tampa Bay seagrass. Support for this recommendation could be linked to monitoring levels three and four recommended by the TAC to include PAR measurements and determinations of biomass and seagrass leaf production at a greater frequency. The TBEP is presently working with the local agencies to develop a scope of study that may include these additional levels of monitoring.

ACKNOWLEDGEMENTS

The Tampa Bay Interagency Seagrass Program has been successful due to the efforts of many people. Holly Greening of the TBEP has been instrumental in coordinating and facilitating organizational meetings. Dr. Ray Kurz, Dr. Dave Tomasko, and Tom Reis provided a lot of insight during the development of the program. Lisa Baltus of Pinellas County, Kerry Harkinson of Manatee County, Nick Toth and Eric Lesnett of Hillsborough County, and Rebecca Conroy from FMRI has been the principal investigator for their respective agencies. I thank

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USING THE PRELIMINARY LIGHT REQUIREMENT OF SEAGRASS TO GAUGE RESTORATION SUCCESS IN THE INDIAN RIVER LAGOON, FLORIDA

Lori J. Morris, Robert W. Virnstein, Janice D. Miller

ABSTRACT

The minimum light requirement of seagrass in the Indian River Lagoon (IRL), Florida, was determined to be 25% of surface light. This value represents the average amount of light, %PAR (photosythetically active radiation), penetrating the water column to the canopy at the deep edge of the seagrass beds. The range of values is from 16% to 32%, with a median of 25%. This light requirement was compared to the amount of light reaching a target depth of 1.7 m. This target depth was established as a potential depth goal for seagrass restoration in the Lagoon. The amount of light, %PAR, reaching this target depth was calculated for each of 25 Lagoon segments. The median %PAR for these segments at 1.7 m is 15%. The difference between the actual percent of surface light available at the 1.7 m target depth (15%) and the light requirement (25%) indicates the level of water clarity improvement needed to meet the target. Only two segments in the IRL have at least 25% of surface light reaching a depth of 1.7 m. But some segments would need almost a 67% improvement in %PAR to sustain seagrass growth at the 1.7 m target depth.

INTRODUCTION

Many estuaries across the world have experienced large losses in seagrass coverage. The Indian River Lagoon (IRL) is no exception; it has lost an estimated 15%–20% seagrass coverage based on historical maps from 1943 (Virnstein and Morris 2000; Virnstein 1999). Other similar water bodies include Tampa Bay, which has lost between 46% and 72% of its seagrass from 1950s historical levels (Haddad 1989; Lewis et al. 1991). Sarasota Bay and Lemon Bay have lost 25% and 21%, respectively, from 1948 to 1974 (Tomasko et al. 1996). Laguna Madre, Texas, experienced a 50% loss from 1965 to 1988 (Onuf 1996).

These large declines in grassbeds have been associated with decreases in light availability (Dixon 2000; Tomasko et al. 1996; Kenworthy and Haunert 1991; Dennison et al. 1993; Onuf 1996). Light availability is the most important factor regulating the depth distribution, abundance, and productivity of submerged vegetation in shallow coastal embayments and estuaries (Dennison and Alberte 1982, 1985, 1986; Duarte 1991). Dennison et al. (1993) reported that maximum depth of seagrass survival could be determined by the percent of surface light reaching the bottom. Other studies supported the efforts to develop a direct correlation between the average percent surface light reaching the bottom and the maximum depth of seagrass growth (Dennison 1987; Duarte 1991; Kenworthy and Haunert 1991).

Developing this correlation between the average percent surface light reaching the bottom and the maximum depth of seagrass growth is a major part of the ongoing restoration efforts in the IRL. The goals of the IRL program at the St. Johns River Water Management District (SJRWMD) include efforts to maintain and enhance water quality necessary for seagrass health. In the IRL, the minimum light requirement for seagrass will be used to establish pollution load reduction goals (PLRGs). In order to establish scientifically defensible PLRGs, it is important to understand the processes that affect water quality and seagrass in the IRL.

The basis for a light requirement should include the assumption that light limits depth and the deep edge of seagrass is thus growing at the light requirement. There-

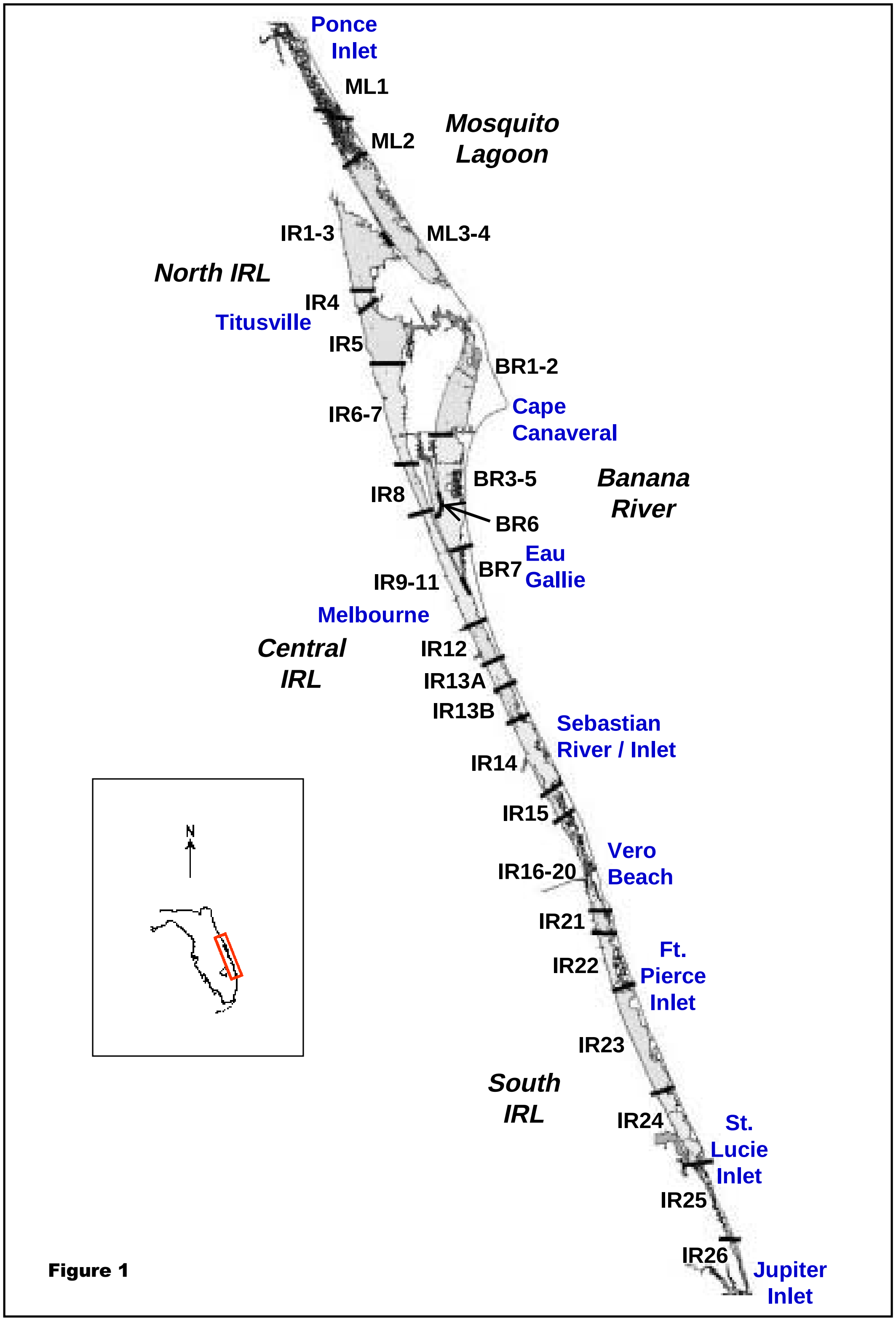

Figure 1. The Indian River Lagoon (IRL) study area. The Lagoon stretches 250 km along the east coast of Florida, from Ponce de Leon Inlet south to Jupiter Inlet. The IRL estuarine system is made up of three interconnected lagoons, Mosquito Lagoon (ML), Banana River (BR), and the Indian River Lagoon (IR); five inlets to the ocean; and more than ten tributaries and major canals contributing fresh water. For this study, the IRL has been divided into 25 segments, based on the condition of seagrass and water quality.

fore, the purpose of this report is to determine the minimum light requirement of seagrass in the IRL. The minimum light requirement is defined as the amount of light reaching the deep edge of the seagrass beds. The steps to determine the light requirement will be described in detail, including which data were used and the justification for using those data. A discussion of the problems encountered and suggested improvements for this approach are also included.

METHODS

For this study, the IRL was divided into 25 segments, based on the condition of seagrass and water quality (Fig. 1). A wide variety of methods has been used to estimate the light requirement of seagrass, but the most reliable estimates are generally determined by comparing *in situ* depth distributions of seagrass with long-term median value of diffuse attenuation coefficient, Kd(PAR) (Gallegos, 2001). Therefore, initial steps in determining the minimum light requirement for the IRL included defining the deep edge of the seagrass bed and determining the amount of light getting there. For a robust light requirement value we chose to use as many measurements as possible. Using five years (1991–1996) of light data, the percent light (%PAR) reaching the deep edge was calculated. The preliminary light requirement was then defined as the %PAR at the deep edge of seagrass in the IRL. This light requirement was compared to the actual light reaching the target depth of 1.7 m.

Study Site and Data Used

Four specific data sets were used in order to establish and compare the current and target %PAR at the deep edge of the seagrass beds. The data sets used were:

- Seagrass maps—The seagrass maps were produced from1:24,000 aerial photography, which were photo-interpreted, groundtruthed, and digitized (Virnstein and Morris 1996). The three specific years chosen, 1992, 1994, and 1996, represent complete, Lagoon-wide coverage (Virnstein et al. 2002).

- Bathymetry—Bathymetry data is a Lagoon-wide point coverage taken at NAVD88 (North Atlantic Vertical Datum ’88). Depths were taken every 50 ft across the IRL along transects 500 ft apart and contoured using ARC/TIN (Virnstein et al. 2002).

- Light data—The light data have been collected monthly, since 1991. The PAR measurements were used to calculate attenuation coefficients (Kd) for the water column. The Kd value is calculated as the slope of a semi-log regression of PAR with depth, using the method of least squares. The protocol included taking 3 replicates of PAR simultaneously at 20 cm and 50 cm below the surface and at canopy height (30 cm up from the bottom) using 3 LI-COR spherical sensors (4) and recorded by a LI-1400 data logger. The Kd was calculated using all nine numbers.

  A study was done to test for backscatter and bottom reflectance using 4 sensors. No significant difference was found: 1) between measurements inside and outside the grass beds, 2) with dark colored and light colored bottom surfaces, and 3) with a bottom plate under the sensor and not.

  An important question arose about using attenuation coefficient (Kd) values outside a grass bed to define the light requirement of seagrass. A different %PAR can be calculated with one measured Kd value for any depth. Therefore, the question of the appropriateness of using the Kd values measured near the Intracoastal Waterway (ICW), at the water quality monitoring network (WQMN) sites, to develop light requirement for seagrass needed to be answered.

A study was conducted in 1996 to test for differences in Kd values in the middle of the grassbed (MID), at the end of the grassbed (END), and at the WQMN station. Twenty sites were chosen where the WQMN stations were within 1 km of a seagrass transect. Water quality samples and light measurements were taken in the middle of the grassbed (MID), the end of the grassbed (END), and at the water quality station (WQMN). The study was run twice in 1996, in May (spring) and October (fall).

When the data from the study were pooled for both sampling periods (Spring and Fall), there was a difference in the Kd values between the MID and the WQMN ($P<0.007$, ANOVA), but not between the MID and the END, or the END and the WQMN.

Because the deep edge of the seagrass beds (END) were the concern for developing seagrass light requirement, and the results showed no difference between Kd values for the END and the WQMN, we decided that using the 10-year light data from the WQMN was valid to develop the seagrass light targets.

- Fixed Seagrass Transects—The seagrass transects are fixed line transects running perpendicular to shore out to the deep edge of the grassbed (Virnstein and Morris 1996; Morris et al. 2000). The 81 transects, Lagoon-wide, are monitored twice a year, summer and winter, since 1994 for visual estimates of species density, canopy height, and water depth.

Determining the Deep Edge

In order to calculate the %PAR at the deep edge of the seagrass beds, the deep edge must first be determined. The mapped seagrass coverages from the years ’92, ’94, and ’96 were chosen because they were complete, Lagoon-wide coverages, with corresponding water quality data. The aerial photos from those years were delineated and photo-interpreted then digitized into polygons to create maps. The mapped seagrass edge was projected onto an interpolated bathymetry coverage using ARC/TIN to provide a number of points for an average depth per IRL segment. The average deep edge of the seagrass per segment for these three years will be referred to as the mapped depth (see Virnstein et al. 2002, for complete methods).

Setting a Target Depth

A target depth is considered to be the depth distribution limit of seagrass under pristine conditions. For the IRL, the potential depth goal, or target depth of 1.7 m. was set. This target depth was established by averaging the depths at the deep edge of the “best” ten seagrass transect sites, (2 in Mosquito Lagoon, 3 in north Banana River, 2 in north IRL, 1 at Sebastian, 1 at Ft. Pierce, and 1 in Hobe Sound) from 1994–1998. These sites are located in the most “pristine,” unimpacted areas of the Lagoon; therefore, they are considered as the restoration goals for the rest of the IRL. Even though these depths are not referenced to a vertical datum, they have been sampled in different seasons for over seven years. The average depth of these 10 sites was 1.7 m.

Percent Surface Irradiance

The percent of surface light (%PAR) reaching the bottom of the water column is calculated from the following equation:

$$\%PAR = e^{-Kd*z} \bullet 100\%$$

where Kd is the attenuation coefficient value and z is the depth in meters. To calculate the %PAR at the deep edge of the seagrass bed, z = the average mapped depth per segment. To calculate the %PAR at the target depth goal, z = 1.7 m.

Light Requirement for Seagrass

The light requirement for IRL seagrass was determined using the same equation to calculate %PAR. By taking the Kd values for the range of dates preceding the dates of photography used in mapped years (e.g. 3/91 through 8/96 for the ’92, ’94 and ’96 coverage) and the average mapped depths per segment, the %PAR was calculated for each Kd value. The median %PAR reaching the mapped deep edge from all the segments was used to represent the preliminary light requirement for seagrass in the IRL.

RESULTS

Determining the Deep Edge

Determining the deep edge and determining the light requirement for IRL seagrass were dependent on the results from the study to test if Kd was constant across a grassbed. It was crucial to know if the near channel data set from the WQMN was usable for calculating percent surface irradiance reaching the seagrass. Since the deep edge of the seagrass beds (END) was the focus for determining seagrass light requirement, and the results showed no difference between Kd values for the END and the WQMN (Fig. 2), we decided to use the robust light data from the WQMN to develop the seagrass light targets.

The mapped depths, projected from bathymetry contours (NAVD88) to the seagrass coverage, was the largest data set to use (n = 2342). The technique used to generate the depths worked well in all segments except in the north part of Mosquito Lagoon (ML1 and ML2). These areas have numerous islands and are so shallow that the possible error in generating depths proved to be too great to be relied on. For complete details on the methods used for the mapped depth, see Virnstein et al. (2002).

Percent Surface Irradiance

The target depth of 1.7 m was used to calculate the %PAR at 1.7 m for each Kd value in the WQMN data set (n = 2299). This calculation estimates the amount of surface light reaching 1.7 m. The median %PAR was taken for all 25 segments with a grand median %PAR at 1.7 m of 15% (Fig. 3). The median %PAR at 1.7 m in Mosquito Lagoon and Banana River are less than 17%. The median %PAR in the northern IRL segments is greater than 22% at 1.7 m. From Cocoa Beach south through Melbourne, there is an increase in Kd, and therefore a decline in the median %PAR at 1.7 m to between 14 and 19%. The distribution shows a slight increase in median %PAR at 1.7 m near Sebastian Inlet to 20%, followed by a sharp decline through Vero Beach to 8.5%, increasing towards Ft. Pierce Inlet to 11%, declining near St. Lucie Inlet to less than 9%, and finishing with a large spike at Jupiter Inlet with 31% (Fig. 3).

Light Requirement for Seagrass

The %PAR from the 23 segments (25 minus the 2 segments in Mosquito Lagoon, where bathymetry data is lacking) was calculated using the mapped depth (n = 2299). The median %PAR ranged from 16% to 32%, with a grand median of 25% to represent a “preliminary light requirement for seagrass” for the IRL (Fig. 4). Almost half the segments have a median %PAR above 25%, and most of those are from North IRL south to Central IRL (Fig. 4).

To show the difference between the %PAR at the deep edge of the seagrass beds (25%) and the %PAR at 1.7 m, a line was added to Figure 3 at 25% to represent the preliminary light requirement. Only 2 of the 25 segments have adequate light available at 1.7-m segments IR5 in the northern IRL and IR26 at Jupiter Inlet.

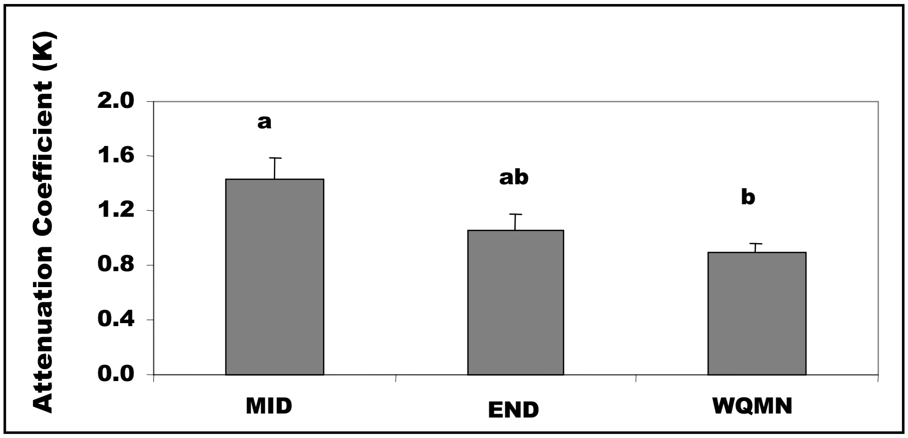

Figure 2. The results from the study to test if Kd was constant across a grassbed. There was a significant difference in the attenuation coefficient (Kd) values between the MID and the WQMN (water quality monitoring network) ($P<0.007$, ANOVA), but not between the MID and END or the END and the WQMN.

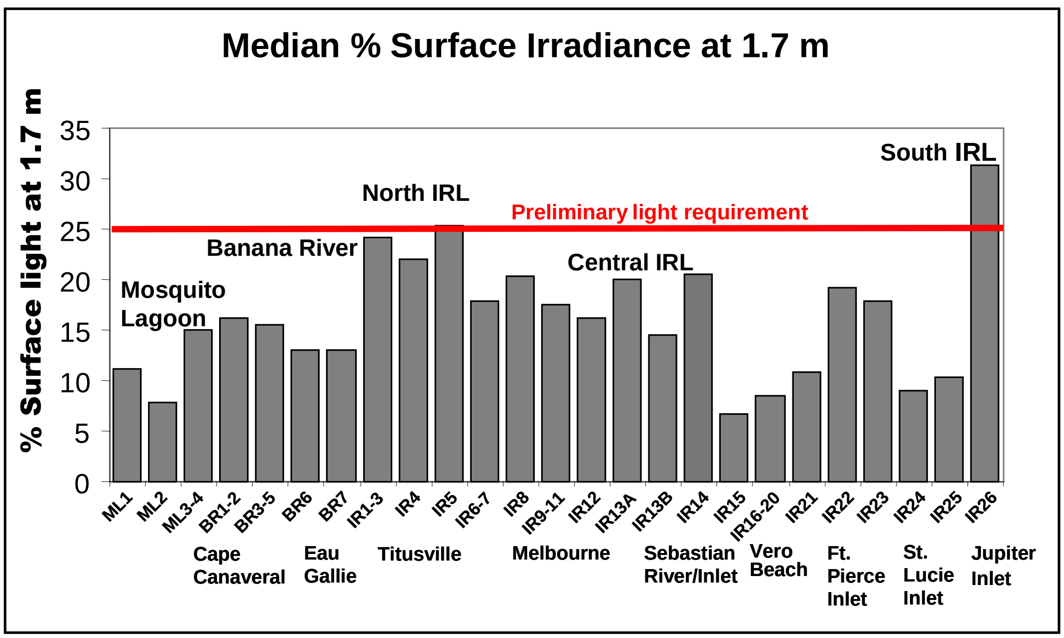

Figure 3. The median percent surface light reaching 1.7 m for each of the 25 segments, with a grand median of 15%. The line at 25% represents the preliminary light requirement for the Indian River Lagoon. Only 2 of the 25 segments have adequate light available at 1.7 m to support seagrass—segments IR5, and IR26.

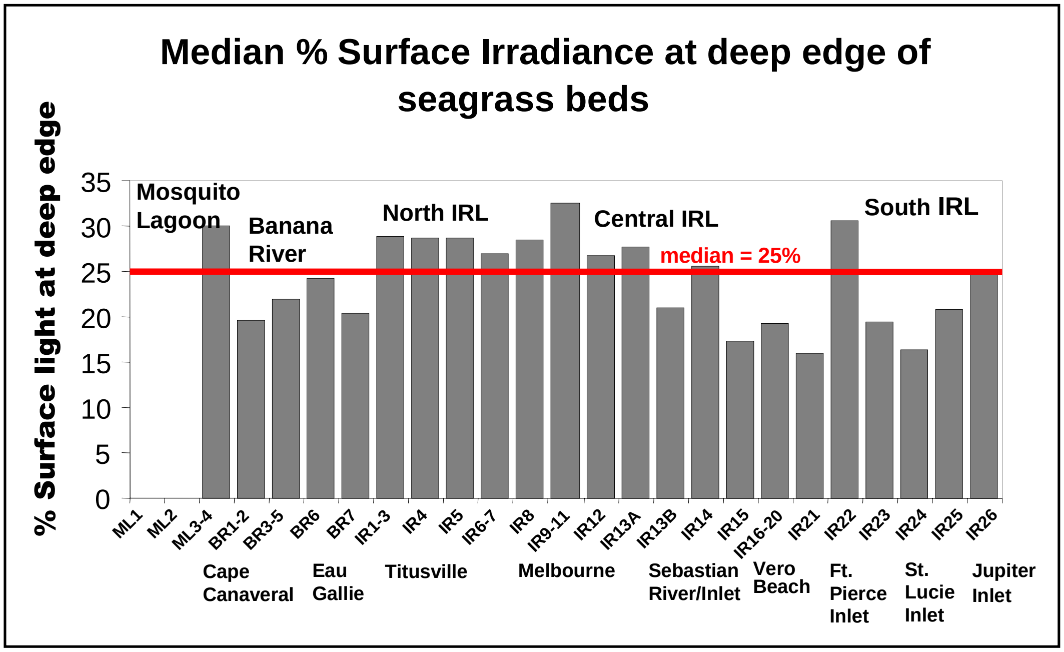

Figure 4. The median percent surface light at the deep edge of the seagrass beds for each of the 23 segments. Lagoonwide, a grand median %PAR of 25% reached the deep edge of the seagrass beds, representing a preliminary light requirement for the Indian River Lagoon.

DISCUSSION

Determining the Deep Edge

Determining the deep edge of the seagrass beds was not as straightforward as originally thought. There are a number of "soft" definitions of "the edge," but not a universally accepted one (Virnstein et al. 2002). We chose to use the mapped edge because they were based on an average of over 50 points per segment versus 3 points per segment from the fixed seagrass transects. However, the mapped depths were not with their problems too. The ARC/TIN software interpolated depths below the bathymetric data limits of 0.3 m, which had to be filtered from the data and discarded (Virnstein et al. 2002).

Percent Surface Irradiance

Before calculating a percent surface irradiance from Kd values, a number of things were investigated:

- Was there any significant differences in the Kd values across the grassbeds? Since we found no differences between Kd values at the WQMN and those at the deep edge of the seagrass beds (END), we were able to use the 10-year, WQMN data set.

- If using spherical, 4 irradiance sensors, bottom-type may be important. Dixon (2000) found a significant increase in PAR measurements taken in bare sand areas as compared to dense seagrass coverage. These higher PAR measurements could produce "false," low Kd values, and thus higher percent surface irradiance numbers. However, we found no significant difference when comparing sensors: 1) between measurements inside and outside the grass beds, 2) with dark colored and light colored bottom surfaces, and 3) with having a bottom plate under the sensor and not.

- To avoid problems with geometric means, it is important to calculate the %PAR for all values using Kd *before* taking the median or mean.

Light Requirement for Seagrasses

Many studies suggest a broad range of minimum light requirement for seagrasses; from as little as 4.4% to as much as 38.3% surface light (Kenworthy and Fonseca 1996; Dennison et al. 1993; Duarte 1991; Onuf 1991; Dixon 2000). Kenworthy and Fonseca (1996) suggest that this variability may be species specific or due to infrequent sampling, thus failing to capture seasonal variations in light attenuation. The light data used for this study is a sub-sample of almost a decade of monthly monitoring in the IRL. The 5-year period used captured many growing seasons as well as storm events and large releases of freshwater into the estuary. Therefore, we believe that the values used to calculate the %PAR are defensible, and the preliminary light requirement of 25% for the Lagoon falls well within the range of these reported values. It is noteworthy to mention that if we only chose one or two segments in the Lagoon to calculate the light requirement, we could have significantly overestimated (32.4%) or underestimated (15.8%) the %PAR at the edge.

Comparison Between the Preliminary Light Requirement and %PAR at 1.7 m

The difference between the actual percent of surface light available at the 1.7 m target depth (15%) and the light requirement (25%) indicates the level of water clarity improvement needed to meet the target. Some segments would need almost a 67% improvement in %PAR to sustain seagrass growth at the 1.7-m target depth.

However, it may not only be light that needs improving. Koch (2001) believes that it is time to look at other factors besides light when determining seagrass habitat requirements. There are many segments with greater than 25% PAR at the deep edge of the seagrass beds, but the edge is much less than 1.7 m. For example, segment IR9-11, in the Melbourne area, has 32% light at the edge, but the edge is only at 1-m deep. The same scenario also exists in Mosquito Lagoon (ML3-4), with 30% light at 1-m, but possibly caused by different reasons than the Melbourne area.

These numbers are preliminary and there are a few other factors we need to consider before using these preliminary numbers to set final restoration targets:

- We did not separate any seasonality or growing season differences (e.g. spring/ summer vs. fall/winter) in light intensity, %PAR, light requirement, and species differences. These seasonality differences could prove to be important for *Halodule wrightii*, the dominant species of seagrass or more particularly for *Halophila decipiens*, the only annual species found in the IRL (Morris et al. 2000). Future investigations need to address other available data sets. For example, fixed seagrass transects (Morris et al. 2000) are monitored summer and winter, since 1994. Water depths at the edge of the beds are always measured but have not been referenced to a vertical datum. Another data set consists of six sites in the Lagoon monitored continuously for light, water quality and seagrass for 2 years (Hanisak 2001).

- Future investigations need to address light attenuation due to epiphytes, which could be as much as 50% (Harden 1994; Dixon 2000). Average epiphyte abundance in the IRL is high (Harden 1994), but epiphytes appear less abundant at the deep edge of the bed (personal observation). Most current papers do not address where in the seagrass bed measurements for increased light attenuation due to epiphytes are made.

- The Chesapeake Bay Program uses two ways to evaluate percent light—

percent light through the water column (PLW) and percent light at the leaf surface (PLL) (Bergstrom 2000). The IRL value of 25% is the percent of light reaching the canopy, or PLW, and doesn't incorporate attenuation due to epiphytes.

- Other factors besides light, or water clarity parameters, may be controlling seagrass growth at the deep edge in a segment. Thresholds for physical, geological, and geochemical parameters may also be important (Koch 2001), but insufficient information is currently available.

- The high variation of %PAR at the edge of the bed (16%–32%) was surprising. In some segments with higher %PAR, light is probably not the primary limiting factor. Perhaps these segments do no provide appropriate measures of *minimum* light requirement, and should not be used in calculating a median value. Rather, these areas with high %PARs at the edge of the beds are where we should look for important factors other than light (Koch 2001).

ACKNOWLEDGEMENTS

We gratefully acknowledge the many hours of work from Ed Carter, for GIS support, and Wendy Tweedale, for providing water quality data from the IRL database. We also thank the staff from the following state and federal agencies for their dedication in monitoring the WQMN stations and the seagrass transects: National Park Service at Canaveral National Seashore; Dynamac Corporation; Brevard County, Surface Water Improvement Department; Florida Department of Environmental Protection, Aquatic Preserve offices in Fellsmere and Port St. Lucie; U.S. Fish and Wildlife Service, South Florida Ecosystem Program; South Florida Water Management District; and Loxahatchee River District. This study was funded primarily by the St. Johns River Water Management District.

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UTILITY OF SEAGRASS RESTORATION INDICES BASED ON AREA, DEPTH, AND LIGHT

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Lori J. Morris, Janice D. Miller

ABSTRACT

The goal of this document is to develop and evaluate three seagrass indices—one based on area of seagrass, one based on maximum depth of seagrass, and a third based on light. The area-based index compares actual area of seagrass to the restoration target. The target is potential seagrass area shallow enough to support seagrass—that area of Indian River Lagoon (IRL) bottom shallower than 1.7 m. The depth-based index compares depth of the deep edge of seagrass to the target depth of 1.7 m. The 1.7-m target is based on twice-a-year monitoring of field transects at 10 "good" sites from 1994 to 1999. The third alternative is a light-based index: the actual amount of light reaching the seagrass depth target of 1.7 m relative to the preliminary light requirement of 25% of surface light.

Because seagrass depends primarily on light (water clarity), seagrass status, as measured by the two indices, was compared to light availability in each of 25 segments in IRL. The area-based index was moderately correlated with light availability ($r^2 = 0.32$; $p = 0.003$). The depth-based index correlated with light availability better ($r^2 = 0.52$; $p = 0.001$), with variation in annual average light availability explaining about half of the variability of seagrass status, as measured by the depth-based index.

Each index has its advantages and disadvantages. Area is inherently the focus of restoration efforts. But, because of differences in bathymetric shape of segments, area correlates poorly with light availability. Depth correlates better with light availability. But the depth-based index requires seagrass maps and bathymetric contours to derive several point measurements. However, several field measurements at the deep edge can also provide a rapid, real-time assessment of maximum depth. The light index can be converted directly to a light attenuation measure and correlated with various optical water quality parameters.

Other factors possibly contributing to the lack of a close relationship of seagrass to light include: wavelength shifts due to colored water, salinity outside the tolerance range of seagrass species, unstable sediments, wave energy, and many sources of measurement error, especially associated with determining the "edge" of the bed. Additionally, because of spatial auto-correlation of both seagrass and water quality, analyses are probably flawed due to lack of information on appropriate time intervals and lag times. For example, is it most appropriate to compare seagrass to the previous 6 months, 2 years, or growing season of light data? Seagrass indices, especially a depth-based seagrass index, are useful tools for characterizing segments of the IRL estuary, even though improvements in the indices are warranted.

INTRODUCTION

Seagrass has recently received much emphasis for its usefulness as a barometer of estuarine ecosystems (Dennison et al. 1993; Batiuk et al. 2000; Johansson and Greening 2000). Seagrass serves as an appropriate barometer because it provides a critical link between water quality and ecosystem biology. This conceptual model can be expressed as:

LOADING → WATER QUALITY → LIGHT → SEAGRASS → ANIMALS

That is, pollutant loadings (sediments and nutrients) affect water quality, which affects light attenuation, which affects seagrass, which affects ecosystem biological productivity and diversity. Light availability is the single most important factor limiting the maximum depth of seagrass (Kenworthy and Haunert 1991; Morris and Tomasko 1993; Onuf 1996; Virnstein and Morris 1996).

In the Indian River Lagoon (IRL), seagrass is considered the most critical habitat type and is the focus of restoration efforts (Steward et al. 1994; IRLNEP 1996; Steward 2002). But to what level do we restore seagrass, and how do we measure success? Large losses have occurred in the central IRL (Woodward-Clyde 1994; Fletcher and Fletcher 1995; Virnstein 1999). By circular reasoning, those segments that have lost the most seagrass have the highest potential for seagrass recovery. But results of restoration efforts need to be measured against a standard or target. By comparing past and present conditions against this target, we can then assess recovery.

Of the 7 species of seagrass present in the IRL, *Halodule wrightii* is the dominant species—about 60% of the seagrass coverage, based on transect monitoring (Morris et al. 2000). *H. wrightii* is especially dominant at the deep edge of beds. But mapping cannot and does not distinguish species. Thus all discussions below refer to seagrass in general and are not species specific.

The purpose of this document is to develop, compare, and evaluate two seagrass indices—one based on seagrass area and one based on seagrass depth. For this document, the process of developing these two indices is as important as the indices themselves. Each index is based on quite different sets of data and criteria. These indices are then used to compare seagrass status to water clarity status, measured as light availability, for 25 segments of the IRL. A few problems and stumbles are addressed, and recommendations are made for improving the indices.

METHODS AND RESULTS OF AREA-BASED TARGETS AND INDEX

Initially, **area of seagrass coverage** was established as the primary quantitative target, or restoration goal (Virnstein and Morris 2000). Two quantitative area-based targets were established. The "Historical" target is wherever seagrass has occurred in the past, derived primarily from 1943 coverage maps. The primary target is the "potential" target—that area of Lagoon bottom less than 1.7 m deep. This target depth of 1.7 m was the average depth of the deep edge of the bed based on twice-a-year (summer/winter) field monitoring of fixed transects in 10 "healthy" areas from 1994-99 (Virnstein and Morris 1996; Morris et al. 2000; Morris et al. 2002). Because these areas are near pristine, we assume they represent conditions that cannot be reasonably improved; that's as good as it's likely to get in the IRL.

For the entire IRL, a potential restoration target of 50,000 hectares (124,000 acres) has been preliminarily established. There are presently about 29,000 hectares (71,000 acres) in the IRL, based on 1999 seagrass maps. In Chesapeake Bay, the first level ("Tier I") of seagrass restoration targets is the area inhabited by seagrass at any time as mapped from 1971 through 1990 (Batiuk et al. 2000). In Tampa Bay, the target is also based on area—approximately the area inhabited by seagrass estimated from 1950 aerial photographs (Janicki et al. 1995; TBNEP 1996; Johansson and Greening 2000).

But area of seagrass is not sufficient by itself to assess the status or condition of seagrass. The area of seagrass needs to be referenced to a common standard. The standard we use is the potential target.

(Eq. 1) Area-based index =
(acres of seagrass ÷ potential acres)*100%.

We used the potential target as the standard rather than the historical target because it is the most consistent and dependable overall. For some areas, "historical" coverage was already highly impacted by 1943.

Because some sections of the IRL are healthier than others (Woodward-Clyde 1994; Virnstein 1999), we divided the Lagoon into segments in order to characterize seagrass and to compare seagrass status to water clarity within each segment. Based on differences in seagrass relative to targets and on differences in water quality, the Lagoon was divided into 25 segments (Fig. 1). Existing seagrass area for each segment was calculated as the average from Lagoon-wide seagrass maps for 1992, 1994, and 1996. This 3-year average area-based index varies from 8% (i.e., only 8% of the potential area is currently occupied by seagrass) for segment IR9-11 (the Cocoa/Melbourne area) to 104% for segment IR26 (near Jupiter Inlet) (Fig. 2A; see Fig. 1 for segment location). Despite large differences in absolute amount (acreage) of seagrass in segments, the index provides a basis for judging which areas are good, fair, and poor, etc., relative to the target (Virnstein et al., in press).

Because seagrass is primarily limited by light availability (Kenworthy and Haunert 1991; Kenworthy 1993; Morris and Tomasko 1993), we expect that the status of seagrass in a segment should be related to the status of water clarity, i.e., segments with good water clarity should have good seagrass. Therefore, the primary water quality indicator is the percent of surface light reaching the target depth of 1.7 m—called "%PAR at 1.7 m" (PAR = Photosynthetically Active Radiation, a measure of light available to plants) (Morris et al. 2002). The seagrass area and depth indices are mean values calculated from 1992, 1994, and 1996 seagrass maps —years with complete map coverage of the entire Lagoon. Water clarity data (%PAR at 1.7 m) are medians of monthly data from 1991 to 1996 from long-term water quality monitoring sites located in each segment (see Morris et al. 2002 for details of light data). We had originally chosen light data from each year previous to the mapped years as the most influential on seagrass, but further discussion and resulting higher regression coefficients led us to use the

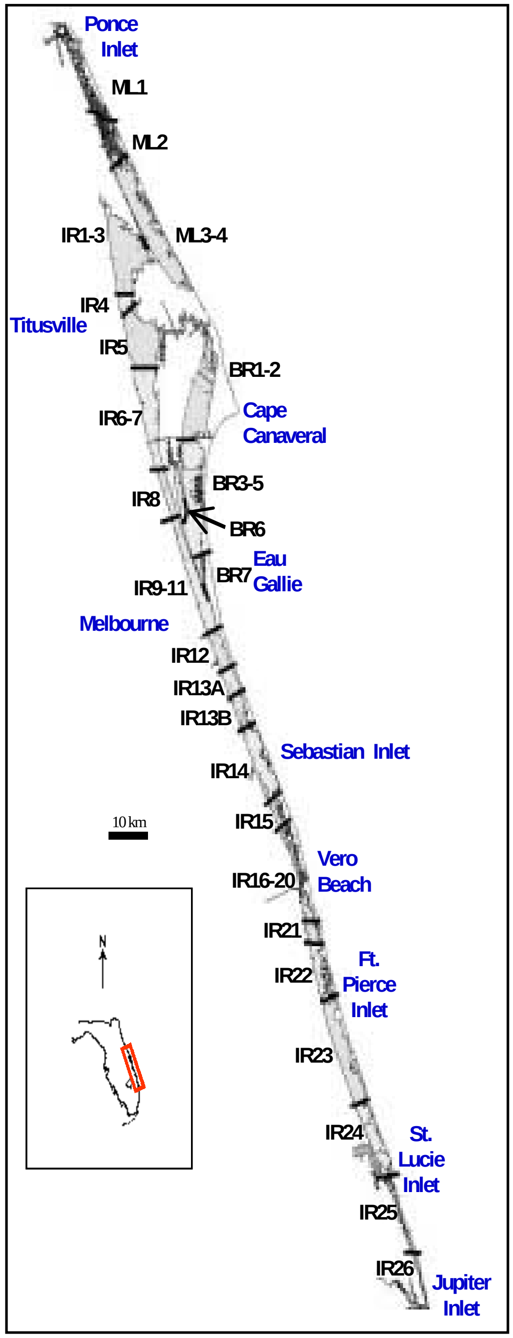

Figure 1. Map showing the 25 segments in the Indian River Lagoon used for analysis of seagrass indices.

median of all light data from March 1991 to September 1996 to best represent long-term "typical" light conditions associated with average seagrass condition over this same time period. When the area-based seagrass index is regressed on %PAR at 1.7 m, 32% of the variation in the area-based seagrass index is explained by the variation

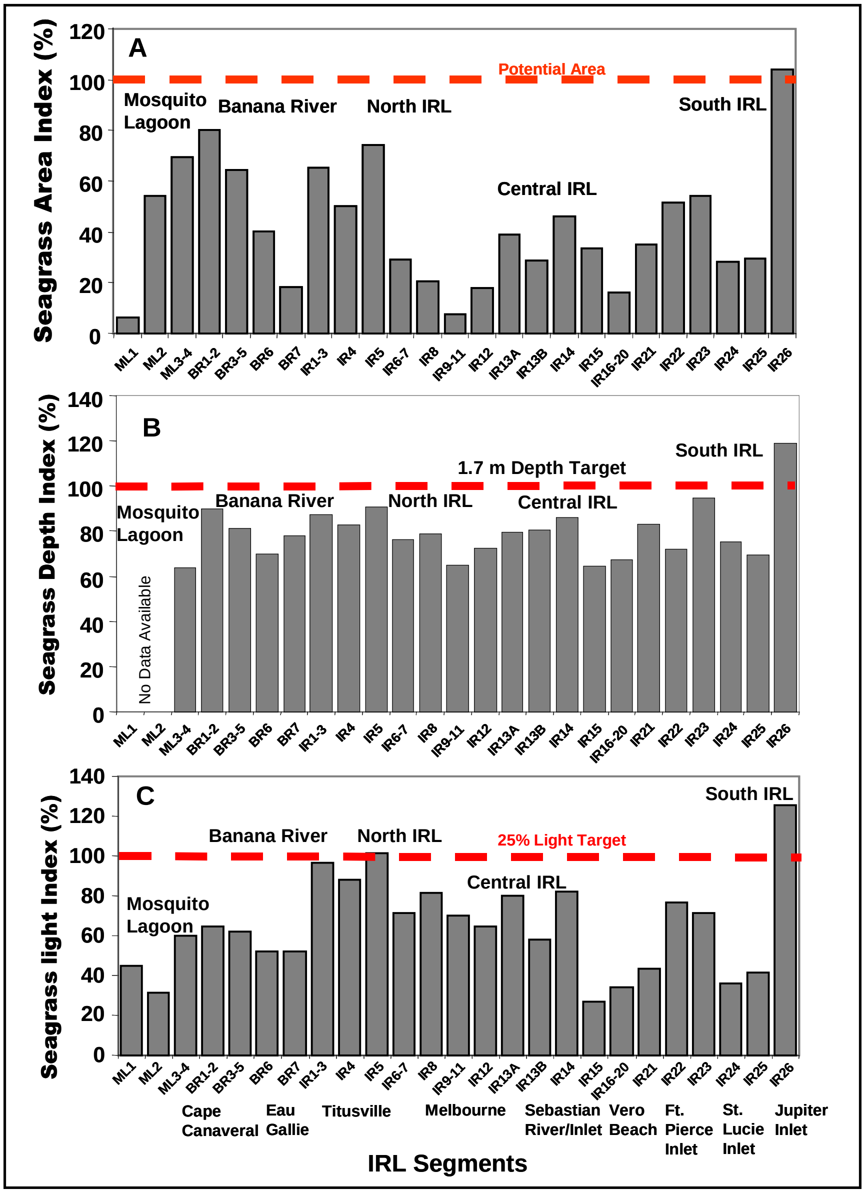

Figure 2. Bar graphs of the three indices relative to the target (dashed line) for each index. Segments for the three major lagoons are listed north to south. See Fig. 1 map for segment location in each of the three lagoons. **A.** Bar graph of the area-based index, by segment. **B.** Bar graph of the depth-based index, by segment. **C.** Bar graph of the light index, by segment. All plots use seagrass averages of 1992, 1994, and 1996.

in light availability ($r^2 = 0.32$; $p = 0.003$) (Fig. 3A).

One problem with this area-based index is that potential targets and acres of seagrass are dependent on the shape of the bottom contours of a segment. That is, a segment with a lot of very shallow bottom that supports seagrass may have a high area-based index despite poor water quality. Conversely, segments where depths drop off rapidly to 1.7 m may have a poor area-based index despite good water quality. To illustrate this paradox, a depth of 0.9 m includes 60% of potential area in shallow segment ML3-4, but only 20% in segment IR8, which has little shallow water (Fig. 4). Although an increase in the average depth of the deep edge of seagrass would necessarily result in some increase in the

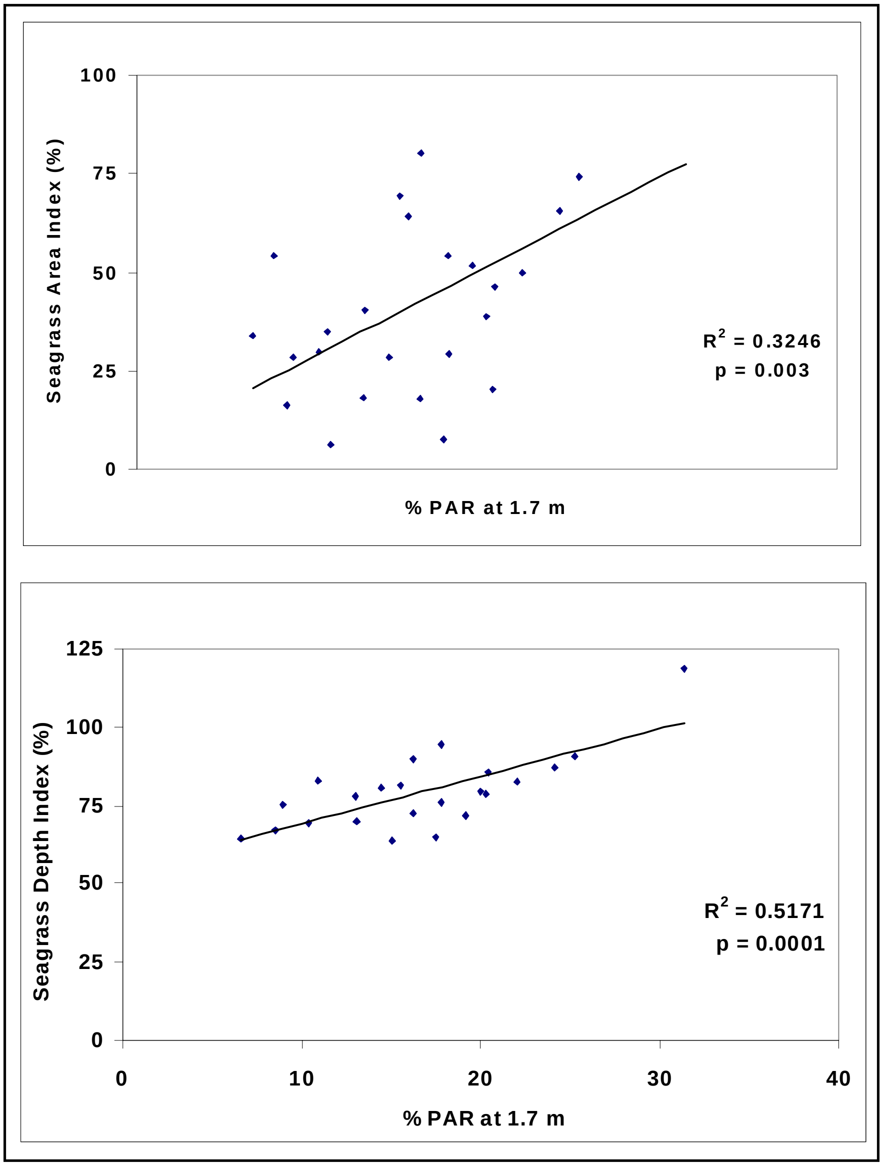

Figure 3. Scatter plots of seagrass indices for each segment versus %PAR at the 1.7-m target depth. The seagrass indices (from Fig. 2) are based on the averages of maps of 1992, 1994, and 1996 seagrass. %PAR at 1.7 m is the median of monthly values from 1991 to 1996. **A**. Seagrass area index versus %PAR at 1.7 m. **B**. Seagrass depth index versus %PAR at 1.7 m. Regression coefficients and level of statistical significance are given.

area of seagrass, the relationship is definitely non-linear and variable (see example in Fig. 4). Thus, an area-based index does not necessarily correlate well with light availability.

METHODS AND RESULTS OF DEPTH-BASED TARGETS AND INDEX

To avoid the inherent problems of an area-based index, we decided to develop a seagrass index based on maximum depth of seagrass growth. Such a depth-based index (e.g., percent of target depth of 1.7 m) should not vary with bathymetric shape and may thus be a better indicator of seagrass status. Also, in Chesapeake Bay, the second and third of the three restoration tiers are potential area based on depth—the 1- and 2-m depth contours (Batiuk et al. 2000). For Tampa Bay, Lewis et al. (1985) estimated the potential area based on

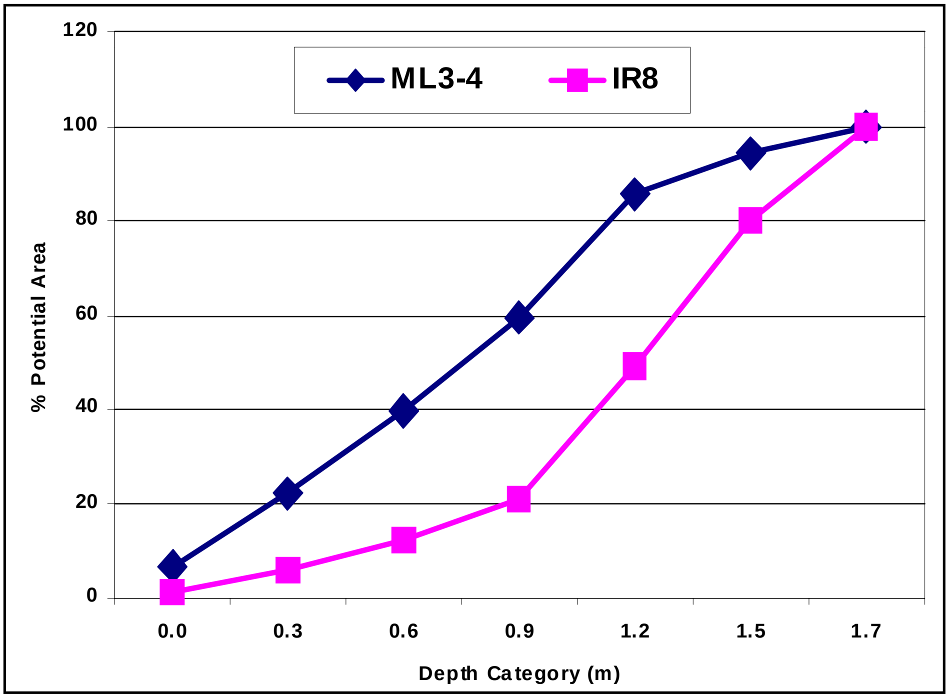

Figure 4. Example of depth-versus-area curves for two segments, based on bathymetric contours (see Fig. 1 for segment location) at 30-cm depth intervals to the target depth of 1.7 m. Note that segment ML3-4 is broad and shallow, with 60% of its potential seagrass area shallower than 0.9 m, whereas only 20% of segment IR8 is shallower than 0.9 m.

estimated depth contours in 1879 at about 31,000 ha. Instead, the historical (1950s) target of 15,400 ha was adopted (TBNEP 1996).

Developing a depth-based seagrass index for the IRL requires several steps. To calculate a depth-based index, we needed to measure the average depth of the deep edge of the seagrass beds in each segment. One source is the fixed transects (Virnstein and Morris 1996; Morris et al. 2000; Morris et al. 2002). But our field measurements of depth are not referenced to a vertical datum (but could be by GPS positioning referenced to a good bathymetric surface), and some Lagoon segments contained as few as two transects, a small sample size for characterizing a segment. Future refinements in analyses (e.g., bathymetric contouring and georeferencing) may allow greater use of these transect data so that field-measured depths may be referenced to a vertical datum.

To generate more replicate measurements for a more robust average depth of the deep edge, we resorted to a map-based alternative. This procedure was done in a GIS using ArcView software. In this procedure, we overlaid 20 equally spaced east-west lines over the map of seagrass polygons for each segment. The deep edge corresponds to about 10% coverage, based on visual estimates. Along each of the 20 lines, a point was placed everywhere the line crossed the mapped deep edge of a seagrass bed (Fig. 5). Then (using the ArcView command TINSPOT) the depths of each point were “read” off the underlying bathymetric surface (a TIN coverage). An average of about 50 points was created for each segment for each mapped year and then averaged. These average depths for the three mapped years were averaged to calculate a mean depth of the edge of bed for each segment. The mapped seagrass years were 1992, 1994, and 1996, those years that covered the entire IRL.

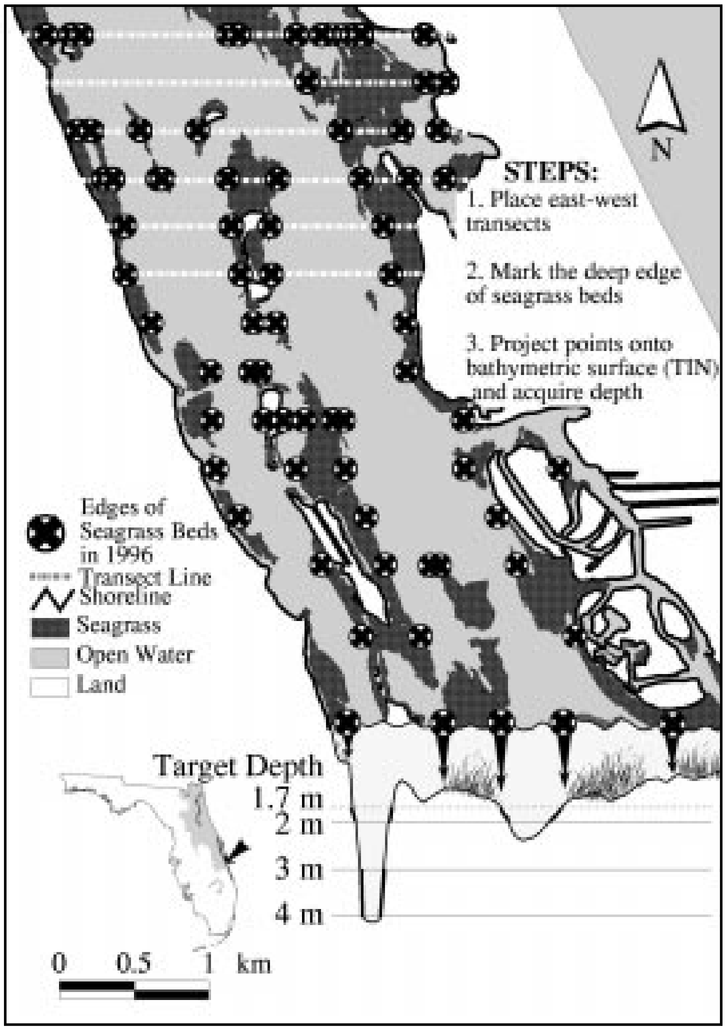

Figure 5. Diagram of the 20-transect method for determining the average depth of the "edge of bed" of seagrass for a Lagoon segment. In this example, the transects intersect the deep edge of seagrass beds at 74 points. The upper portion of the diagram shows 6 of the 20 transects, and points are marked with "⊗" where the transect line intersects the deep mapped edge of a seagrass bed. The lower part of the diagram illustrates a cross section and how the depth for each point is projected onto the bathymetric surface. The Intracoastal Waterway is near the western shoreline. The final step is to average the depths of all points for each segment. This example is from the Grant area (south Brevard County, segment 13B in Fig. 1).

To convert this depth of edge of bed to an index, we simply converted it to a percent of the target depth of 1.7 m. Thus, the

(Eq. 2) Depth-based seagrass index = (average depth of edge of bed ÷ 1.7 m)*100%.

This depth-based seagrass index is considered statistically robust because (1) it's map based, and it measures the same "edge" by which we map and measure restoration success, (2) 2342 points are measured, creating a robust average, (the number and orientation of these "transect" lines could be adjusted to local conditions), and (3) depths are standardized by being referenced to a vertical datum, in this case, NAVD88. In subsequent years, these same lines could be used to detect small changes in maximum depths of seagrass, acting as "virtual" fixed transects. Through use of GPS, these virtual transects could be located to correspond exactly to actual field transects for field verification.

As with the area-based index, this depth-based index can be used to characterize and compare segments of the Lagoon. Using the same segments (Fig. 1), the depth-based index varies from 63% to 119% (Fig. 2B). Segments around Kennedy Space Center, protected by NASA ownership, and segments near inlets have high index values (generally above 80%). Conversely, segments with extensive land development and poor flushing (Cocoa/Melbourne and Vero Beach areas) have low seagrass index values (below 67%). Southern Mosquito Lagoon (Segment ML3-4), a protected pristine area with extensive seagrass, is an outlier, with a low index of 63%, possibly due to its shallow nature and high wind-driven resuspension (Fig. 2B).

Again relating seagrass status to light availability, we expect segments with a good depth-based seagrass index to have good water clarity. When the depth-based seagrass index is regressed on light availability (%PAR at 1.7 m), there is a significant relationship ($r^2 = 0.52$; $p = 0.0001$) (Fig. 3B). Variation in light availability explains half of the variation in the seagrass depth index, an improvement over the 32% using the area-based index (above, Fig. 3A).

AN ALTERNATIVE LIGHT-BASED TARGET AND INDEX

Although seagrass is the most direct measure of status of a Lagoon segment, we could also use as a target the water clarity necessary for seagrass to grow to the target depth of 1.7 m. Morris et al. (2002) established that an annual median of 25% of sub-surface light is the minimum required for seagrass growth. Based on 5½ years of monthly data, Morris et al. (2002) calculated the median amount of light reaching the target depth of 1.7 m for each of the 25 segments. The status of a segment could thus be judged by comparing the actual light to the light target of 25% at 1.7 m. This ratio would thus be the

(Eq. 3) Light-based seagrass index = %PAR at 1.7 m ÷ 25%*100%.

Only two segments meet this light-based target (Fig. 2C). Again these are segments in the north Indian River and extreme southern Indian River near Jupiter Inlet. Mosquito Lagoon, because of its shallowness and presumed high rate of resuspension reaches only 32-63% of the light target (Fig. 2C).

This light-based target of 25% PAR at 1.7 m requires a light attenuation coefficient (Kd) of not more than 0.8/m. This target light attenuation coefficient can then be correlated with various optical water quality factors, e.g., turbidity, color, suspended solids, and phytoplankton chlorophyll. But some of these coefficients may need to be determined empirically for each segment (Gallegos 2001). Targets for these water quality factors can be related to loading from point sources and stormwater. Thus, targets provide an effective tool to evaluate management strategies (Steward 2001).

DISCUSSION: COMPARISON OF AREA- AND DEPTH-BASED INDICES

Each index has advantages and disadvantages. Amount of seagrass, measured as area, is inherently the focus of restoration efforts. However, the area index may not accurately reflect the degree of water quality improvement needed. For example, if extensive areas of shallow water are already occupied by seagrass, a large increase in water clarity might possibly result in only a small increase in area of seagrass. Conversely, segments with large areas of available shallow bottom might experience a large increase in seagrass as a result of a small increase in water clarity.

Depth of seagrass is inherently valuable because it relates directly to water clarity. But the depth-based index is point-based and may not as accurately measure how close we are to the seagrass area target (it depends on and requires bathymetry). Although map data are robust, calculating depth data from seagrass maps requires several steps. The multi-layer complexity of these steps (aerial photos, georeferencing, photo-interpretation, digitizing, locating points along bed edges, and projected onto a bathymetric surface to generate depths) results in substantial time lags, often >1 year. Mapping protocol using digital hyperspectral imagery may substantially improve this turnaround time. There are alternative measures of depth that offer advantages of simplicity and rapidity of measurement. For example, besides extracting depths from seagrass maps and bathymetry contours, depth can easily and very rapidly be assessed by quick field measurements or point checks. These measurements would need to be referenced to a vertical datum. These field checks do not require the good water clarity required for aerial photos; good water clarity is a rare event in segments of some estuaries, including many segments of the Indian River Lagoon. The ratio of predicted area gain to increased depth of the seagrass bed might provide a cost-benefit guide for prioritization of restoration efforts.

Although variation in water clarity (light availability) explains about half the variation in depth of seagrass, half the variability is still unexplained. Possible explanations of the other half of variability may include other optical water properties and non-optical properties (physical/chemical/biological factors) (Koch 2001). Some examples: In the IRL, potentially compounding biological factors include competition with macroalgae and grazing by manatees. Wavelength shifts in highly colored waters (Gallegos 1993; Gallegos and Kenworthy 1996); PAR may then not accurately measure the amount of utilizable light. Salinity may be too low (Doering and Chamberlain 2000) for seagrass survival, despite high light levels. Wind-driven waves and boat wakes may make sediments too physically unstable for seagrass (Fonseca and Robbins 2001; Lewis 2002). This wave energy would interact with sediment properties such as grain size and cohesiveness. Sediments may be nutrient poor (Short 1987). All conditions may be appropriate, but no seagrass occurs because of recruitment limitations of the various seagrass species (Inglis 2000; Kenworthy 2000; Short et al. 2001). Also, none of the measurements are made without error. Sources of error calculating the depth-based seagrass index include inaccuracies of position, locating the “edge” of the bed, and bathymetric contouring (Johansson 2002; Kurz 2002; Virnstein et al. 2002). And to characterize water clarity adequately, how frequently light attenuation needs to be measured is a nagging question. Hanisak (2001) found that light measurements every 2 weeks give a good representation of conditions. For analysis, however, there is little information on the appropriate time intervals and lag times for comparing snapshots of seagrass status to water clarity status. For example, is it most appropriate to use the previous 6 months or 2 years or growing season of water clarity data, and should the older sampling be weighted less than more recent sampling? Or should absolute light intensity be used in analyses rather than percent of surface light? Only more intensive sampling (e.g., Hanisak 2001) and time series analysis can answer these questions.

Although we normally tend to use statistics based on central tendencies (i.e., average values), seagrass presence in an area may also be limited by extreme conditions caused by rare or short-term events (Gaines and Denny 1993; Gallegos 2001). For example, if light and all other factors are suitable, but salinity is too low, seagrass will not be present. Some of the limiting rare events may include major storms, rainfall, controlled water releases, and natural cycles. Besides extreme events, the degree of variability, e.g., in salinity, can also impact seagrass (Montague and Ley 1993).

As an example of the latter natural cycles, the amount of seagrass biomass (plus detritus and drift macroalgae), as measured along a transect, reached very high levels in the poorly flushed northern Indian River (north end of segment IR1-3, Fig. 1) prior to 1995. By summer 1997, virtually all of the seagrass (about 120 ha) in the northern portion of this segment had disappeared; the same area completely recovered by summer 2001 (personal observation based on fixed transects and field surveys; Morris et al. 2000). This cataclysmic loss and recovery happened despite no major changes in water quality. Thus, there may be many reasons that average seagrass conditions do not mirror average water quality conditions.

CONCLUSIONS AND RECOMMENDATIONS

Both seagrass indices, especially the depth-based seagrass index, could be improved. Recommendations for refining and improving indices include: better bathymetric contouring—a difficult and expensive task; more points measuring the edge of the bed; ground-truthing of the "edge" of the bed as measured from aerial photos versus in the field (Johansson 2001; Virnstein et al. 2001); and improvements in analytical tools for comparing a snapshot of seagrass condition to previous water clarity conditions—e.g., appropriate lag times and previous condition of seagrass. Standard statistics are hampered by temporal and spatial auto-correlation of both seagrass and water quality data. Often, seagrass is monitored on an annual or multi-year basis and water quality on a monthly basis. There is no clear or standardized approach for comparing such "messy" data.

If factors besides water quality affect the condition of seagrass, we cannot expect a high degree of correlation between seagrass and water quality. Suppose, for example, that shrimp burrowing or sediment instability limits the deep edge of seagrass. Then changes in water quality may have no impact on the deep edge of seagrass. Identifying those areas where factors other than light limit the depth of seagrass and eliminating those from the above analyses may vastly improve the light-seagrass relationship and our understanding of it.

Although indices need to be improved, seagrass indices, especially a depth-based seagrass index, are useful tools for characterizing segments and prioritizing restoration needs (Johansson and Greening 2000; Steward 2001).

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COMPLEMENTARY USE OF DIFFERENT SEAGRASS TARGETS AND ANALYTICAL APPROACHES IN THE DEVELOPMENT OF PLRGS FOR THE INDIAN RIVER LAGOON

Joel S. Steward

ABSTRACT

Pollutant load reduction goals (PLRGs) can be developed using two different, but complementary approaches. One approach is dependent on the use of a three-dimensional mechanistic model of the Indian River Lagoon system that should be fully developed by 2002. The other approach is an inference method (a.k.a. a basin spreadsheet model) that can be used now for estimating pollutant loads based on land use and other relevant characteristics that affect loading rates. *Areal coverage* (e.g., acres) of seagrass is the key metric involved in the inference method approach, whereas, *light measured at depth* is the metric used in the application of the mechanistic model. The mechanistic model will be applied toward the development of PLRGs based on their adequacy to meet the seagrass light-depth target. The inference method is used to satisfy the immediate need of certain local governments for planning targets or *provisional* PLRGs, which are intended to be conservative approximations of desired pollutant reductions. The two approaches can be used in complementary fashion to converge on a meaningful set of defensible, resource-based PLRGs.

INTRODUCTION

Pollutant load reduction goals (PLRGs) for the Indian River Lagoon (Figure 1) are aimed at the improvement of water quality or clarity for the sake of seagrass restoration. The relationship between water quality and seagrass is well established (Morris and Tomasko 1993). The relationship suggests that PLRGs should target those primary "optical" pollutants whose concentrations restrict the level of solar light for seagrass growth. In the Indian River Lagoon, total suspended matter and chlorophyll *a* (i.e., phytoplankton) are believed to be critical constituents that significantly contribute to light attenuation (Hanisak 2001; SJRWMD data analysis). In turn, chlorophyll *a* concentrations or phytoplankton abundance may be controlled by nitrogen and/or phosphorus (Phlips and others document 2000). Therefore, total suspended solids (TSS), N, and P are the pollutants of concern that will be addressed in the development of PLRGs. (Dissolved organic matter— also measured as color—is another important constituent that affects light penetration in certain Lagoon areas. However, there is no plan to establish a 'color' PLRG because it is assumed that color concentrations can be reduced by the reduction of base flow and storm discharge volumes that are needed to meet TN, TP, and TSS PLRGs.)

PLRGs for TSS, N, and P will be developed using two different, but complementary approaches. One approach is dependent on the use of a three-dimensional mechanistic model. This model represents the dynamic and interactive ecological processes that regulate light attenuation and, thus, seagrass distribution. The model result will be some measure of light at depth in the Lagoon. If the model-simulated light at a particular depth matches the minimum seagrass light requirement, then it is assumed that seagrass coverage to that depth is possible.

The other approach is an inference method; a simple, mass balance algorithm (a.k.a. a basin spreadsheet model) for estimating pollutant loads based on land use of a specific year and other relevant characteristics that affect watershed loading rates. This approach selects the best-documented year for maximum seagrass coverage— 1943 in this case (the earliest historical

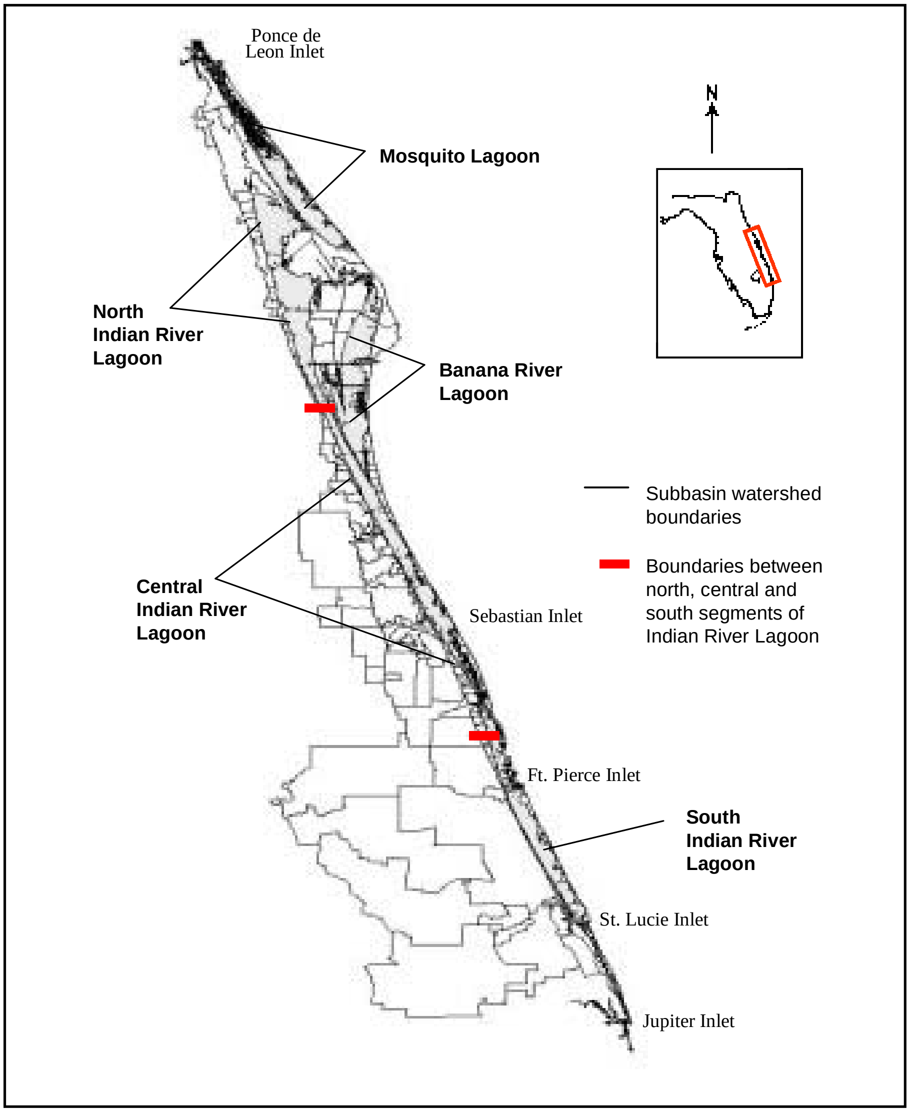

Figure 1. Map of Indian River Lagoon basin showing sub-basins and major lagoon segments (Mosquito Lagoon; Banana River Lagoon; and north, central and south Indian River Lagoon)

period for which there is well-documented seagrass coverage [Virnstein and Morris, 2000])—and applies land use of that year to estimate a watershed loading rate, also referred to as an allowable loading rate. This loading rate can then be translated into a load reduction when subtracted from a current or projected loading rate.

Areal coverage (e.g., acres) of seagrass is the key metric involved in the inference method approach, whereas, *light measured at depth* is the metric of choice used in the application of the mechanistic model. Each metric defines a different type of seagrass target and means by which the seagrass resource can be monitored. Both approaches are similar in that they rely on the empirical analyses of monitoring data to develop the targets and they provide current assessments relative to seagrass acreage, depths of coverage, and light levels.

The two approaches are described in more detail below, including how the two approaches and their respective seagrass targets are utilized in complementary fashion to develop provisional and final PLRGs.

MECHANISTIC MODEL APPROACH FOR DEVELOPING FINAL PLRGS

Final PLRGs for the Lagoon will be derived from a 3-D mechanistic model, which is an integration of several established models: CH3D; wave and sediment transport model; CH3D-WQ; Gallegos optical model; and either Fong's or Montague's SAV model (Sheng 1999; Gallegos and Kenworthy 1996; Fong and Harwell 1994; Fong and others 1997; and Montague, in development, 2001). This integrated model, known as the Pollutant Load Reduction (PLR) Model is, in effect, an enhanced light model that accounts for all the major hydrodynamic, water quality, biological and optical processes that interact to produce a submarine light result for the Lagoon (Steward and others 1996) (Figure 2). The PLR Model result can be expressed as light extinction or, more practically, as a percentage of the sub-surface light at some targeted depth. In this case, sub-surface light is the light penetrating the water surface and does not include the reflected incident light.

The component (sub-)models of the PLR Model (Figure 2) are presently being calibrated and verified with extensive sets of data collected in the Lagoon during the past several years. Once calibrated and verified, the real power of the model is realized—its predictive capability. Given that capability, the model will be applied in the assessment of recommended PLRGs based on their level of adequacy to meet the seagrass light-depth target. The model will also be used to assess certain large-scale strategies proposed for achieving PLRGs

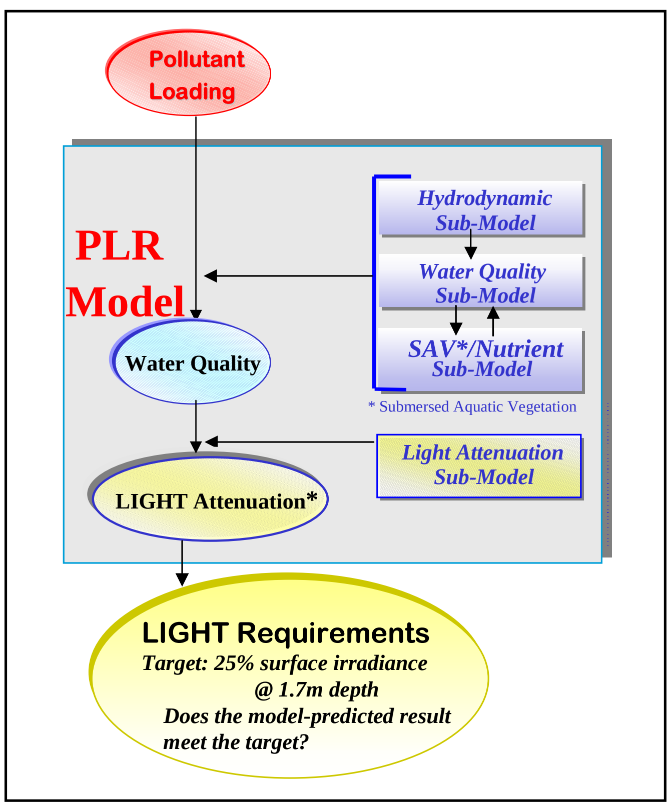

Figure 2. Schematic of the PLR Model and its component sub-models. Pollutant loading is one of several inputs to this mechanistic model; the major output is the amount of solar light available at any given depth.

(e.g., diversion of inter-basin drainage to reduce loading or removal of causeway bridges to increase flushing, etc.). This model should provide a comprehensive, objective, and scientifically defensible means toward developing and achieving final PLRGs.

Perhaps one disadvantage of this modeling approach is the length of time required to develop the framework of the integrated PLR Model and to fully calibrate and verify model results. PLR Model development began in early 1995; its completion date is set for the end of 2002. The application of the PLR Model in the development of recommended PLRGs should occur during 2002–2004.

THE INFERENCE METHOD APPROACH FOR DEVELOPING PROVISIONAL PLRGS

Awaiting the development and application of the PLR Model is not timely in some instances. There is a more immediate need on the part of certain local governments in the Lagoon basin to use pollutant load reduction estimates in their current water management planning. Because of this need, *provisional* PLRGs are being developed and provided to those local governments. These provisional targets, as a precursor to the final PLRGs, are intended to be conservative, within-order-of-magnitude approximations of pollutant reductions that may be necessary for seagrass restoration. They are considered as planning targets that can be used to expedite work on conceptual surface water management designs. It is also intended that provisional PLRGs will be evaluated through application of the PLR model, and refined as necessary to produce final PLRGs.

Because of this immediate need for provisional PLRGs, the method for their development must be fairly rapid. If the requisite land use data are available, then the inference method can be applied to most watersheds in a matter of weeks to produce reasonable load reduction estimates for total nitrogen (TN), total phosphorus (TP), and TSS.

The expected utility of the inference method, as applied to Lagoon watersheds, is based on two major assumptions:

1. Restoration of seagrass coverage to some historical period (e.g., 1943) requires that the water quality or clarity of the Lagoon would need to be restored to that historical period.

2. Water quality restoration would be achieved by reducing the loads of pollutants that affect water clarity to levels consistent with that historical period.

The inference method relies on a simple, mass-balance algorithm known as the Pollution Load Screening Model (PLSM) (Adamus and Bergman 1995). The PLSM incorporates land use and soil type data, and rainfall-runoff coefficients to calculate annual, non-point, surface water loads of TN, TP, and TSS. This spreadsheet model is applied to generate three main outputs—current loads (c. 1995), historical loads (c. 1943), and future or “build-out” loads (c. 2010 to 2020)—by varying the land uses for each of the representative years. The historical loads are considered the *allowable* loads. The difference between the “build-out” and the historical or allowable pollutant loads is the estimated load reduction. It is important that the “build-out” load estimates be considered in the development of provisional PLRGs to provide some reasonable assurance that allowable loading rates will not be exceeded in the future.

Generating realistic results depends on the accuracy of the input data (e.g., land uses, runoff and pollutant concentration coefficients). Refinements to the input data relative to the current conditions in the selected drainage basin are required to achieve good PLSM estimates of current pollutant loadings. These refinements are, in effect, a calibration of the PLSM that improves confidence in its ability to hindcast and forecast loading estimates. For example, data on current land use are quality assured and corrected where necessary. (Similar checks are made on historical land use utilizing various photo and map materials, and for future land use, professional judgement is used to check the reasonableness of land use transitions from present to future.) A Geographic Information System or GIS becomes an invaluable tool at this point. Also, rainfall-runoff coefficients and pollutant concentrations relative to the basin's soils and land uses (especially percent imperviousness) are updated using results from recent studies conducted in north and central Florida (Hendrickson and Konwinski 1998; Harper 1994).

APPLICATION OF THE PLSM INFERENCE METHOD TO THE CRANE CREEK WATERSHED

Crane Creek is a tributary to the Indian River Lagoon that drains approximately 13,450 acres of urban land in and around Melbourne, Florida (Figure 3). The area of the Lagoon receiving Crane Creek discharge has suffered a 77% loss of seagrass coverage between 1943 and 1996 (SJRWMD data). According to the provisional PLRG method and its assumptions, historical loadings of TN, TP, and TSS were based on 1943 land use (interpreted by Dynamac Corp. 1997; archived at SJRWMD). The historical loading estimates were also based on an average annual rainfall volume of 55.25” calculated for that time period (1932–1952, Titusville, Florida; National Climatic Data Center).

Current and future loading estimates were based on 1995 and projected 2010/20 land use data, respectively (SJRWMD GIS data base and Brevard County Comprehensive Land Use plans), and a recent period, annual rainfall volume of 48.13” (1960–1995, Melbourne Airport; National Climatic Data Center).

Runoff coefficients were developed for the recreational/open space and the transportation land cover classes since no literature-derived coefficients were found to be applicable. Runoff coefficients were derived from the estimation of the pervious/impervious surface ratio within the subject land cover classes interpreted from 1995 digital orthographic quads. The appropriate pollutant concentrations obtained from the literature were then applied to calculate annual loads.

Pollutant load reduction or treatment coefficients were applied to the current and future loading estimates of TN, TP and TSS. This was necessary because most development since 1984 is required to comply with regulatory stormwater treatment criteria (Chapter 40C-42 of the Florida Administrative Code). Treatment coefficients were applied to areas where land use changes occurred from the mid-1980s to 1995 and then to 2010/20. The earliest land cover information available for the 1985–1995 time period is from 1989 (SJRWMD aerial photography and GIS database). Where changes in land use between 1989 and current or future conditions were detected, treatment coefficients were assigned as follows: TN = 0.30, TP = 0.50, and TSS = 0.70 (i.e., 30% reduction TN, 50% reduction TP, and 70% reduction TSS).

It is clear that extensive urbanization has been occurring in the Crane Creek sub-basin since 1943. Furthermore, development and modifications in drainage systems caused an expansion of the sub-basin by about 660 acres. These factors are responsible for the substantial increases in

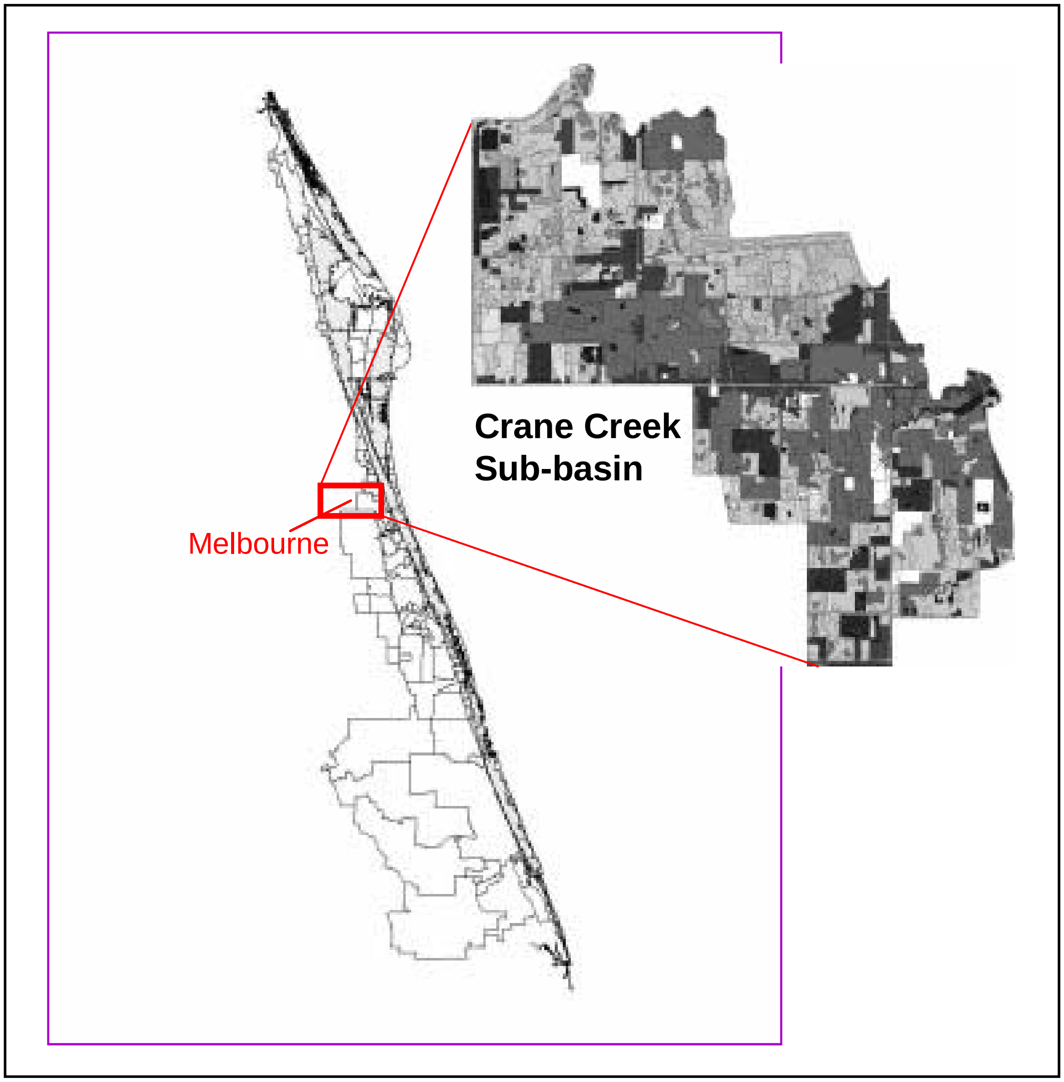

Figure 3. Location of the Crane Creek sub-basin in the Indian River Lagoon basin.

nutrient and TSS loadings to the creek estimated by the PLSM. For example, TN, TP, and TSS loadings have increased 77%, 159%, and 153%, respectively, from 1943 to 1995. These increases even take into account the treatment coefficients for new development since 1984/85. The increases in nutrient and TSS loadings from 1995 to c. 2010/20, also factoring in the treatment coefficients are 20% (TN), 18% (TP), and 19% (TSS).

The provisional PLRGs are calculated by taking the difference between the future loading (with treatment coefficients) and the 1943 loading (Figure 4). In terms of areal load rates, the provisional load reductions are up to 3.6 lb/ac/yr TN, 0.6 lb/ac/yr TP, and 140 lb/ac/yr TSS. In short, the provisional PLRGs for the Crane Creek basin represent a 53% (TN) and 67% (TP and TSS) reduction in future or "build-out" annual loadings.

Good to excellent agreement was found between PLSM-calculated and measured results for TN and TP (14%; Table 1). But, the PLSM result for TSS was approximately 130% above the measured result (Table 1). Much of this large difference may be explained by the sedimentation of a portion of the incoming TSS load. In the

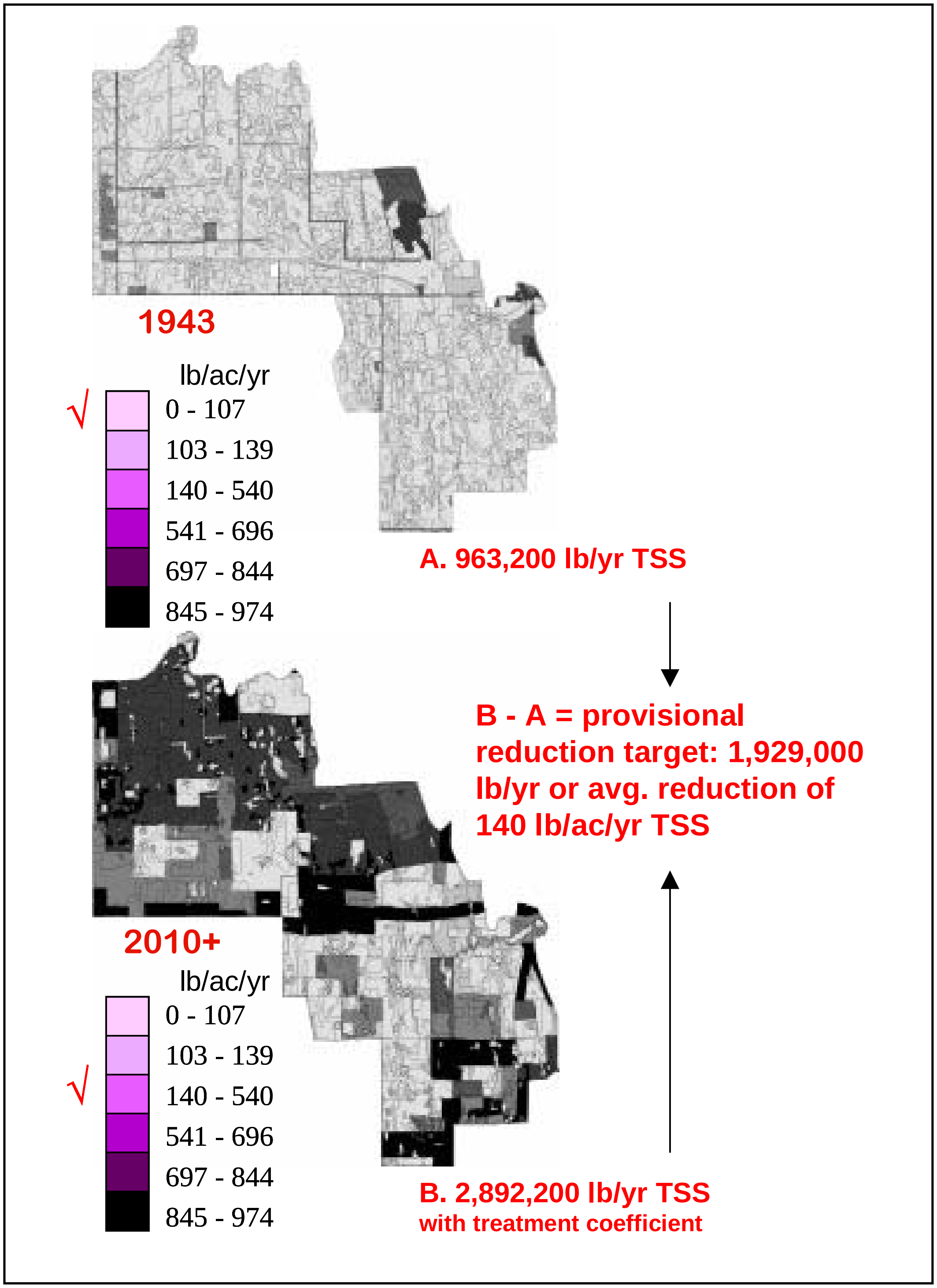

Figure 4. Crane Creek sub-basin TSS loading: comparison between the 1943 and 2010 ("build-out") TSS loading; the difference between the two yields the provisional pollutant load target.

Table 1. Comparison between PLSM-estimated and measured loading rates, Crane Creek.

	TN lb/yr	TP lb/yr	TSS lb/yr
PLSM-estimates loadings, 1995	117,400	15,400	2,436,700
Measured loadings (1991–1998)	125,700	13,400	1,050,000

case of Crane Creek, over 56% of the total annual TSS load generated from the various points of input to the creek (represented by the PLSM result) may drop out along the length of the creek to its mouth (where the measured result is taken) (Steward and Green document 2000).

COMPLEMENTARY USE OF THE PLR MODEL AND INFERENCE METHOD

Both approaches, the PLR Model and the inference method, can be used in complementary fashion to determine final PLRGs (Figure 5). However, before either approach is taken, two types of empirically derived seagrass targets are determined:

1. *The historical year coverage (acres) that is the maximum coverage found for any given segment of the Lagoon*. For most of the Lagoon, the maximum seagrass coverage that is documented is based on 1943 aerial photography (Virnstein and Morris 2000).

2. *The maximum depth to which seagrass can grow in the Lagoon (1.7 m) and the minimum amount of sub-surface light that would allow seagrass to grow at that depth (25% of sub-surface light;* see Morris and others , this publication). It's interesting to note that throughout most of the Lagoon the maximum depth extent of 1943 seagrass coverage is about 1.7 m (SJRWMD mapped bathymetry and seagrass coverages).

Establishing such seagrass targets enables the meaningful use of both the inference and mechanistic model approaches. The inference method approach assumes that c. 1943 land use can be used to estimate the allowable pollutant loading rates. Then, those results can be evaluated via PLR model simulation runs as a means to determine a final and more realistic solution. It may actually mean model-testing different sets of loading rates, including the estimated 1943 rates, to eventually "bracket in" on an optimum set of final PLRGs.

The PLR model output is a light-at-depth result; therefore, the seagrass light-at-depth target will be used to assess whether the result is desirable. If the 1943 loading rates or some other set of proposed loading rates produce the desirable result, then a recommended final PLRG is close at hand.

CONCLUSIONS

There are clear advantages associated with using multiple seagrass targets and analytical approaches in the development of PLRGs. These advantages are briefly explained below.

- The adoption of different types of seagrass targets allows the utilization of different analytical approaches to converge on a meaningful and defensible PLRG. Use of a single target may restrict one's options in the selection of an analytical approach.

- The use of multiple seagrass targets presumes that multiple aspects of the resource will be monitored (e.g., areal coverage and depth extent of seagrass, light and "optical" pollutant levels). This approach promotes a more

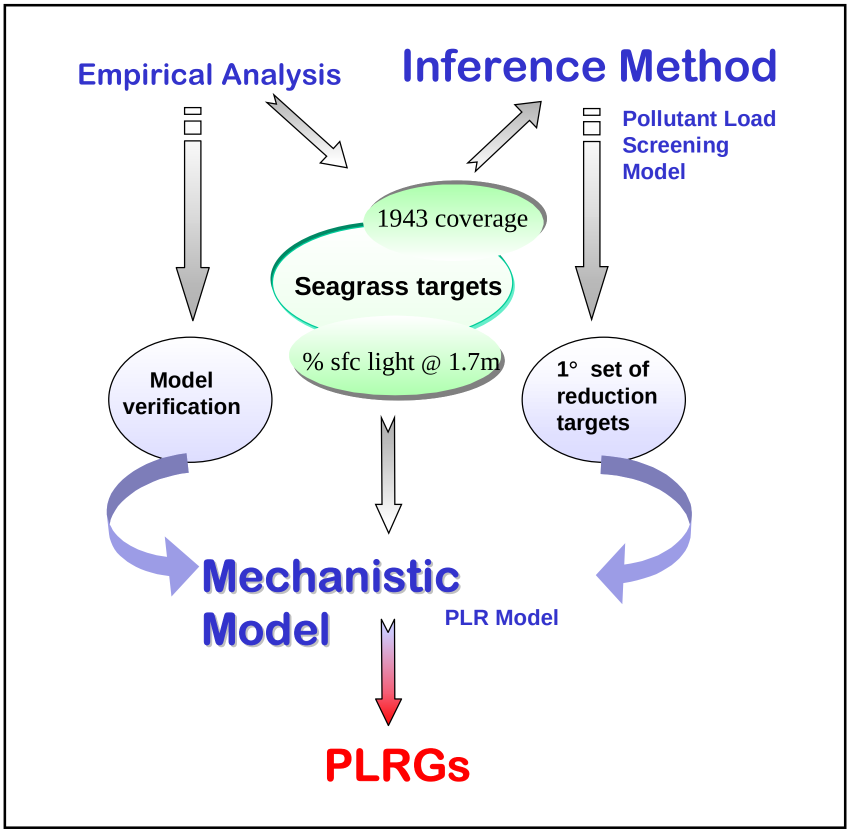

Figure 5. Flow diagram showing the relationship among the complementary approaches used to develop PLRGs with an emphasis on the inference method and the mechanistic model approaches.

comprehensive understanding of seagrass ecology, which leads to better management of the resource.

- The use of an inference method (i.e., basin spreadsheet models) allows the rapid development of *provisional* PLRGs that can be easily used in the development of watershed plans. These provisional PLRGs should be considered as conservative approximations, which can help determine high-end estimates for levels of treatment and their associated costs.

- The inference method approach should be coupled with a more robust empirical or mechanistic model approach to refine PLRG estimates, particularly if costs to implement a watershed plan become an issue (and they usually are). The mechanistic model is recommended because of its predictive capability.

The utilization of the analytical approaches described above is believed to strike a fair balance between the immediate planning needs of local governments and the restoration needs of the seagrass resource. The next challenge is the establishment of an equitable allocation of the PLRGs among the major watershed sources of pollutant loading. Allocation will likely become a process unto itself which may need be deliberated and agreed upon among the regulatory agencies and affected local jurisdictions. Nonetheless, the PLSM and the PLR Model can both serve a purpose in the allocation process. The PLSM can roughly estimate the proportional allocation of PLRGs among watershed sources and then the PLR model can help refine or optimize that allocation for the benefit of the seagrass resource.

ACKNOWLEDGEMENTS

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LIGHT ATTENUATION BY COLOR, CHLOROPHYLL *A*, AND TRIPTON IN INDIAN RIVER LAGOON

David Christian, Y. Peter Sheng

ABSTRACT

Six synoptic sampling trips were made in the Indian River Lagoon (IRL) between April and June 1997 to collect TSS (total suspended solids), color, chl-*a* (chlorophyll-*a*), and light (photosynthetically active radiation [PAR]) data. These data were used to develop our understanding of light attenuation dynamics and for verification of a numerical light model.

Data from our study show that tripton (non-algal particulate matter calculated here from TSS and chl-*a* corrected for pheophytin data) has a dominant effect on light attenuation in the IRL. When the average downwelling light attenuation coefficient (K_dPAR) for each synoptic trip is plotted against average tripton concentration, a positive relationship is found. A negative relationship is found between average K_d(PAR) and average color, while no clear relationship is found between K_d(PAR) and chl-*a* concentration. The relationship between color and K_d(PAR) is the only one of the three to show significance at the 0.05 level. A positive relationship is also found between average wind speeds and average tripton concentration. Even though this relationship is not significant at the 0.05 level ($p = 0.1842$), it suggests that increased wind speed led to increased tripton concentration, probably as a result of sediment resuspension and turbulent mixing.

Relative contributions of color, chl-*a*, and tripton to light attenuation were found to be 25.2%, 9.6%, and 60.5% of K_d(PAR), respectively.

A numerical light attenuation model based on Gallegos (1993) has been developed to calculate K_d(PAR) using TSS, color, and chl-*a*. The model results compare well with data.

INTRODUCTION

The Indian River Lagoon (IRL) is a 341 km long lagoon on the east coast of Florida between Jupiter Inlet to the south and Ponce de Leon Inlet to the north. The IRL averages about 1.7 m in depth and ranges in width from 0.4 km to 12.1 km (Steward et al., 1994). It contains between 70,000 and 90,000 acres of seagrass beds (Virnstein and Morris, 1996). The seagrass, along with macroalgae in the IRL, helps provide habitat for fishes and invertebrates, contributes to nutrient cycling and the detrital food web, and helps stabilize sediments and protect the shoreline (Busby and Virnstein, 1993). The IRL seagrass beds therefore contribute largely to the estimated $1 billion annual economic impact that fishing has on the region (Virnstein and Morris, 1996). The amount of photosynthetically active radiation (PAR) reaching the seagrass is critical. If poor water quality causes a decrease in PAR reaching the seagrass, the total amount of seagrass is also likely to decrease (Busby and Virnstein, 1993).

In order to ensure water quality conditions conducive to seagrass growth, the University of Florida (Sheng et al., 2000), with support from the St. Johns River Water Management District, is developing an IRLPLR (Indian River Lagoon Pollutant Load Reduction) Model. The goal of the model is to determine pollutant load reduction goals (PLRGs) for the IRL. The IRLPLR model includes a hydrodynamic model, a sediment transport model, a water quality model, a light attenuation model, and a seagrass model. This paper describes part of our effort in the understanding of light attenuation dynamics and the development of a light attenuation model.

METHODS

For this research, six sampling trips were made to the Indian River Lagoon (IRL) on a biweekly schedule between April 8 and June 25, 1997. Forty-five sites (numbered 1 to 45) throughout the IRL were sampled each trip with the lower numbered sites being toward the south. Samples for total suspended solids concentration (TSS), color, and chlorophyll *a* concentration (chl-*a*) were collected at each site using a modified Niskin bottle at 20% and 80% of the total depth using Quality Assurance Quality Control (QAQC) procedures outlined in Melanson and Sheng (1997). For data analysis and modeling purposes, the upper and lower measurements were averaged to represent average conditions throughout the water column.

Photosynthetically active radiation (PAR), used to calculate light attenuation, was measured using Licor 4 sensors. On the first trip the light sensors were deployed three times at each site. The first deployment had one just below the surface and one at 20% of total depth. The second through fifth trips had the sensors deployed at 20% of total depth, 50% of total depth, and 80% of total depth. The sixth sampling trip deployed the sensors at just below the surface, 50% of total depth, and 80% of total depth. At least three repetitions were done for each site when possible. The outputs from the sensors were stored in Licor Dataloggers in the field, and converted to light measurements using calibration curves back at the lab.

The light attenuation, K_d(PAR), was calculated using the Lambert-Beer Equation,

(Eq. 1) $$I_z = I_0 * \exp(-K_d(PAR) * z)$$

where I_0 is the light just below the water surface, I_z is the light at depth z, and z is the depth of interest. Equation (1) can be rearranged to calculate K_d(PAR):

(Eq. 2) $$K_d(PAR) = -1/z \ln(I_z/I_0)$$

For our calculations, z is the distance between the upper and lower measurements, I_z is the lower light measurement, and I_0 is the upper light measurement.

Tripton was calculated using TSS and chlorophyll *a* data. A relationship between chlorophyll *a* corrected for pheophytin and dry phytoplankton weight of 1:100 was assumed (Phlips et al., 1995). The dry phytoplankton weight was then subtracted from the TSS data to obtain a tripton concentration.

For partitioning of K_d(PAR) into components, it was assumed that K_d(PAR) is the sum of $K_d(PAR)_{Seawater}$, $K_d(PAR)_{Color}$, $K_d(PAR)_{Chl\text{-}a}$, and $K_d(PAR)_{Tripton}$. $K_d(PAR)_{Seawater}$ is taken to be 0.0384 m^{-1} (Lorenzen, 1972). $K_d(PAR)_{Color}$ was calculated by multiplying color in Pt units by 0.014 $Pt^{-1}m^{-1}$ (McPherson & Miller, 1987). $K_d(PAR)_{Chl\text{-}a}$ was calculated by multiplying chlorophyll *a* concentration (not corrected for pheophytin) by 0.016 $m^2 mg^{-1}$ (Phlips et al., 1995). $K_d(PAR)_{Tripton}$ was then calculated by subtracting the others from K_d(PAR) calculated from measured light data.

Average values for tripton, chlorophyll *a*, color, and K_d(PAR) were plotted against location to determine spatial trends. Average K_d(PAR) for all of the sites for each trip was plotted against tripton, color, and chl-*a* averaged for each trip.

TSS, chlorophyll *a*, and color were then used as inputs for a numerical light attenuation model based on a spreadsheet model for the Indian River Lagoon developed by Gallegos (1993) and translated into FORTRAN by Kornick (1998). The output from the model, which also takes into account attenuation by water, was compared to the K_d(PAR)s which were calculated from light measurements.

The numerical model was chosen due to its robustness. This model calculates K_d(PAR) using the equation:

(Eq. 3) $K_d(PAR) = 1/\mu_0[a^2+G(\mu_0)ab]^{1/2}$

where μ_0 is the cosine of the solar zenith angle, $G(\mu_0)$ is a function which depends on the cosine of the solar zenith angle and the photic depth of interest, *a* is absorption, and *b* is scatterance (Kirk,1984). The scatterance was calculated from suspended particles expressed as turbidity. The absorption is the sum of absorption due to the water itself, phytoplankton, dissolved yellow matter, and detritus. Absorption due to water was taken from literature values (Smith and Baker, 1981). Absorption due to phytoplankton was estimated by multiplying the inputted chlorophyll *a* concentration by the literature values for chlorophyll-specific absorption (Prieur and Sathyendranath, 1981). Color in Pt units is the value used to find absorption due to dissolved organic matter. The color measurements were done using the Hazen method in which the water color is compared visually to standard Pt-Co solutions (Kornick, 1998). The color value is used to calculate the absorption of dissolved yellow matter at 440 nm wavelength. Turbidity was then used again to calculate absorption by detritus.

For the above calculations, four adjustable coefficients are used. The first is the spectral slope used to find the absorption due to dissolved yellow matter at wave-lengths other than 440 nm. The other three are used in the calculation of absorption by detritus. The first is the longwave absorption cross section of the particle; the second is the absorption due to detritus at 400 nm; and the third is the spectral slope used to calculate the absorption at wavelengths other than 400 nm.

The coefficients calibrated by Gallegos (1993) in the Indian River Lagoon were for two different sites. The coefficients used here were recalibrated using Gallegos's coefficients as a basis (Christian, 2001). The model was originally developed for use with turbidity as opposed to TSS. Since the suspended sediment portion of the IRLPLR model models TSS, TSS is input, and then converted to turbidity for use in the light attenuation model.

Sediment size varies from the southern portion of the lagoon to the northern portion. From site 22 north to site 45, the sediment size stays fairly stable with a D_{50} of about 0.14 mm. We assume that sediment size plays a role in the relationship between TSS and turbidity. In order to account for this, a relationship between TSS and turbidity was found for this region using data provided by the St. Johns River Water Management District Water Quality Monitoring Network:

(Eq. 4) $\text{Turbidity} = (0.4149*\text{TSS}) - 0.1754$ $(r^2 = 0.8955)$

To the south of site 22, the lagoon sediment size varies greatly. Insufficient data were available to create all of the relationships needed. Therefore, the relationship found by Gallegos (1993) was used:

(Eq. 5) $\text{Turbidity} = (0.209*\text{TSS}) + 0.71$

RESULTS

The average K_d(PAR) throughout the IRL for each trip was graphed against the average tripton, color, and chlorophyll *a* concentrations throughout the lagoon for each trip to see if there were any lagoon wide temporal trends. As shown in Figure 1a, the average K_d(PAR) for each sampling trip throughout the IRL is positively correlated with the average tripton through-out the IRL for each trip. It is not significant at the 0.05 level ($p = 0.0790$). Average K_d(PAR) for each trip related to average color with an $r^2 = 0.7671$, but with a negative correlation (Figure 1b), while the average K_d(PAR) for each trip was poorly correlated to average chlorophyll *a* concentration for each trip with an r^2 of essentially 0 (Figure 1c). The relationship

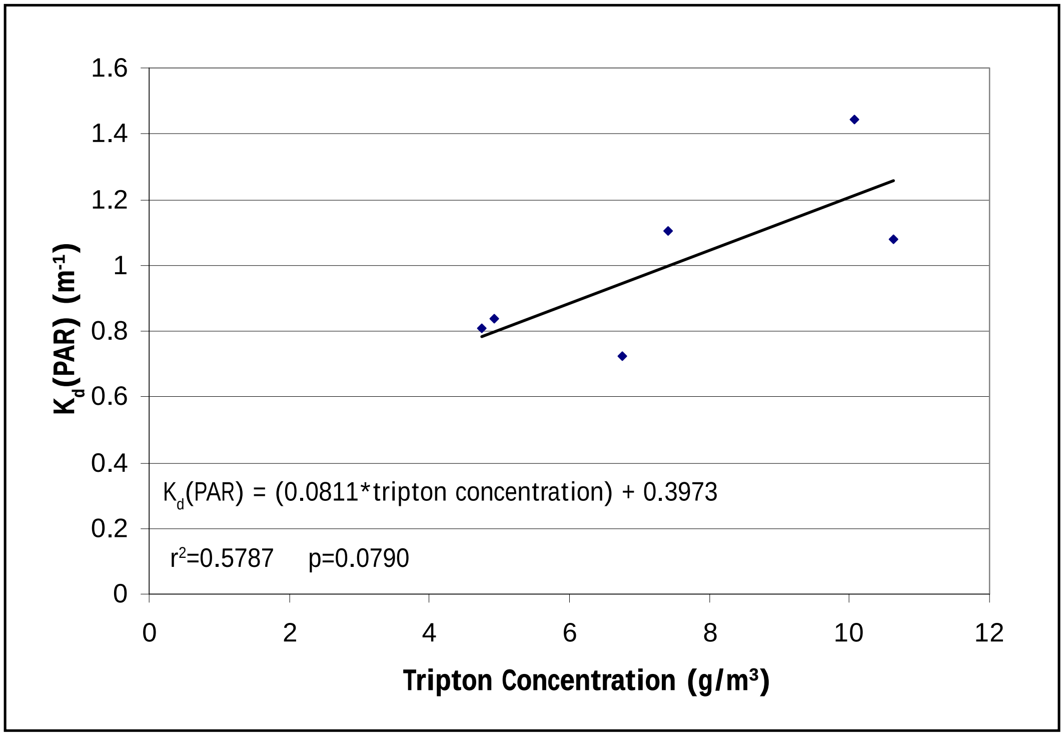

Figure 1a. K_d(PAR) vs. tripton concentration.

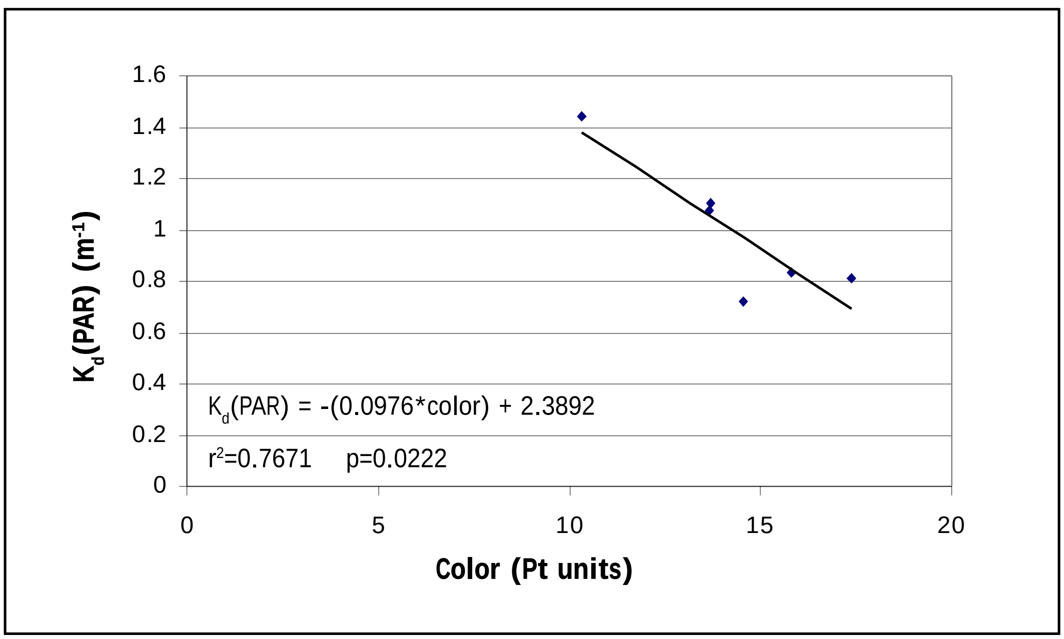

Figure 1b. K_d(PAR) vs. color.

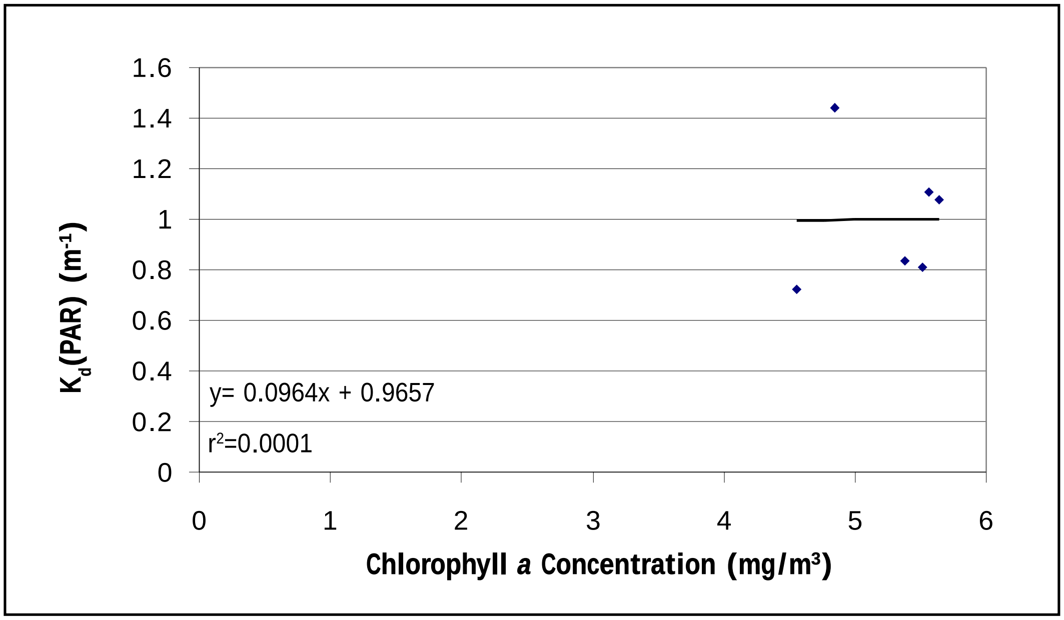

Figure 1c. K_d(PAR) vs. chlorophyll *a* concentration.

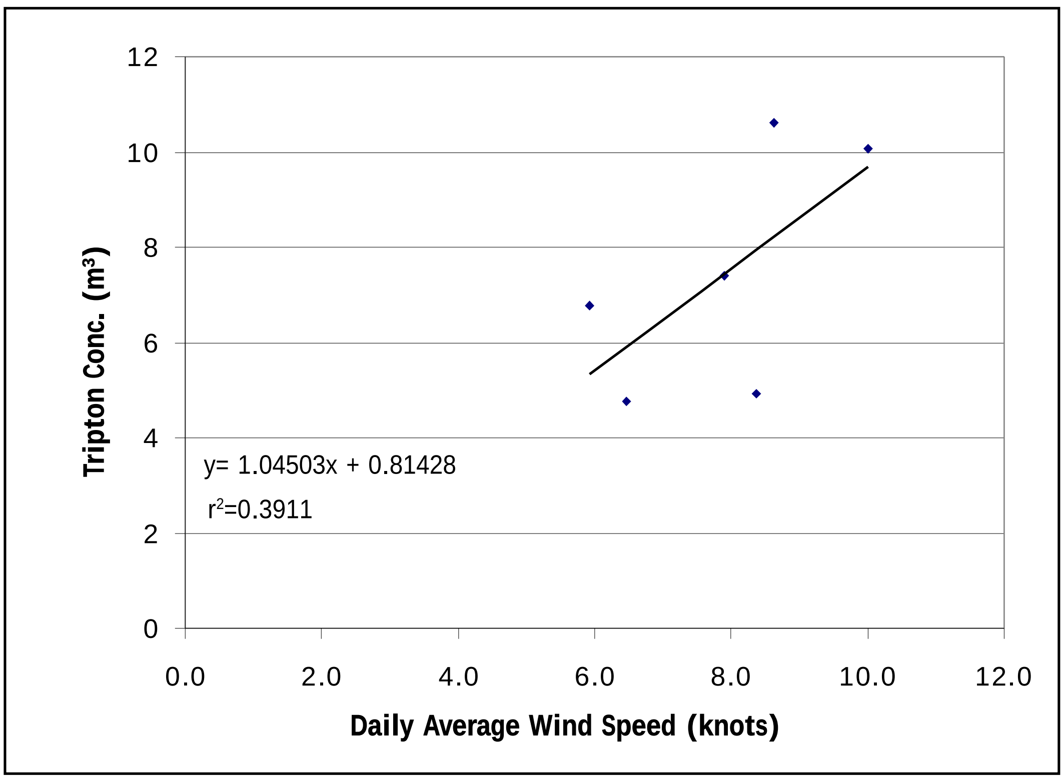

Figure 1d. Tripton concentration vs. daily average wind speed.

between color and K_d(PAR) is significant at the 0.05 level ($p = 0.0222$), but the relationship between chl-*a* and K_d(PAR) is not ($p = 0.9840$).

The average tripton concentration throughout the lagoon for each sampling day was also plotted against the daily average wind speed averaged from three locations around the IRL (NASA Shuttle Facility, Melbourne Airport, and Patrick Air Force Base). The tripton concentration averaged for each trip had a positive correlation to the daily average wind speed (Figure 1d). It is not, however, significant at the 0.05 level ($p = 0.1842$). The average values were used for plotting tripton against wind speed since only daily averaged wind speeds were available. Since sampling occurred throughout the day, wind variations at different times of day did not allow for plotting of each site. The other plots were done the same way for uniformity.

Figures 2a-c show the data from all stations and all sampling dates for the relationships between K_d(PAR) and tripton, color, and chlorophyll *a*, respectively. These relationships show positive relationships between tripton and K_d(PAR) and chl-*a* and K_d(PAR). Both are statistically significant ($p < 0.0001$ for each). The r^2 for the tripton relationship is 0.3774, while for chl-*a* it is only 0.0691, showing no real correlation. Color again shows a negative relationship with K_d(PAR) and while the relationship is statistically significant ($p = 0.0003$), the r^2 for all of the data is only 0.0569, showing no real correlation.

When the percentage of K_d(PAR) due to color, chlorophyll *a* concentration, and tripton concentration are each plotted against K_d(PAR) (Figures 3a-c) a division is seen at K_d(PAR) = 1 m^{-1}. Below K_d(PAR) =1m^{-1} , color and chlorophyll *a* concentration have a greater influence on K_d(PAR) than at higher K_d(PAR)s accounting for 30.1% and 10.8%, respectively. Tripton accounts for 53.5% of K_d(PAR). Above K_d(PAR) = 1, however, tripton accounts for 75.4% of K_d(PAR), while color and chlorophyll *a* concentration account for only 14.7% and 7.1%, respectively. Overall, tripton accounts for 60.5%, while color and chlorophyll *a* account for 25.2% and 9.6%, respectively. Each variable was averaged over all six trips for each site. These averages were plotted against the site numbers (with lowest numbered sites in the south) to determine any spatial trends throughout the sampling period (Figures 4a-d).

The results of the numerical light attenuation model run are shown in Figure 5. The model results compare to the data with an root mean squared error of 0.39 m^{-1}.

DISCUSSION

Possibly the most interesting result was the negative trend when the average K_d(PAR) versus average color. This relationship had the highest r^2 of the regressions performed on the averages at 0.7671. It also shows significance at the 0.05 level ($p = 0.0222$). The plot of all of the K_d(PAR) data versus color data shows a negative trend as well, but unlike the average values, an extremely weak correlation ($r^2 = 0.0569$). Color is considered an important factor in light attenuation in the numerical model, and common sense would dictate that increased color would increase light attenuation. Indeed, color contributed 25.2% of K_d(PAR), second to the 60.5% of K_d(PAR) contributed by tripton. A possible explanation for why color may be lower at the higher K_d(PAR) values may be due to the dominant role played by tripton. Tripton has the dominant influence on K_d(PAR) at higher K_d(PAR)s. Hence tripton may be so dominant that it overwhelmed the effect of color. Figures 6a-b show trends supporting this hypothesis. Figure 6a shows that as average daily wind speed increased, color decreased. In a shallow estuary such as the

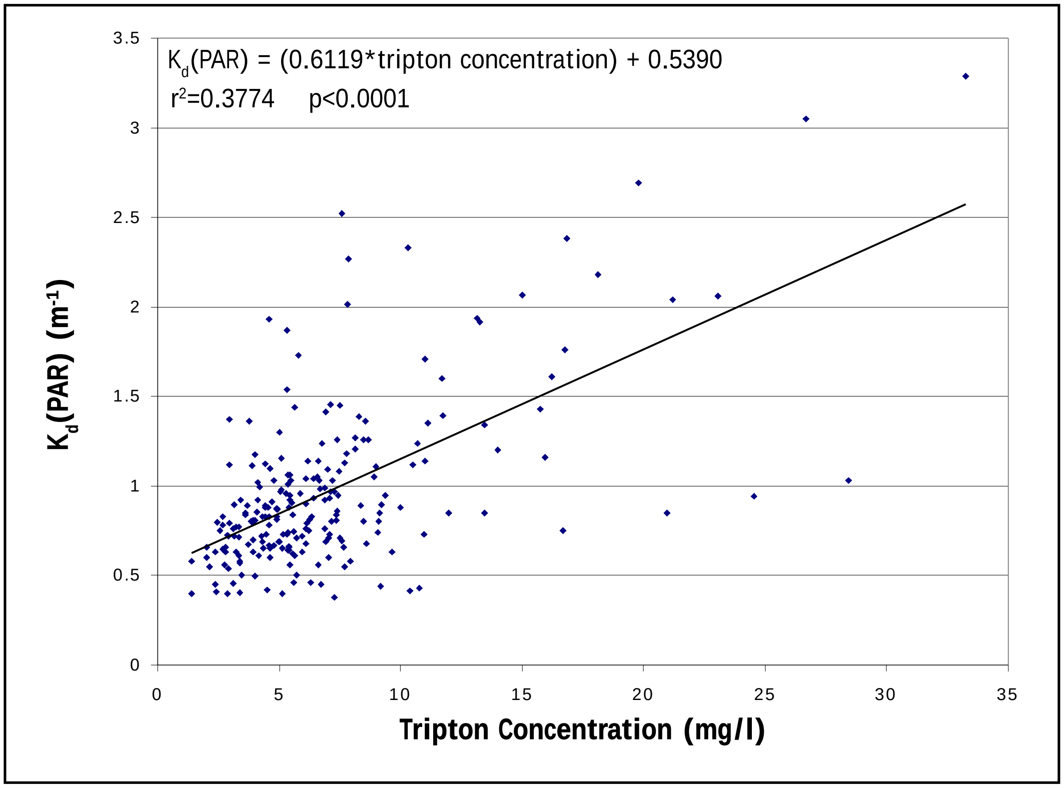

Figure 2a. Kd(PAR) vs. tripton concentration for all data.

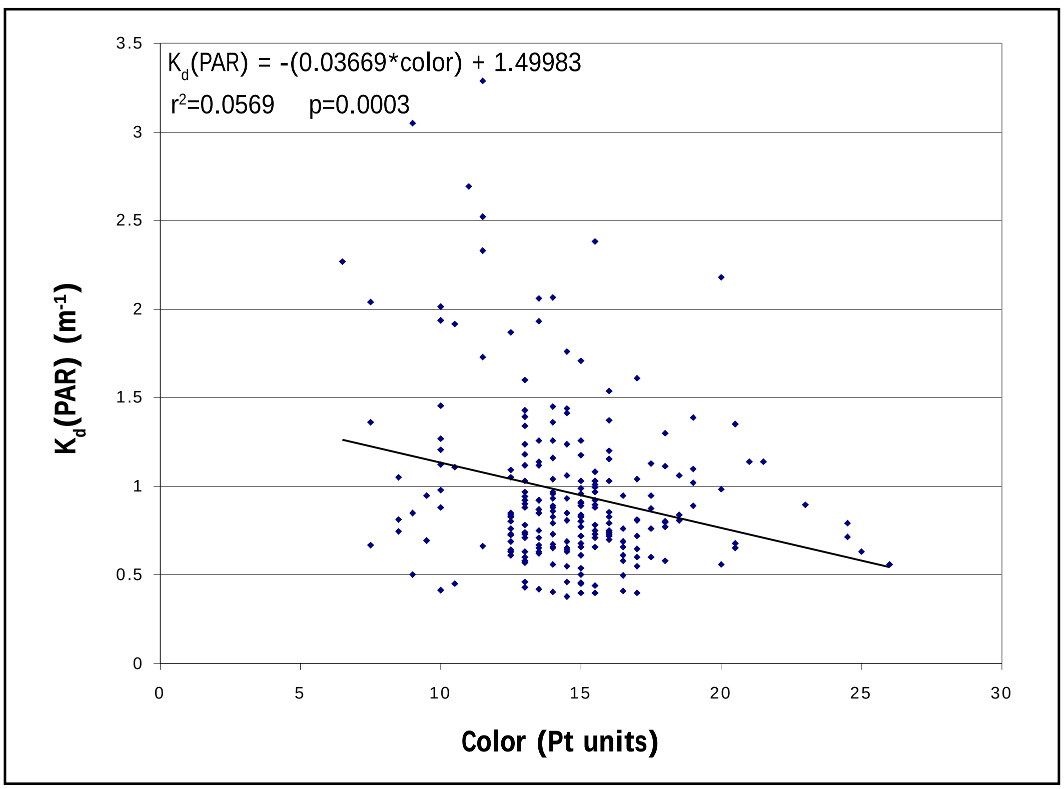

Figure 2b. Kd(PAR) vs. color for all data.

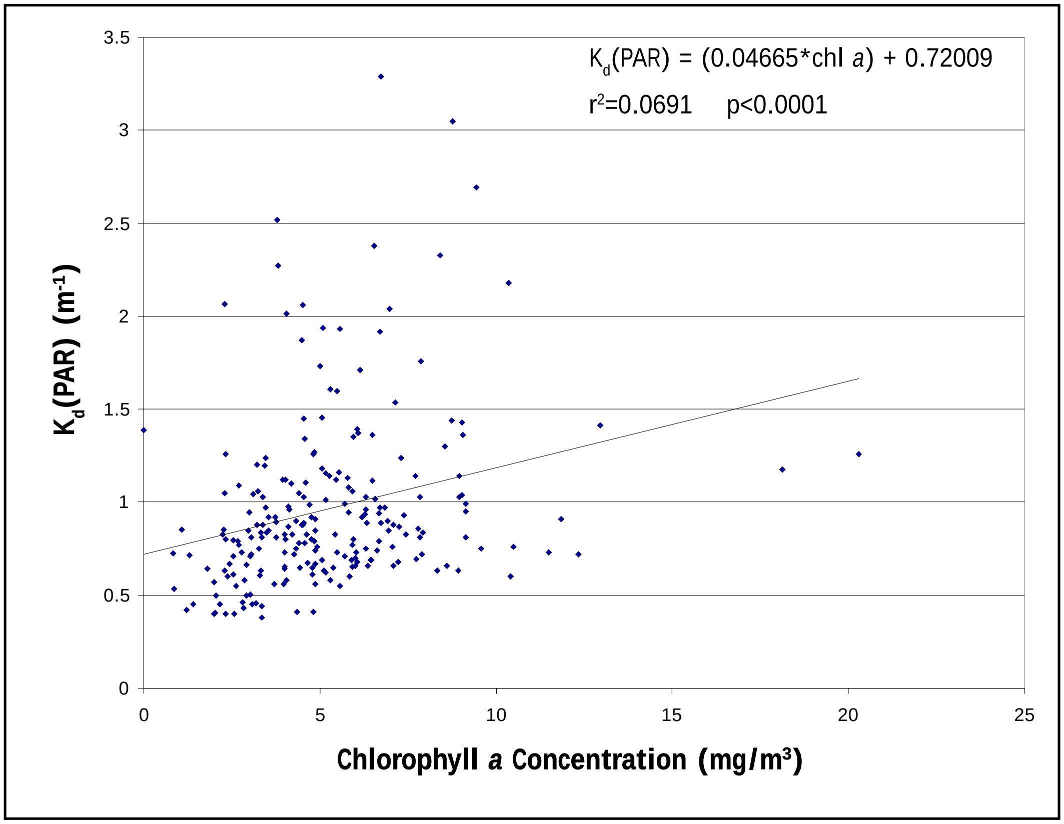

Figure 2c. K_d(PAR) vs. chlorophyll *a* concentration for all data.

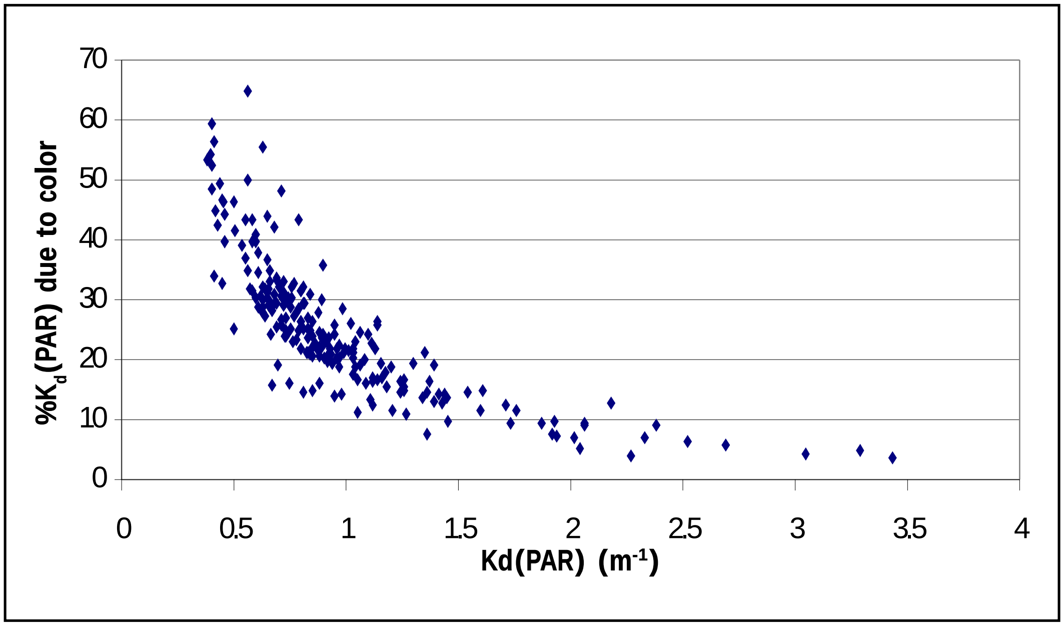

Figure 3a. Percent K_d(PAR) due to color vs. K_d(PAR)

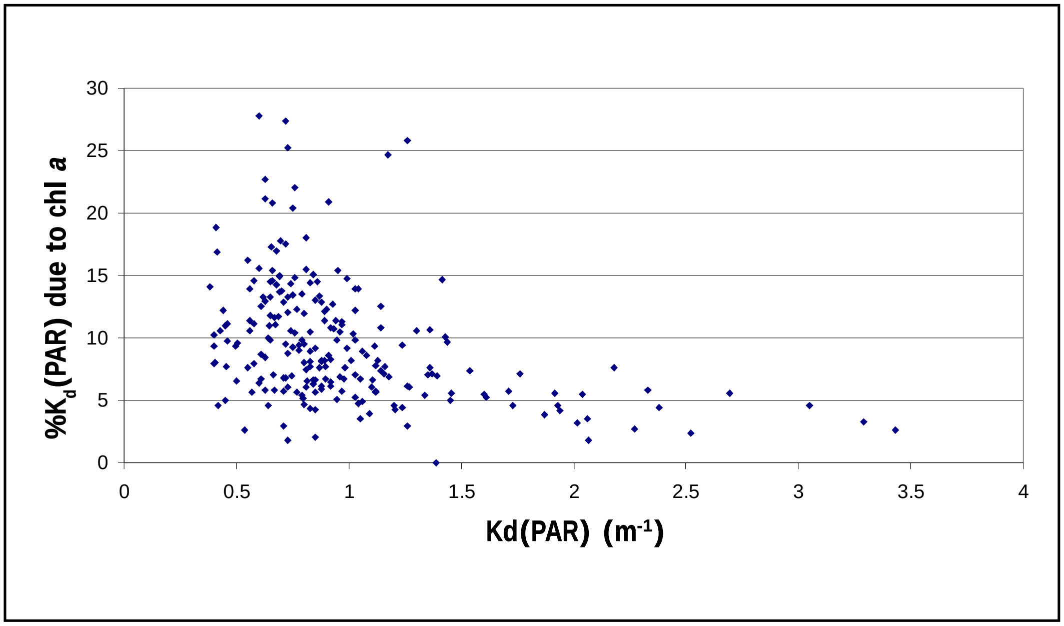

Figure 3b. Percent K_d(PAR) due to chlorophyll *a* vs. K_d(PAR)

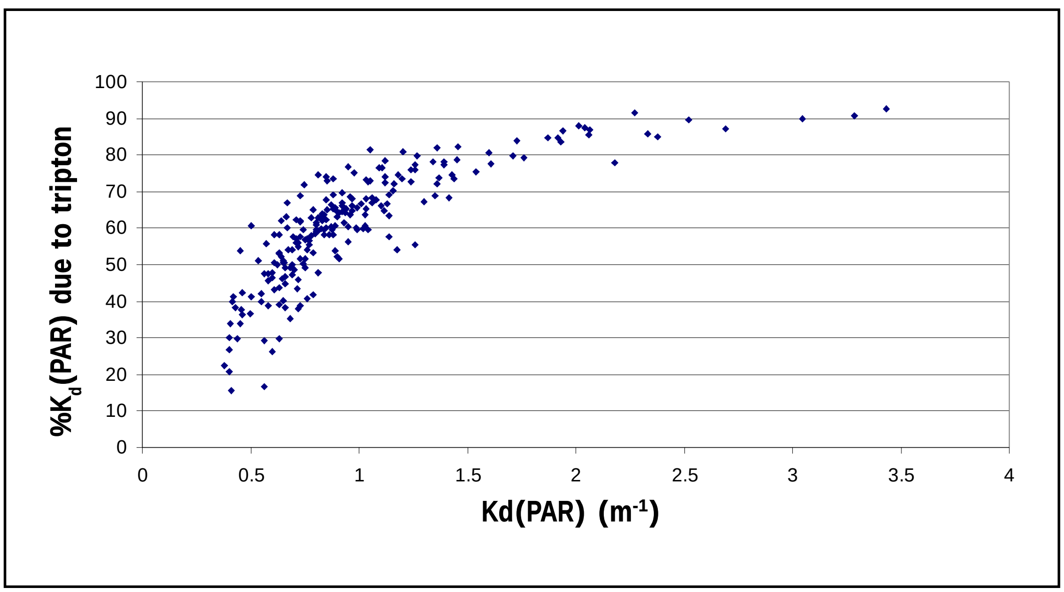

Figure 3c. Percent K_d(PAR) due to tripton vs. K_d(PAR)

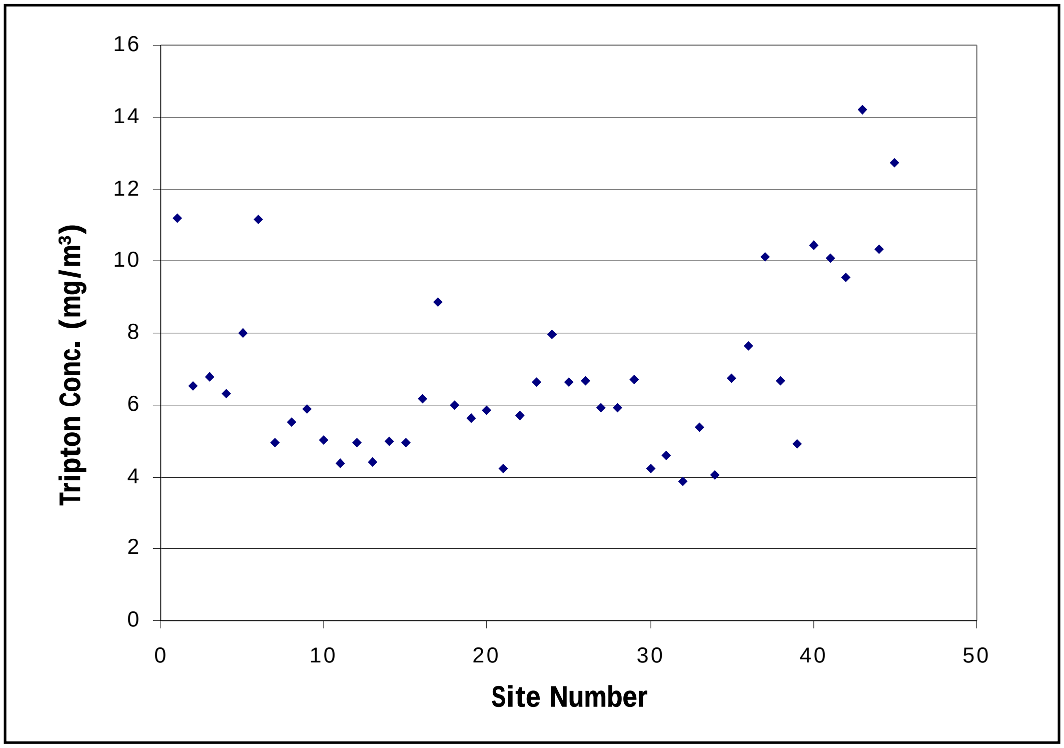

Figure 4a. Tripton concentration vs. site number.

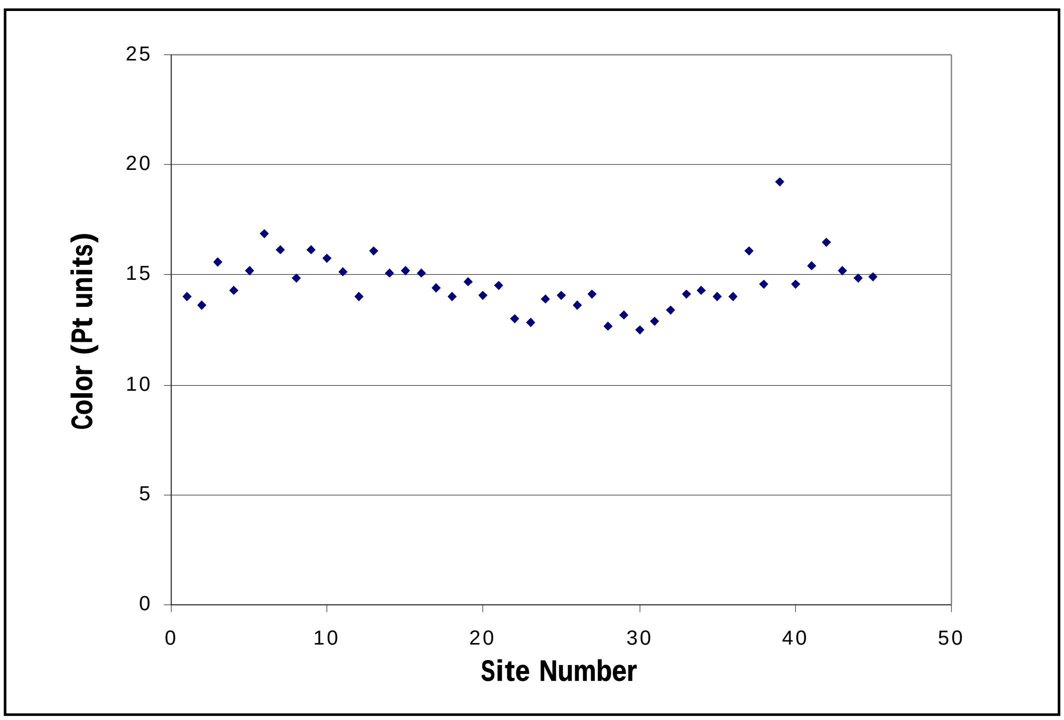

Figure 4b. Color vs. site number.

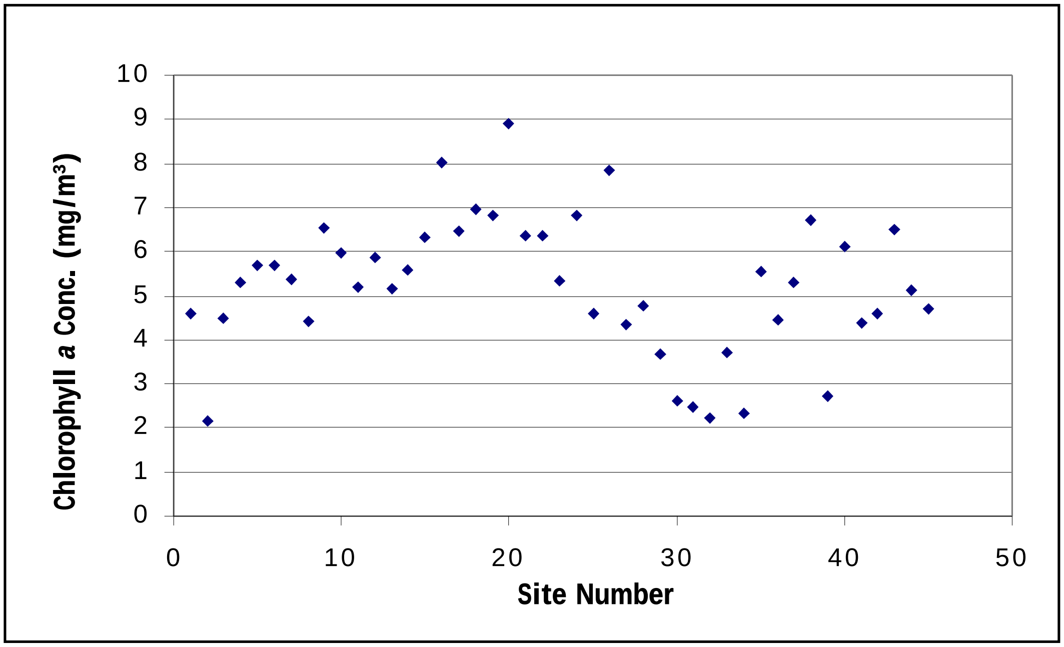

Figure 4c. Chlorophyll *a* concentration vs. site number.

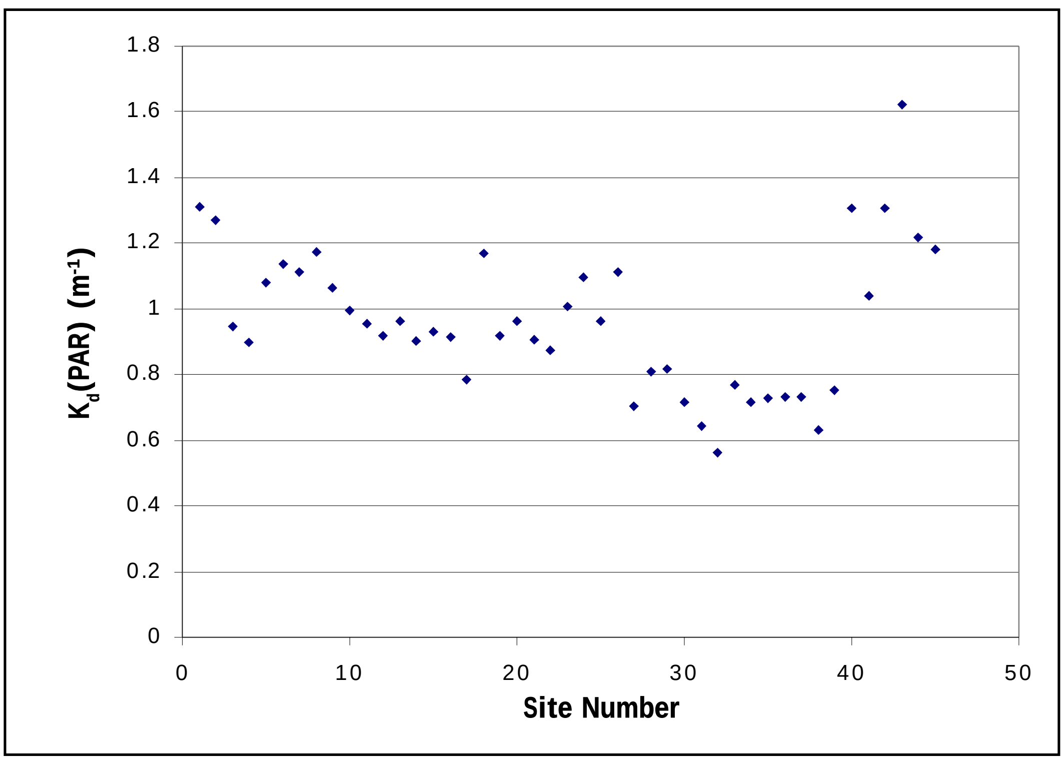

Figure 4d. K_d(PAR) vs. site number.

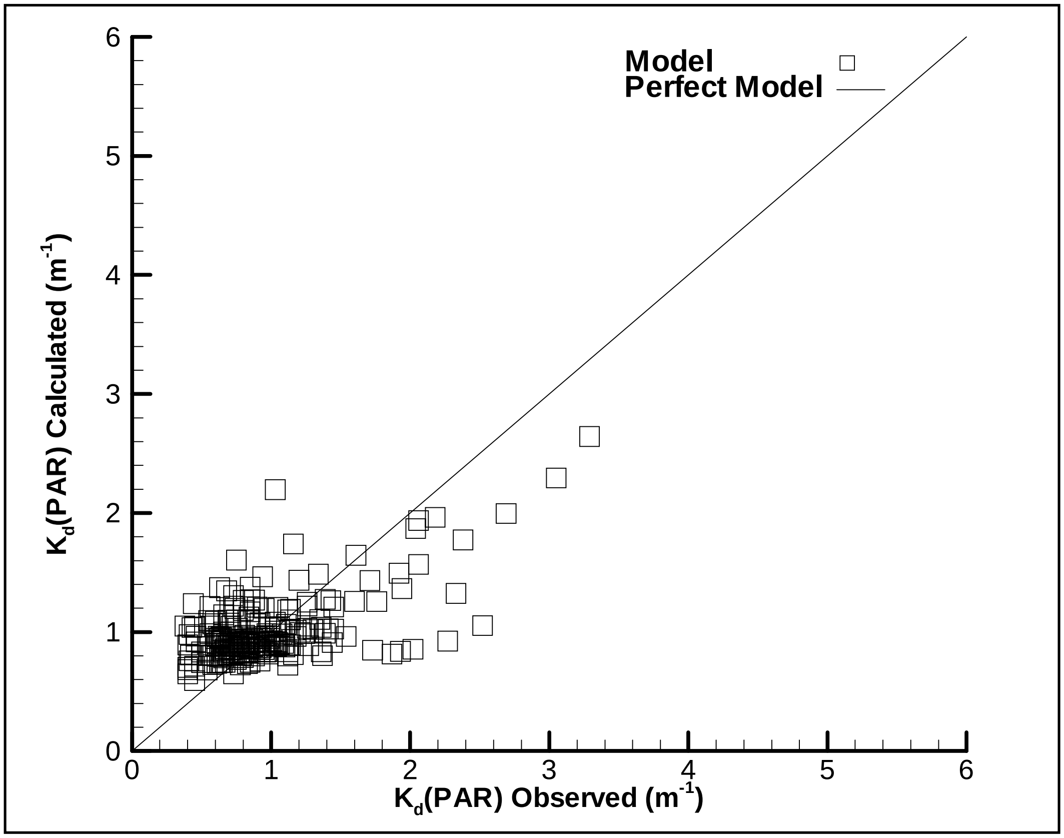

Figure 5. UF synoptic sampling trips 1–6 numerical model results.

IRL, wind can cause suspension of sediment, as seen in Figure 1d. Figure 6b shows that as average tripton concentration increased during the sampling trips, average color decreased.

McPherson and Miller (1987) found the percent contribution of each component to K_d(PAR) for Charlotte Harbor, Florida. Their findings show tripton accounted for 72.5% of K_d(PAR), color from dissolved matter accounted for 21%, and suspended chlorophyll accounted for 4%. Phlips et al. (1995) found percent contributions in Florida Bay. Their results show tripton accounted for 75% of K_d(PAR), chlorophyll containing particles accounted for 14%, and color accounted for 7%. Both show a higher percentage due to tripton than in the IRL. Percent due to color in Charlotte Harbor was comparable to the IRL, while it was much lower in Florida Bay. Chlorophyll *a* accounted for a higher percentage of K_d(PAR) in the IRL than in Charlotte Harbor, but less than in Florida Bay.

Figure 1d shows there is a positive correlation between the average daily wind speed and tripton concentration, but with an $r^2 = 0.4095$. It is not significant at the 0.05 level ($p = 0.1842$) Many factors may come into play here. While daily average wind speed was obtained, we could not obtain either wind direction, or wind speed at the exact time of sampling. Since the IRL is much longer than it is wide, wind direction would have a great effect on resuspension. Wind out of the north or south would have a much greater fetch than wind out of the east or west, thus being able to create more resuspension. Also, knowing the wind at the time and place of each sample would have improved this analysis. While it may have been calm in the morning when sampling was done, it may

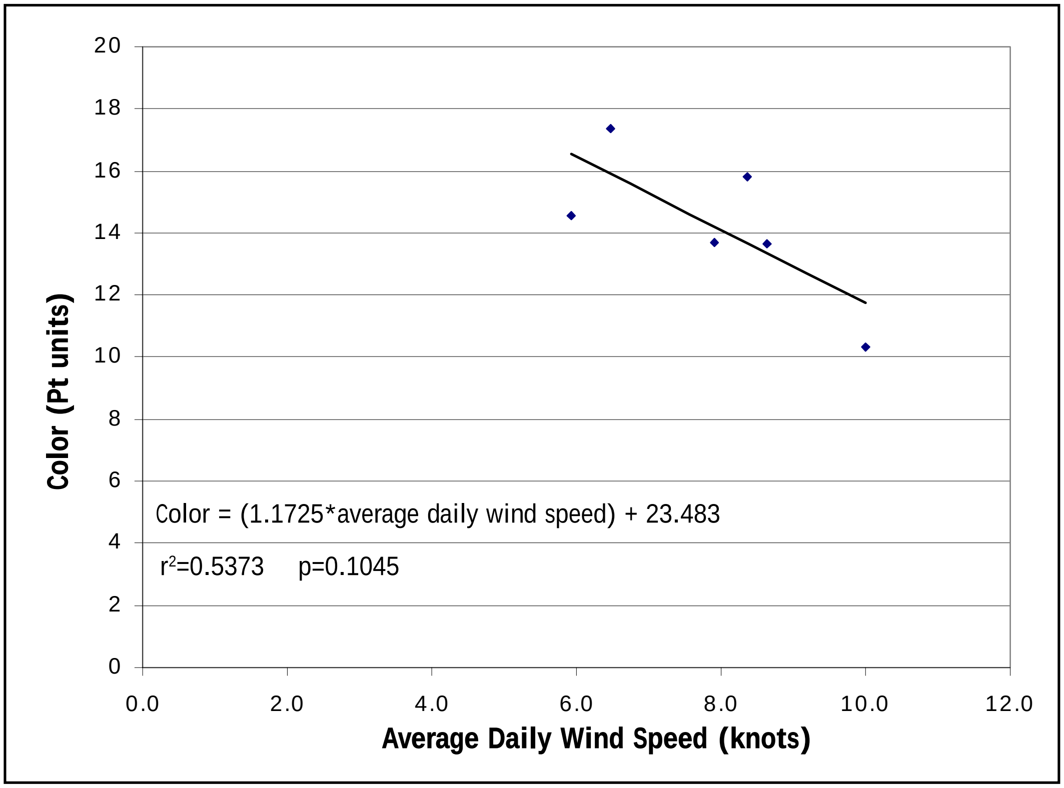

Figure 6a. Color vs. average daily wind speed.

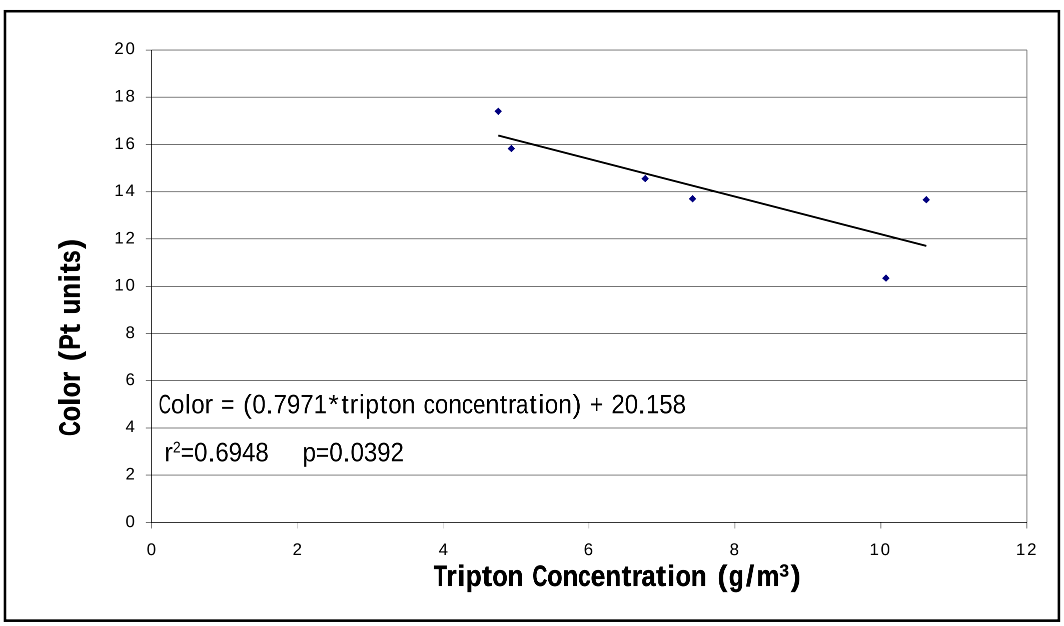

Figure 6b. Color vs. tripton concentration.

have become very windy in the afternoon. Hence daily average wind would not necessarily give the best picture of what was happening during sampling.

The importance of tripton to light attenuation is also seen in Figures 4a and 4d. Figure 4a shows an increase in tripton concentration at the higher numbered sites in the north. Figure 4d shows a similar increase in K_d(PAR) at the same sites.

The high percentage of K_d(PAR) due to tripton indicates how important a role suspended particles would play in any light attenuation model for the IRL. Tripton makes up a large portion of TSS which is used as a input for the numerical model used here. Therefore, the tripton results presented here are also very relevant to the modeling. Turbidity must be calculated from the TSS concentrations for use in the model. Gallegos(1993) calibrated his model originally for turbidity because it provided a better fit for the data than TSS. Since turbidity would take into account sediment size, future work should concentrate on creating better relationships for TSS and turbidity, taking into account different sediment sizes.

CONCLUSIONS

Based on the analysis of data collected during April to June 1997, we can conclude that tripton concentration, color, and chlorophyll *a* concentration all vary both temporally and spatially in the Indian River Lagoon. Of the three, tripton has the greatest influence on the light attenuation coefficient. Overall, tripton accounted for 60.5% of K_d(PAR), while color and chlorophyll *a* account for 25.2% and 9.6% respectively. If K_d(PAR) is below 1 m^{-1}, color and chlorophyll *a* have a relatively greater influence on K_d(PAR), accounting for 30.1% and 10.8%, respectively, while tripton accounts for 53.5%. At K_d(PAR) greater than 1 m^{-1} however, tripton accounts for 75.4% of K_d(PAR), while color and chlorophyll *a* each account for only 14.7% and 7.1% respectively. This dominance of tripton may explain why the lagoon wide averages for each sampling trip showed a decrease in color at higher K_d(PAR)s. The tripton concentration increased at the same time and could be increasing the K_d(PAR), even though color, another constituent of light attenuation, is lower.

Though the relationship is not significant at the 0.05 level, tripton concentration showed a positive correlation with daily average wind speed, even though the winds were averaged and direction of the winds were not taken into account. This suggests the role of wind-induced sediment resuspension. An increase in the number of data points is needed to better show this relationship.

The significant effect of tripton on light attenuation has major implications on management strategy. A few strategies are suggested here to reduce the tripton related light attenuation. First of all, one may want to reduce the loading of muck into the IRL. The next step is to identify areas with high wave-induced resuspension of sediments, and consider removal or capping of fine sediments. Last but not least, one may want to quantify the sediment resuspension due to boating activities and develop a strategy to minimize boat-induced resuspension.

A numerical light model capable of estimating K_d(PAR) in the Indian River Lagoon has been developed. Our results suggest that the effect of sediment size on light attenuation should be further investigated.

ACKNOWLEDGMENTS

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DECADAL CHANGES IN SEAGRASS DISTRIBUTION AND ABUNDANCE IN FLORIDA BAY

M.O. Hall, M.J. Durako, J.W. Fourqurean, J.C. Zieman

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ABSTRACT

The Florida Bay ecosystem has changed substantially in the past decade, and alterations in the seagrass communities have been particularly conspicuous. In 1987, large areas of *Thalassia testudinum* (turtlegrass) began dying rapidly in western Florida Bay. Although the rate has slowed considerably, die-off continues in many parts of the bay. Since 1991, seagrasses in Florida Bay have been subjected to decreased light availability due to widespread, persistent microalgal blooms and resuspended sediments. In light of these recent impacts, we determined the current status of Florida Bay seagrass communities. During the summer of 1994, seagrass species composition, shoot density, shoot morphometrics and standing crop were measured at 107 stations. Seagrasses had been quantified at these same stations 10 yr earlier by Zieman et al. (1989). *T. testudinum* was the most widespread and abundant seagrass species in Florida Bay in 1984 and 1994, and turtlegrass distribution changed little over the decade. On a baywide basis, *T. testudinum* density and biomass declined significantly between surveys; mean short-shoot density of *T. testudinum* dropped by 22% and standing crop by 28% over the decade. *T. testudinum* decline was not homogeneous throughout Florida Bay; largest reductions in shoot density and biomass were located principally in the central and western bay. Percent loss of *T. testudinum* standing crop in western Florida Bay in 1994 was considerably greater at stations with the highest levels of standing crop in 1984 (126-215 g dry wt m^{-2}) than at stations with lower levels of biomass. While turtlegrass distribution remained consistent over time, both the distribution and abundance of two other seagrasses, *Halodule wrightii* and *Syringodium filiforme* declined substantially between 1984 and 1994. Baywide, *H. wrightii* shoot density and standing crop declined by 92%, and *S. filiforme* density and standing crop declined by 93% and 88%, respectively, between surveys. Patterns of seagrass loss in Florida Bay between 1984 and 1994 suggest die-off and chronic light reductions were the most likely causes for decline. If die-off and persistent water-column turbidity continue in Florida Bay, the long-term future of seagrasses in the bay is uncertain.

INTRODUCTION

Seagrasses are characteristic of shallow coastal waters worldwide, however, few areas contain meadows as extensive as those found in Florida Bay (Iverson and Bittaker 1986, Zieman. et al. 1989). Seagrass beds, dominated by *Thalassia testudinum* Banks ex König (turtlegrass), historically covered most of the subtidal mud banks and basins in Florida Bay (Zieman et al. 1989). South Florida seagrasses provide food and/or shelter to numerous fish and invertebrate species in the region, including the economically important pink shrimp, stone crab and spiny lobster (Davis and Dodrill 1989, Holmquist et al. 1989a, Thayer and Chester 1989, Tilmant 1989, Robblee et al. 1991). A variety of wading birds as well as endangered species such as bald eagles, manatees, crocodiles and sea turtles also depend, in part, on seagrass communities (Holmquist et al. 1989b, Mazzotti 1989, Boesch et al. 1993). Declines in seagrass habitat, or changes in the species composition of seagrasses within Florida Bay could have serious consequences for the economy and ecology of South Florida (Robblee et al. 1991, Thayer et al. 1994, Thayer et al. 1999).

During the past decade, many components of the Florida Bay ecosystem have changed substantially, and alterations in the seagrass communities have been particularly conspicuous (Boesch et al. 1993, Butler et al. 1993, Fourqurean and Robblee 1999). Extensive areas of *Thalassia testudinum* began dying rapidly during the summer of 1987, particularly in western

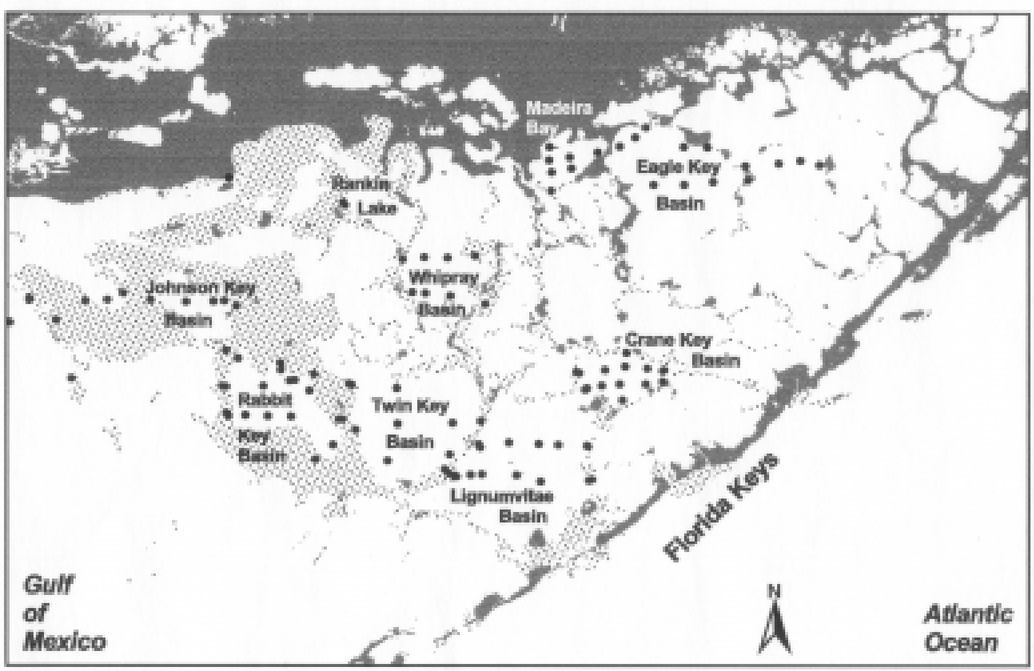

Figure 1. Florida Bay showing the locations of the 107 study sites. Areas with coarse stippling represent carbonate mud banks. Finely stippled areas respresent islands and the mainland.

Florida Bay (Robblee et al. 1991). Although the rate has slowed considerably, turtlegrass die-off continues in many parts of the bay. The patchy mortality characteristic of die-off is very different from the gradual thinning and loss of seagrasses due to increased water-column turbidity experienced in many other parts of the world. In *T. testudinum* meadows affected by die-off, there is often a sharp transition between die-off patches and visually healthy seagrasses. Factors that may contribute to *T. testudinum* die-off are physiological stressors such as elevated water temperature and prolonged hypersalinity, excessive seagrass biomass leading to increased respiratory demands, hypoxia and sulfide toxicity, and disease. The causative mechanism behind die-off remains incompletely understood (Robblee et al. 1991, Carlson et al. 1994, Durako 1994, Durako and Kuss 1994, Fourqurean and Robblee 1999).

While die-off appears to affect only turtlegrass, an ecosystem change relevant to all Florida Bay seagrass species is the widespread decline in water clarity that began in 1991 (Boyer et al. 1999, Stumpf et al.1999). This increased light attenuation is due principally to microalgal blooms and resuspended sediments, and is most severe in the western and central bay (Phlips et al. 1995, Phlips and Badylak 1996). Environmental changes that lead to reductions in available light have been implicated in seagrass declines worldwide (Peres and Picard 1975, Cambridge and McComb 1984, Orth and Moore 1984, Giesen et al. 1990, Dennison et al. 1993, Onuf 1994). Thus, the turbid conditions that have become apparent in many parts of Florida Bay over the past few years may also negatively affect seagrasses (Thayer et al. 1994, Phlips et al. 1995, Fourqurean and Robblee 1999).

Because seagrasses have been subject to several major ecosystem changes in the recent past, we determined the current status of Florida Bay seagrass communities. During the summer of 1994, seagrass species composition, shoot density, shoot morphometrics, and standing crop were measured at more than 100 stations in Florida Bay. Seagrasses had been quantified at these same stations 10 yr earlier by Zieman et al. (1989) to establish a baseline for long-term monitoring of Florida Bay macrophyte communities. Revisiting the stations originally sampled in 1984 enabled us to update information regarding the distribution and abundance of seagrass communities in Florida Bay, and provided us with the unique opportunity to assess changes that have occurred since the onset of turtlegrass die-off and persistent water-column turbidity.

MATERIALS AND METHODS

Seagrasses

Seagrasses were quantitatively sampled during June 1994 at 107 stations distributed throughout most of Florida Bay (Fig. 1; see Fourqureanand Robblee 1999 for a detailed description of Florida Bay). As previously stated, seagrasses at these stations were sampled by Zieman et al. (1989) in the summer of 1984. Four seagrass cores (15 cm diameter) were obtained at each station by haphazardly choosing sampling locations several meters off the bow, stern, and port and starboard of the boat. Cores were washed free of sediment in the field, stored in plastic bags, and frozen for subsequent analysis. In the laboratory, seagrasses were sorted by species, and short-shoot density and standing crop were determined from the material in each core. Plant material for standing crop (leaf dry weight m^{-2}) estimates was washed in 10% HCl, dried at 60°C, and weighed. The number of blades per shoot, and blade lengths and widths were determined for *Thalassia testudinum* shoots. Sediment depth was measured at each station with a stainless steel probing rod, and water depth was measured with a PVC pole marked in 1-cm increments.

Statistical Analyses

Paired *t*-tests were used to assess baywide differences in shoot density, blade lengths, number of blades per shoot, and standing crop between summer 1984 and 1994. Wilcoxon-Signed-Rank analyses were used in lieu of *t*-tests when data could not be transformed to meet assumptions of normality. Stations were initially divided into three depth categories (0.25–1.25 m, >1.25– 2.25 m, and >2.25 m) to assess differences in seagrass parameters with respect to water depth in 1984 and 1994. One-way ANOVA revealed that the latter two categories did not differ significantly from each other, and stations with depths >1.25 m were combined; *t*-tests were used to assess differences in seagrass parameters with respect to the two water depth categories ($\leq$1.25 m versus >1.25 m) for 1984 and 1994. Mann-Whitney Rank Sum analyses were used in lieu of *t*-tests when data could not be transformed to meet assumptions of normality and homogeneity of variance. Two-way ANOVA was used to investigate possible date × depth interactions. The effect of water depth on percent change in *Thalassia testudinum* standing crop between 1984 and 1994 was assessed by Mann-Whitney Rank Sum analysis. Relationships between water depth and sediment depth, and seagrass density and standing crop in 1984 and 1994 were explored by linear regression analysis.

Zieman et al. (1989) divided Florida Bay into six regions based principally on the distribution and abundance of macrophyte communities (Fig. 2; see Zieman et al. 1989 for a detailed description of these regions). Paired *t*-tests and Wilcoxon-Signed-Rank analyses were used to assess differences in shoot density, blade lengths, and number of blades per shoot and standing crop between summer 1984 and 1994 within each region. No attempt was made to address differences in seagrass parameters among regions due to the substantial variation in number of stations within regions (range = 8–37).

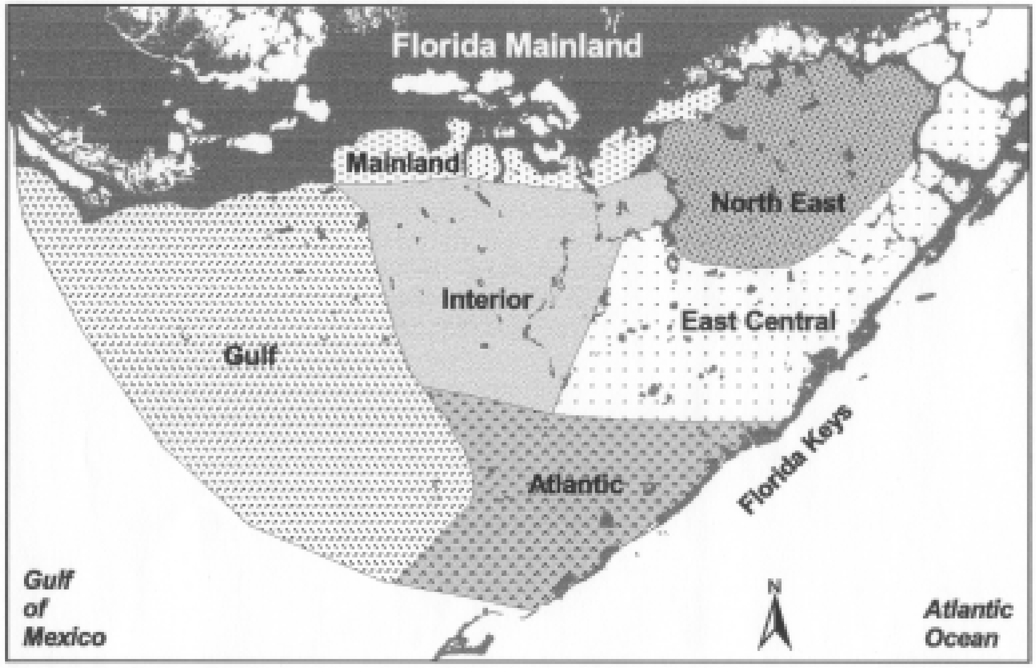

Figure 2. Location and extent of ecological regions within Florida Bay based on macrophyte distribution and abundance (after Zieman et al. 1989).

Initial turtlegrass die-off was reported to occur primarily in the densest seagrass beds in western Florida Bay (Robblee et al. 1991). Stations in western Florida Bay were divided into three density categories based on the 1984 standing crop (low biomass: 10–50 g dry wt m^{-2}, n=19; medium biomass: 51–125 g dry wt m^{-2}, n=26; and high biomass: 126–215 g dry wt m^{-2}, n=20). Kruskal-Wallis one-way ANOVA on ranks was used to assess differences in percent biomass change among the three standing crop categories measured in 1994. Dunn's multiple comparison procedure was used to isolate treatments where significant differences occurred. Stations with mean standing crops <10 g m^{-2} in 1984 were not included in the analysis due to extreme variation in percent change.

RESULTS

Baywide Changes in Seagrass Communities

Thalassia testudinum

Thalassia testudinum was the most widespread and abundant seagrass species in Florida Bay during both 1984 and 1994 (Table 1). Turtlegrass distribution changed little over the decade; *T. testudinum* was present at 102 stations in 1984 and at 99 stations in 1994. Geographic patterns of *T. testudinum* abundance (i.e. short-shoot densities and leaf standing crop) were also similar between surveys. Shoot densities were generally lowest in northeastern Florida Bay, and increased towards the southwest, where highest densities were observed (Figs. 3a and 3b). Shoot density was positively associated with standing crop (1984: $r^2=0.62$, $p<0.001$; 1994: $r^2=0.74$, $p<0.001$), thus geographic patterns in turtlegrass standing crop followed those for density (Figs. 4a and 4b). Geographic variation in *T. testudinum* abundance in Florida Bay corresponds to a gradient of increasing sediment depth from northeast to southwest (see Zieman et al. 1989); turtlegrass shoot density (1984: $r^2=0.35$, $p<0.001$; 1994:$r^2= 0.30$, $p<0.001$) and standing crop (1984: $r^2=0.34$, $p<0.001$; 1994: $r^2=0.30$, $p<0.001$) were positively related to sediment depth in both years (Fig. 5). Relationships between turtlegrass

Table 1. Percent occurrence, short-shoot densities and standing crop of seagrasses in Florida Bay in 1984 and 1994.

	SPECIES		
	Thalassia	*Halodule*	*Syringodium*
% Occurrence (n=107)			
1984	95.3	47.7	11.2
1994	92.5	13.1	3.7
Baywide Standing Crop (g dry wt m^{-2} ± SE)			
1984	63.9 ± 5.5	3.8 ± 1.1	4.1 ± 1.7
1994	46.0 ± 4.0	0.3 ± 0.2	0.5 ± 0.3
Standing Crop Where Present (g dry wt m^{-2} ± SE)			
1984	67.1 ± 5.6	7.5 ± 2.0	36.8 ± 10.8
1994	48.2 ± 4.1	0.6 ± 0.4	5.6 ± 3.7
Baywide Shoot Density (# m^{-2} ± SE)			
1984	694.7 ± 47.5	267.5 ± 80.0	83.3 ± 34.9
1994	539.3 ± 49.5	22.5 ± 11.0	5.6 ± 3.8
Shoot Density Where Present (# m^{-2} ± SE)			
1984	720.5 ± 47.8	550.1 ± 156.0	742.4 ± 245.3
1994	565.7 ± 50.5	42.5 ± 22.1	49.5 ± 31.8

abundance and sediment depth observed in 1984 closely resembled those observed in 1994.

The ranges in mean shoot density and standing crop remained similar at the 107 stations over the decade (1984: 0–2133 shoots m^{-2} and 1994: 0–2137 shoots m^{-2}; 1984: 0–215 g dry wt m^{-2} and 1994: 0–185 g dry wt m^{-2}). However, abundance was lower at many of the stations in 1994 than in 1984 (Figs. 3c and 4c), and on a baywide basis, *Thalassia testudinum* density and biomass declined significantly between surveys. Mean short-shoot density of *T. testudinum* in Florida Bay dropped by 22% (694.7±47.5 to 539.3±49.5 shoots m^{-2}, $p<0.001$), and standing crop by 28% over the decade (63.9±5.5 to 46.0±4.0 g dry wt m^{-2}, $p<0.001$) (Table 1). When considering only the 102 sites where turtlegrass occurred, mean density fell from 720.5±47.8 to 565.7±50.5 shoots m^{-2} ($p<0.001$) and standing crop from 67.1±5.6 to 48.2±4.1 g dry wt m^{-2} ($p<0.001$). *T. testudinum* decline was not homogeneous throughout the bay; most of the stations with the largest reductions in shoot density and biomass were located in central and western Florida Bay (Figs. 3c and 4c). Although there was a significant decline baywide, turtlegrass abundance actually increased at a number of stations from 1984 to 1994, especially in the eastern bay.

The mean number of turtlegrass blades per shoot did not change in Florida Bay during the decade (2.6±0.8 in 1984 versus 2.6±0.7 in 1994, p= 0.269; range at 107 stations 1.9–4.1 in 1984, and 1.7–5.2 in 1994). The range in blade length was also consistent over time (3.2–31.7 cm in 1984, and 3.9–31.0 cm in 1994); however, on a baywide basis, mean blade length declined by 28% between surveys (12.1±0.8 to 8.7±0.4cm, $p<0.001$).

Mean turtlegrass density was significantly higher in shallow (1.25 m, n=47) versus “deep” (>1.25 m, n=60) waters of Florida Bay in 1984 (932.0±70.5 versus 508.9±53.4 shoots m^{-2}, p= 0.001) and 1994 (779.2±88.5

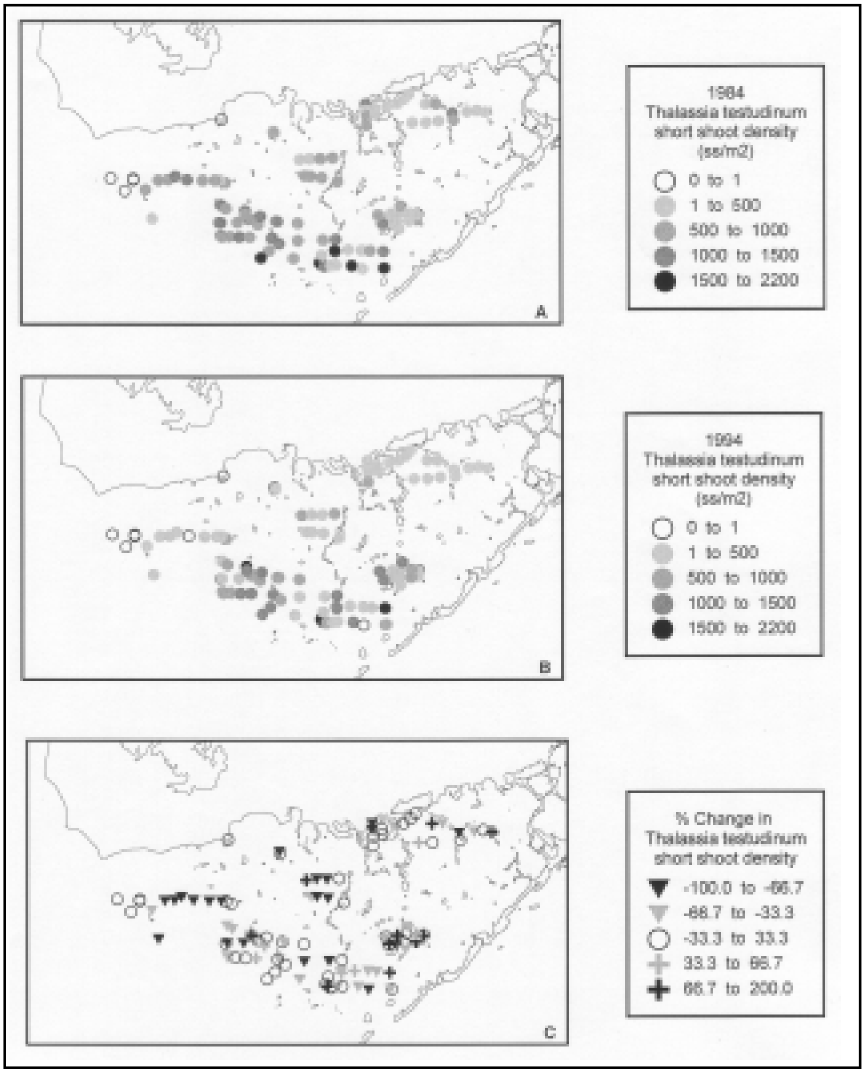

Figure 3. Short-shoot densities (#m^{-2}) of *Thalassia testudinum* in 1984 (A) and 1994 (B) at the study locations within Florida Bay. Percent change in *T. testudinum* shoot densities at the study locations (C).

versus 351.4±41.3 shoots m^{-2}, p=0.019). Results were similar for leaf standing crop (shallow versus deep: 85.8±8.1 versus 46.8±6.8 g dry wt m^{-2}, p<0.001 in 1984, and 63.0±6.6 versus 32.7±4.2 g dry wt m^{-2}, p<0.001 in 1994). Although *T. testudinum* abundance varied significantly with both date (1984 versus1994) and water depth (shallow versus deep), results of two-way ANOVA revealed no significant date × depth interaction for shoot densities (p= 0.734) or standing crop (p= 0.494). Neither *T. testudinum* blade length (shallow versus deep: 12.5±1.0 versus 10.8±0.9 cm, p= 0.208 in 1984, and 8.6±0.5 versus 8.9±0.7cm, p=0.73 in 1994), nor number of blades per shoot (shallow versus deep: 2.7±0.1 versus 2.6±0.1, p=0.421 in 1984, and 2.6±0.1 versus 2.6±0.1, p= 0.972 in 1994) varied significantly with water depth.

Percent change in *Thalassia testudinum* standing crop in western Florida Bay in 1994 varied significantly with regard to levels of leaf biomass in 1984 (p = 0.004; Fig. 6). Mean biomass reduction in 1994 was considerably greater at stations with high levels of standing crop in 1984 (52% decline) than at stations with medium

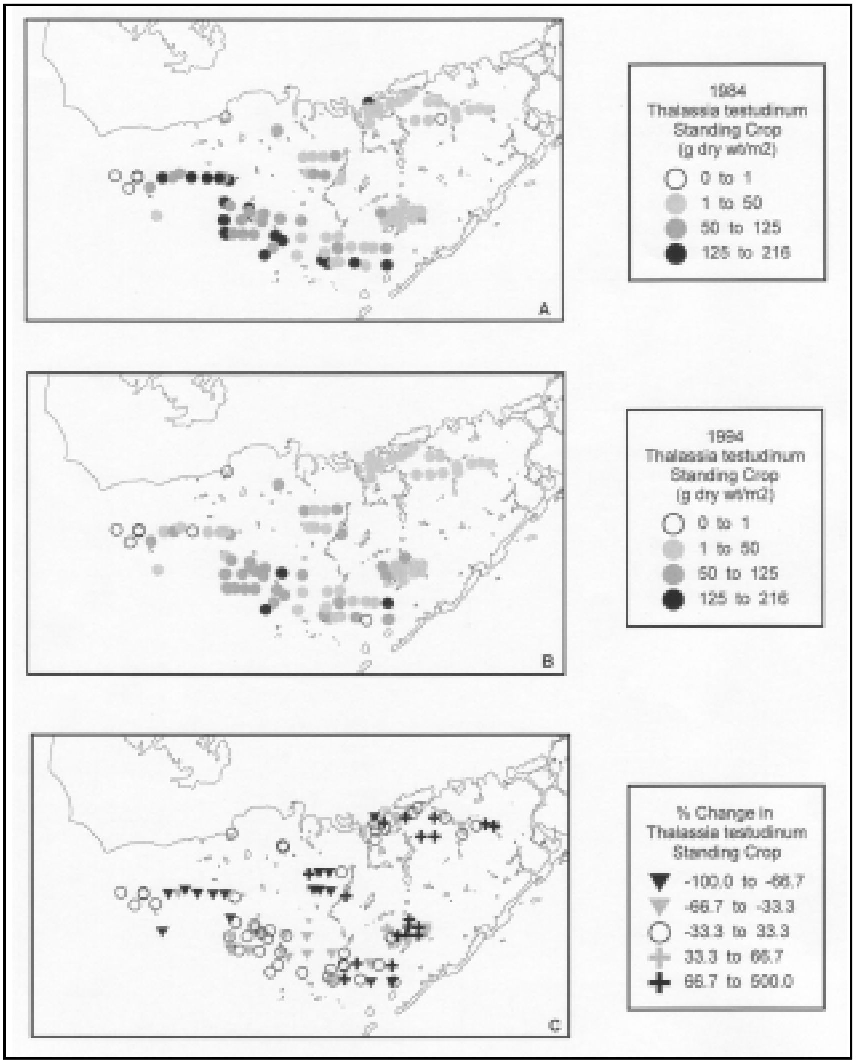

Figure 4. Standing crop (g dry wt m^{-2}) of *Thalassia testudinum* in 1984 (A) and 1994 (B) at the study locations within Florida Bay. Percent change in *T. testudinum* standing crop at the study locations (C).

levels of standing crop in 1984 (12% decline). Leaf biomass at stations in the low biomass category in 1984 remained almost unchanged over the decade (0.5% increase).

Halodule wrightii Ascherson

Halodule wrightii (shoalgrass) was less widespread than *Thalassia testudinum* in Florida Bay in 1984, occurring at only 48% of the stations surveyed (Table 1). Short-shoot densities generally increased from eastern to western Florida Bay, and maximum *H. wrightii* abundance was reached in the northwestern bay in both 1984 and 1994 (Figs. 7a and 7b). While turtlegrass distribution remained consistent over time, shoalgrass distribution declined substantially between surveys. In 1994, *H. wrightii* was found at only 12% of the sampling locations, and at considerably reduced abundances (Fig. 7c). Baywide, mean short-shoot density dropped from 267.5±80.0 to 22.5±11.0 shoots m^{-2} between surveys ($p<0.001$; range = 0–6400 shoots m^{-2} in 1984 , and 0–962.2 shoots m^{-2} in 1994). Patterns for standing crop closely followed those for density: shoalgrass leaf biomass declined from 3.8±1.1 g dry wt m^{-2} to 0.3±0.2 g dry wt

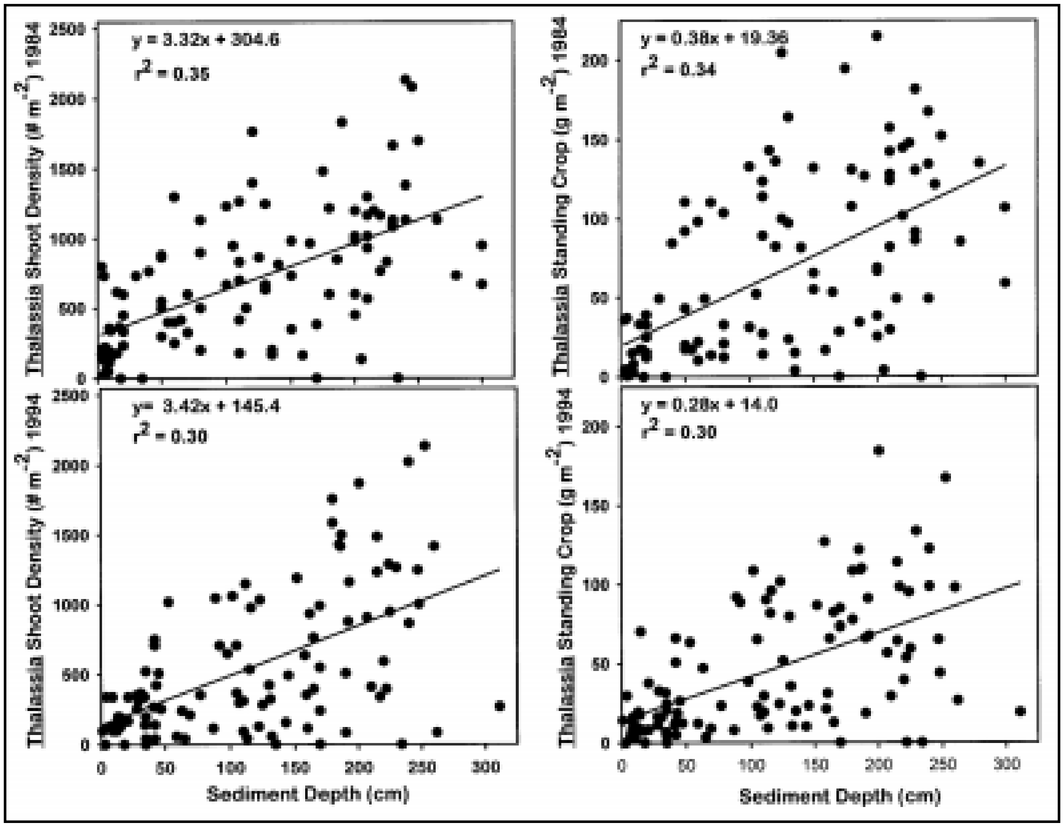

Figure 5. Linear regressions showing the relationship between *Thalassia testudinum* short-shoot density (#m^{-2}) and sediment depth (cm) in the study locations in 1984 and 1994, and between *Thalassia testudinum* standing crop (g dry wt m^{-2}) and sediment depth (cm) in 1984 and 1994.

m^{-2} from 1984 to 1994 ($p<0.001$; range = 0–87.5 g dry wt m^{-2} in 1984, 0–21.3 g dry wt m^{-2} in 1994). When considering only the 52 sites where shoalgrass occurred, mean density fell from 550.1±156.0 shoots m^{-2} to 42.5±22.1 shoots m^{-2} ($p<0.001$), and standing crop from 7.5±2.0 g dry wt m^{-2} to 0.6±0.4 g dry wt m^{-2} ($p<0.001$) over the decade.

Syringodium filiforme Kutzing

The distribution of *Syringodium filiforme* (manatee grass) was the most limited of the three seagrass species in Florida Bay. In summer 1984 *S. filiforme* was present at only 12 stations in western Florida Bay (Table 1; Fig. 8a). A decade later, *S. filiforme* was restricted to four stations, and at considerably reduced abundances (Figs. 8b and 8c). Mean short-shoot density fell 93% between surveys, from 83.3±34.9 shoots m^{-2} to 5.6±3.8 shoots m^{-2} ($p<0.001$, range = 0–2366.7 shoots m^{-2} in 1984 , and 0–382.1 shoots m^{-2} in 1994). Results were similar for standing crop, which declined from 4.1±1.7 g dry wt m^{-2} in 1984 to 0.5±0.3 g dry wt m^{-2} in 1994, ($p = 0.003$, range = 0–99.3 g dry wt m^{-2} in 1984, 0–19.2 g dry wt m^{-2} in 1994), an 88% reduction. When considering only the stations where manatee grass occurred, mean density fell from 742.4±245.3 shoots m^{-2} in 1984 to 49.6±31.8 shoots m^{-2} in 1994 ($p = 0.001$), and standing crop from 36.7±11.3 g dry wt m^{-2} in 1984 to 4.3±2.1 g dry wt m^{-2} in 1994 ($p = 0.017$).

Regional Changes in Seagrass Communities

Thalassia testudinum

Regional analyses again illustrated an increase in turtlegrass abundance from northeastern to southwestern Florida Bay (Figs. 9a, 9b). Mean shoot densities of

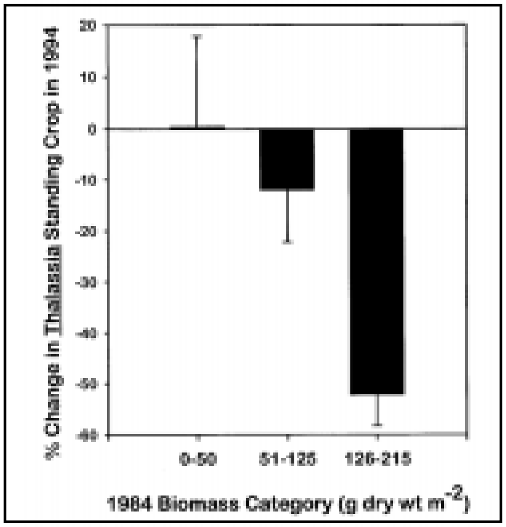

Figure 6. Percent change in *Thalassia testudinum* standing crop in 1994 with respect to 1984 biomass category.

Thalassia testudinum generally declined in all regions except the East Central region between 1984 and 1994 (Fig. 9a). Densities became significantly lower in the Mainland ($p = 0.047$, 474.0±151.1 shoots m^{-2} in 1984 versus 206.4±63.6 shoots m^{-2} in 1994), Interior ($p = 0.010$, 714.1±80.8 shoots m^{-2} in 1984 versus 422.3±96.9 shoots m^{-2} in 1994) and Gulf ($p= 0.001$, 809.0±66.8 shoots m^{-2} in 1984 versus 560.0±82.0 shoots m^{-2} in 1994) regions between surveys. Results for standing crop resembled those for density (Fig. 9b); however, declines in biomass over the decade were significant only in the Gulf region ($p<0.001$, 101.1±9.0 g dry wt m^{-2} in 1984 versus 58.7±6.6 g dry wt m^{-2} in 1994). Discrepancies in the amount of reduction in density versus biomass within particular regions appeared to reflect corresponding changes in blade length between surveys (Fig. 9c). Mean blade length fell by 32% in the Gulf region between 1984 and 1994 (18.2±1.2 versus 12.5±0.8 cm, $p<0.001$), but by only 18% and 19.5%, respectively, in the Interior (11.7±1.0 cm versus 9.6±0.8 cm, $p = 0.007$) and Mainland regions (9.7±1.4 cm versus 7.8±0.5 cm, $p = 0.222$). Blade lengths also became significantly shorter in the Atlantic region over the decade ($p<0.001$, 11.8±0.8 cm versus 7.9±0.5 cm). Number of blades per shoot did not differ between surveys in any region of Florida Bay (Fig. 9d).

Halodule wrightii

Halodule wrightii was present in all regions of Florida Bay in 1984, but was abundant only in the Mainland, Interior, and Gulf regions. Shoalgrass abundance generally became lower in all regions from 1984 to 1994 (Fig. 10). As with turtlegrass, *H. wrightii* shoot densities declined significantly in the Gulf region between surveys ($p<0.001$, 350.2±110.2 shoots m^{-2} in 1984 versus 7.7±4.2 shoots m^{-2} in 1994). Patterns were similar for shoalgrass standing crop, which decreased by more than 97% in the Gulf region over the decade ($p<0.001$, 5.8±1.6 g dry wt m^{-2} in1984 versus 0.2±0.1 g dry wt m^{-2} in 1994). Standing crop also decreased significantly from 1984 to 1994 in the Northeast region ($p = 0.016$, 0.63±0.3 g dry wt m^{-2} in 1984 versus 0.04±0.04 g dry wt m^{-2} 1994).

Syringodium filiforme

Syringodium filiforme was present only in deeper waters of the Gulf region in both 1984 and 1994 (Fig. 11). The abundance of manatee grass in this region of Florida Bay decreased significantly over the decade. *S. filiforme* density declined from 240.8±96.5 shoots m^{-2} in 1984 to 16.1±10.7 shoots m^{-2} in 1994($p<0.003$), and standing crop from 11.9±4.6 dry wt m^{-2} in 1984 to 1.4±0.8 g dry wt m^{-2} in 1994($p = 0.003$).

DISCUSSION

Although the Florida Bay ecosystem changed considerably between 1984 and 1994, *Thalassia testudinum* continued to be the dominant seagrass species in the bay, and its distribution at the study locations remained almost unchanged between surveys. Spatial patterns of turtlegrass abundance did not change over the decade; shoot densities and standing crop continued to increase from the northeastern to

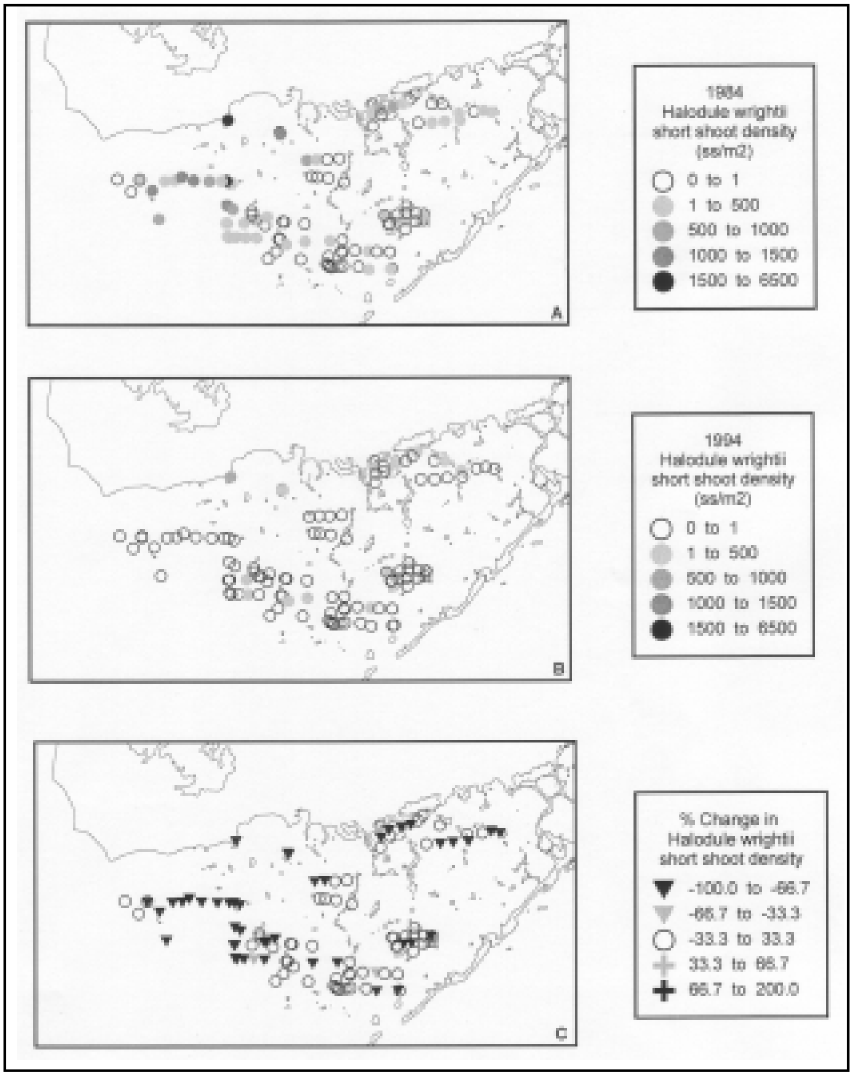

Figure 7. Short-shoot densities (#m^{-2}) of *Halodule wrightii* in 1984 (A) and 1994 (B) at the study locations within Florida Bay. Percent change in *H. wrightii* shoot densities at the study locations (C).

southwestern bay following gradients in sediment depth (Zieman et al. 1989) and phosphorus availability (Fourqurean et al. 1992). *T. testudinum* shoot density and biomass fell significantly over the decade, especially in western and central Florida Bay.

A variety of factors may have influenced turtlegrass decline in Florida Bay between 1984 and 1994, but the most likely causes for loss were *Thalassia testudinum* die-off (most extensive from 1987 to 1990), and decreased water clarity due to sediment resuspension and phytoplankton blooms (1991–1997). Establishing the relative contribution of die-off versus reduced light availability to the turtlegrass decline observed here is problematic because: 1) die-off was patchily distributed, thus there was no way to accurately identify stations where die-off actually occurred; and 2) long-term data regarding light conditions in Florida Bay are scarce since water clarity was of little concern prior to 1991. Although it is not possible to definitively establish cause and effect relationships, examining observed patterns of seagrass loss in Florida Bay relative to expected patterns of decline from turtlegrass die-off

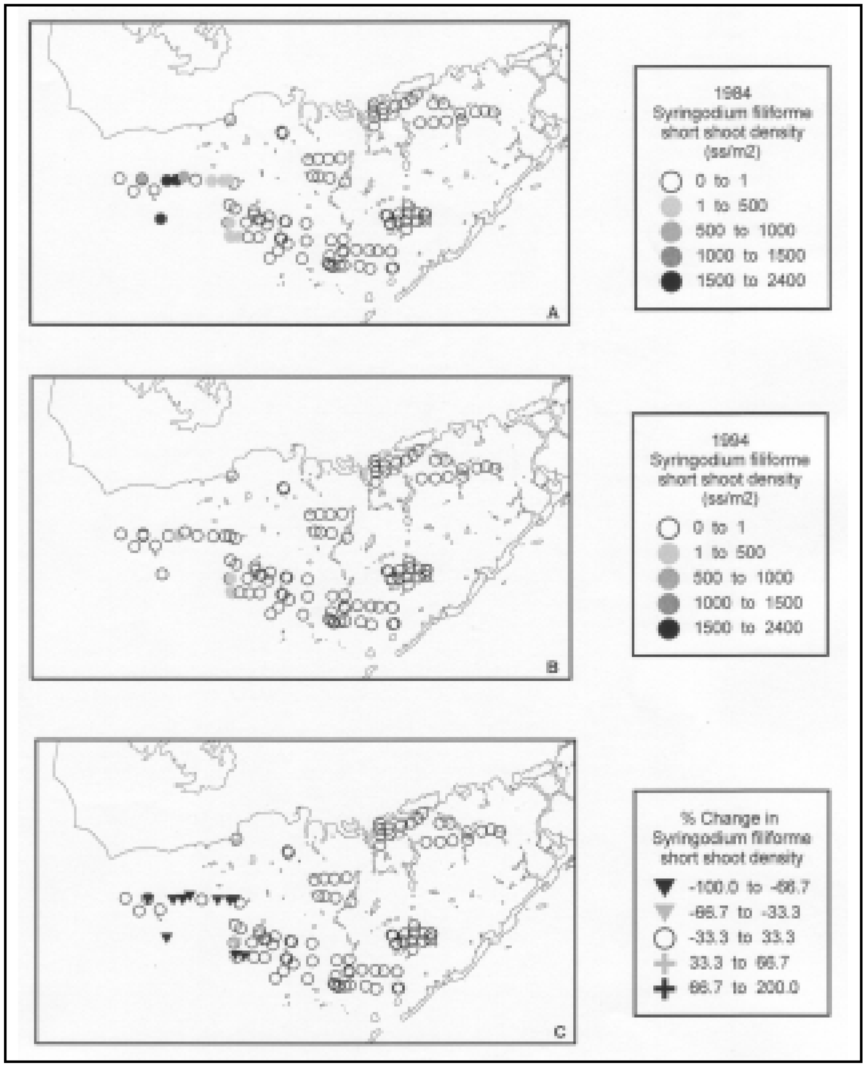

Figure 7. Short-shoot densities (#m^{-2}) of *Syringodium filiforme* in 1984 (A) and 1994 (B) at the study locations within Florida Bay. Percent change in *S. filiforme* shoot densities at the study locations (C).

(see Robblee et al. 1991, Durako et al. 1994), and decreased light availability (see Peres and Picard 1975, Cambridge and McComb 1984, Orth and Moore 1984, Giesen et al. 1990, Dennison et al. 1993, Onuf 1994) lends insight. If the majority of seagrass deterioration in Florida Bay between 1984 and 1994 was due to die-off, one would expect to see: 1) declines in *T. testudinum* primarily in western and central Florida Bay; 2) greatest mortality in very dense turtlegrass beds, located mainly in shallower water along basin margins; 3) patchy rather than widespread declines in *T. testudinum* abundance in western and central Florida Bay; and 4) no decline in *Halodule wrightii* or *Syringodium filiforme*, which apparently were not affected by the die-off. If reduced light availability were responsible for turtlegrass decline, then one would expect to see: 1) *T. testudinum* declines in regions that were unaffected by die-off; 2) widespread reductions in turtlegrass density and biomass rather than localized declines; and 3) larger *T. testudinum* declines in deeper water due to increased light attenuation with depth. Although *T. testudinum* may require higher levels of irradiance for survival than *H. wrightii* or *S. filiforme* (Fourqurean 1992),

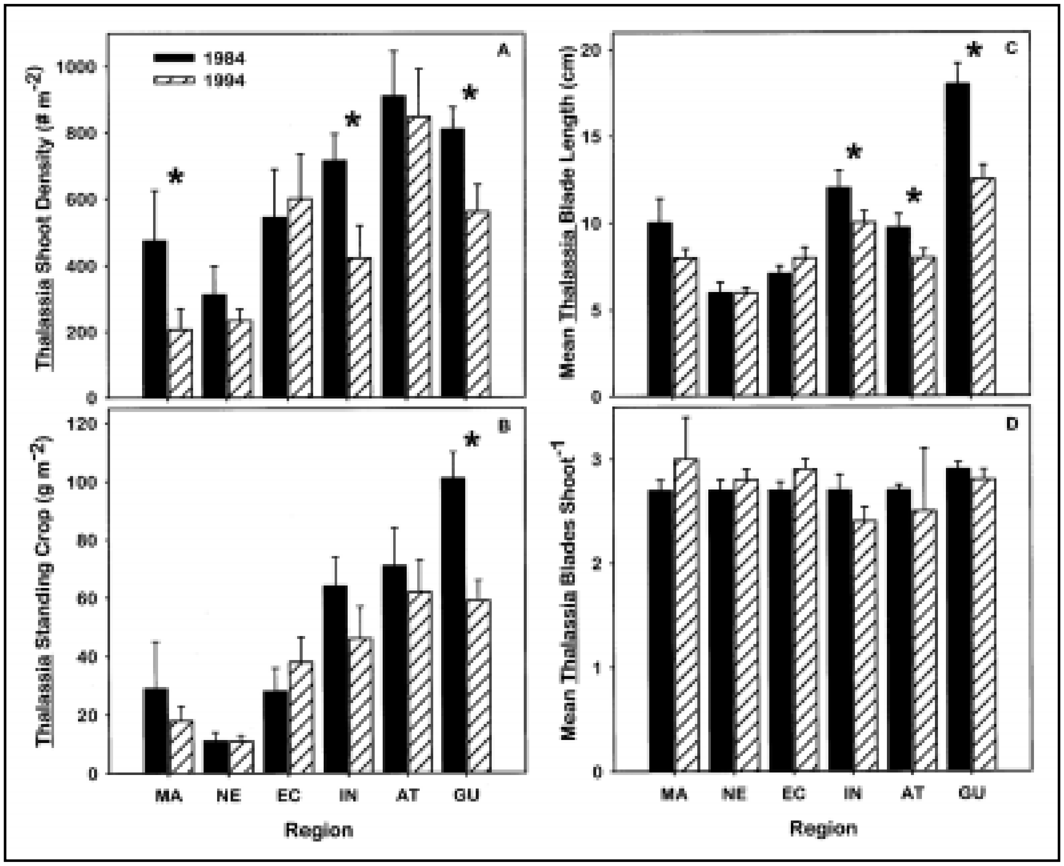

Figure 9. *Thalassia testudinum* short-shoot densities (A), standing crop (B), blade lengths (C), and blades per shoot (D) in 1984 and 1994 within the six ecological regions of Florida Bay.

given sufficient light reduction, one would also expect to see: 4) deterioration of *H. wrightii* and *S. filiforme* as well as *T. testudinum*; and 5) more severe losses in *H. wrightii* and *S. filiforme* than in *T. testudinum* because of the smaller below-ground reserves in the former two species (Hall et al. 1991, Czerny and Dunton 1995). Patterns of seagrass loss in Florida Bay between 1984 and 1994 were partially consistent not only with predictions associated with die-off, but also with those expected from chronic light reduction.

Thalassia testudinum shoot density and standing crop decreased substantially in western and central Florida Bay in areas where both die-off (Robblee et al. 1991) and light reduction (Phlips et al. 1995, Boyer et al. 1999, Stumpf et al. 1999) occurred between 1984 and 1994. However, large turtlegrass losses also occurred in locations with no reported die-off, but where water clarity had deteriorated since 1991. These results concur with those of Zieman et al. (1999), who found *T. testudinum* abundance declined at both die-off and control (no die-off) stations in the western bay from 1989 to 1995. In addition, the distribution and abundance of *Halodule wrightii* and *Syringodium filiforme* declined drastically during the 10-yr period. Such declines in seagrass abundance are commonly observed as the result of decreased light availability in estuaries worldwide (Peres and Picard 1975, Cambridge and McComb 1984, Orth and Moore 1984, Giesen et al. 1990, Dennison et al. 1993, Onuf 1994). Along with substantial reductions in shoot density and biomass, *T. testudinum* blades became shorter in Florida Bay over the decade. *T.*

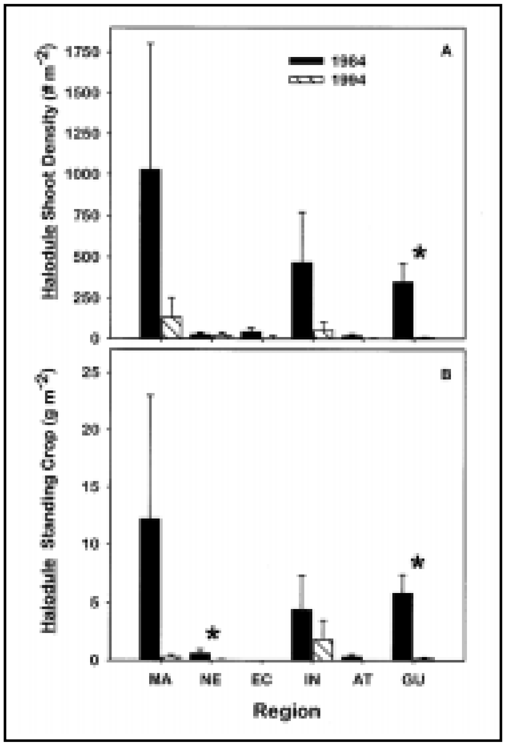

Figure 10. *Halodule wrightii* short-shoot densities (A) and standing crop (B) in 1984 and 1994 within the six ecological regions of Florida Bay.

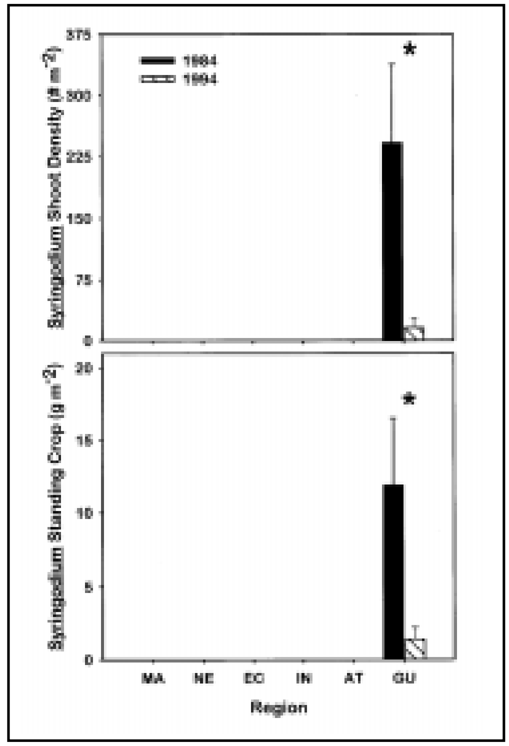

Figure 11. *Syringodium filiforme* short-shoot densities (A) and standing crop (B) in 1984 and 1994 within the six ecological regions of Florida Bay.

testudinum blade lengths have been shown to decline significantly in response to experimental shading (75% reduction of ambient light) in Tampa Bay, Florida (Carlson and Acker 1985). Experimental light reductions of >50% also caused declines in blade lengths of *Posidonia sinuosa* Cambridge and Kuo (Neverauskas 1988, Gordon et al. 1994), a seagrass that is structurally similar to *T. testudinum* (i.e. both seagrass species have relatively large belowground reserves, Czerny and Dunton 1995). However, significant declines in *T. testudinum* blade lengths also occurred in the Atlantic region of Florida Bay, where light levels appeared to have changed very little between 1984 and 1994 (Phlips et al. 1995, Boyer et al. 1999, Stumpf et al. 1999), but where die-off was present. These results suggest that die-off probably played a role in turtlegrass decline in Florida Bay over the decade, but that increased light attenuation was also an important controlling factor.

Our findings regarding *Thalassia testudinum* decline in shallow (1.25m) versus deep (>1.25m) water raises questions concerning the role of light reduction in Florida Bay seagrass loss. Turtlegrass shoot density and standing crop declined significantly in both shallow water and deep water between 1984 and 1994, but the relative amount of decline was not substantially greater in deep than in shallow water. These results would seem inconsistent with reports of seagrass loss due to reduced light in many other estuaries, where declines were generally most severe in the deepest growing seagrasses (Orth and Moore 1983, Cambridge and McComb 1984, Giesen et al. 1990, Dennison et al. 1993, Onuf 1994, Fletcher and Fletcher 1995). Larger and more rapid declines were measured at the deep versus the shallow edge of a *T. testudinum* meadow in Tampa Bay, Florida, in response to experimental shading (60%–65% reduction of ambient light; Hall et al. 1991). In the aforementioned systems, the maximum depth of seagrasses was controlled by light availability, thus any reduction in the amount of light reaching the deepest growing seagrasses should have resulted in declines in density and biomass. This was not the case in western and central Florida Bay, where before 1991, *T. testudinum* growing at approximately 3 m (the deepest stations sampled in these surveys) was not light-limited (see Fourqurean and Zieman 1991). Perhaps *T. testudinum* declines that occurred in Florida Bay between 1984 and 1994 were no greater in deep than in shallow water, because, prior to the turbidity increases that began in 1991, *T. testudinum* meadows in the deeper areas of western and central Florida Bay were well above the light-limited maximum depth. The more pronounced declines in deeper versus shallower water typically seen in response to decreased water clarity may have not yet been evident after only 3 yr due to the high belowground reserves and long life-spans of *T. testudinum* short-shoots. Recruitment and mortality rates of *T. testudinum* short shoots estimated in 1994 using reconstructive aging techniques (see Durako 1994) indicated that density declines should become larger in deeper (> 1m) versus shallower (≤ 1m) water in western Florida Bay (Durako unpublished data). The pattern of equal decline in both shallow and deep sites may be the consequence of two separate causes of decline in *T. testudinum*: an early loss in mostly shallow water (Robblee et al. 1991) due to the poorly understood die-off phenomenon, followed by a subsequent loss in deeper water attributable to chronic light reduction.

The differences in percent change in 1994 *Thalassia testudinum* standing crop among 1984 biomass categories were dramatic. The remarkably large decline in standing crop at stations in the high biomass category was consistent with previous accounts concerning the primary locations affected by die-off in western Florida Bay. The prevalence of die-off in dense *T. testudinum* beds may be related to hypoxia and sulfide toxicity in combination with reduced photosynthetic rates due to blade damage caused by a slime mold (*Labyrinthula* sp.) (Durako and Kuss 1994), which has been isolated from *T. testudinum* in die-off patches (Porter and Muehlstein 1989). Sulfide levels in Florida Bay are quite high relative to other Florida estuaries (Carlson et al. 1994), and seagrasses must translocate photosynthetically produced oxygen to belowground tissues to avoid the effects of hypoxia and sulfide toxicity. *T. testudinum* may be more vulnerable to sulfide-induced hypoxia than *Halodule wrightii* or *Syringodium filiforme* due to its high ratio of belowground to aboveground biomass (Fourqurean and Zieman 1991, Carlson et al. 1994). Blade lesions produced by *Labyrinthula* sp. reduce *T. testudinum* photosynthesis, making it more susceptible to hypoxia and sulfide toxicity, which is generally considered the proximal cause of death in die-off (Durako and Kuss 1994, Fourqurean and Robblee 1999). Infection of *T. testudinum* by *Labyrinthula* sp. in Florida Bay appears to be density dependent, and is quite likely transmitted by leaf-to-leaf contact (see Muehlstein 1992), perhaps explaining the predominance of die-off in dense beds. Environmental stressors that weaken seagrasses (e.g. elevated water temperatures and hypersalinity) may increase the probability of infection by *Labyrinthula* (Short et al. 1988). While these results suggest die-off contributed to the significantly higher percent losses measured in *T. testudinum* beds with high levels of standing crop, the areal extent of *T. testudinum* decline measured in 1994 exceeded regions of reported die-off. Lower photosynthetic rates due to widespread light reduction may have also played a role in the larger declines measured in the dense *T. testudinum* beds of western and central Florida Bay. Reduction in available light by resuspended sediments and algal blooms likely depressed turtlegrass photosynthesis and subsequent translocation of oxygen to the rhizosphere, and could have led to *T. testudinum* decline through increased effects of hypoxia and sulfide toxicity (see Goodman et al. 1995). Dense *T. testudinum* beds might also have been more acutely affected by reduced light than sparser beds as the result of greater self-shading.

The seagrasses *Syringodium filiforme* and *Halodule wrightii* also declined in Florida Bay over the decade, but unlike *Thalassia testudinum, S. filifome* and *H. wrightii* declined at almost every station where they occurred in 1984. Declines in all seagrass species within the bay, and larger declines in *S. filiforme* and *H. wrightii* than in *T. testudinum* suggests decreased water clarity contributed to seagrass losses in Florida Bay between 1984 and 1994. The declines in *S. filiforme* and *H. wrightii* in western and central Florida Bay were probably related to the increased light attenuation that has occurred in these areas since 1991 (Phlips et al. 1995, Boyer et al.1999, Stumpf et al. 1999). Both Robblee et al. (1991) and Thayer et al. (1994) suggested that *H. wrightii* rapidly colonized die-off patches in western Florida Bay. However, during subsequent visits to their study sites in 1993, Thayer et al. (1994) found no seagrasses in previous die-off patches. These authors suggested that the decreased water clarity that began in 1991 limited seagrass recolonization in die-off areas. While declines in *H. wrightii* in western Florida Bay appear to be related to increased light attenuation, shoalgrass abundance also decreased at a number of stations in eastern Florida Bay where the light climate appears to have changed very little during the past decade (Phlips et al. 1995, Boyer et al. 1999, Stumpf et al.1999). Zieman et al. (1989) suggested that long-term reduction of freshwater inflow may have played a role in the decline of shoalgrass in eastern Florida Bay during the past several decades. Higher, more stable salinities may promote *T. testudinum* over *H. wrightii*, especially in areas of clear water. *T. testudinum* abundance actually increased at a number of stations in eastern Florida Bay between 1984 and 1994, supporting the suggestion of Zieman et al. (1989). Reduced phosphorus availability due to less freshwater inflow over many decades might also be involved in the decline of *H. wrightii* in eastern Florida Bay (Powell et al. 1989). It must be noted that we compared two snapshots of Florida Bay seagrass communities taken 10 yr apart. Because *H. wrightii* distribution and abundance can vary substantially over short time periods (Zieman 1982, Thayer et al. 1994, Fourqurean et al. 1995), and *S. filiforme* was encountered at only a few stations, declines in these seagrasses might be attributed to random variation or seasonal fluctuations in abundance. However, the magnitude and extent of *H. wrightii* and *S. filiforme* declines between 1984 and 1994 suggest that seagrass losses were the result of significant environmental change rather than random effects.

The patchy distribution of die-off in Florida Bay coupled with the haphazard placement of stations in 1984 may have influenced our ability to assess the full extent of die-off in the decline of *T. testudinum*. For example, only two of the 107 stations we surveyed were located in Rankin Lake (see Fig. 1), which was one of the basins most severely affected by die-off. Thus, the chance of encountering stations in die-off patches was lower than would be expected with a more widespread controlling factor like reduced light. Even if increased light attenuation rather than die-off was the direct factor leading to *T. testudinum* loss at most stations in the present study, die-off probably played a major role in seagrass decline through its secondary effects on water clarity. *T.*

testudinum die-off in western Florida Bay led to extensive areas of exposed sediments, which apparently caused the widespread turbidity now present in this region (Phlips and Badylak 1996, Stumpf et al. 1999). In addition, nutrients released to the water column through the remineralization of dying seagrasses and sediment resuspension in central Florida Bay may have contributed to algal bloom development. Die-off and persistent water column turbidity continue to affect seagrasses in Florida Bay, and recent observations (Durako and Hall, unpublished data) indicate continued dramatic losses of *T. testudinum* in western Florida Bay. Thus, the long-term future of seagrasses in Florida Bay is uncertain.

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THE DISTRIBUTION OF SEAGRASS AND BENTHIC HABITATS WESTWARD OF THE PATCH REEF SYSTEM BOUNDARY IN BISCAYNE NATIONAL PARK, FLORIDA, USA

Roy R. Lewis III, Ann B. Hodgson, Marcus Tooze, Curtis D. Kruer

ABSTRACT

The purpose of this study was to map the distribution of seagrasses and benthic habitats lying between the patch reef system that demarcates the eastern boundary of Biscayne National Park and the western shoreline of the Park. The park encompasses a 55,000-hectare marine ecosystem embayment of the Atlantic Ocean, offshore southeast of Miami on the eastern coast of Florida, USA. Benthic habitats were delineated using RGB color aerial photographs acquired of the study area in 1997. Photographs were mosaiced, imported into ARCInfo software and rectified. Photographic signatures of delineated polygons of seagrass and benthic habitats were verified by diving throughout the mapping study area, then map accuracy was verified again by diving using draft habitat delineation maps produced in 1999. A digital GIS map of the seagrass and benthic habitat distribution in the study area was produced. This study presents a novel classification of the benthic habitat types in Biscayne National Park and provides a contemporary, detailed map of the benthos in the Park. Eight habitat type classes in three categories: seagrasses, live bottom and bare sand, were identified and 49,811.36 hectares of all types of benthic habitats were mapped. The dominant habitat was moderately dense to dense seagrass distributed over 25,445.93 ha of the area. Less common habitats included dense patches of seagrass in a hard bottom matrix, moderate to dense discontinuous seagrass with blowouts or sand or mud, and sand or mud with small, scattered seagrass patches. Hard bottom habitats covered 13,793 ha, and bare sand occurred on 441 ha. Anthropogenic factors including stormwater runoff, propeller scarring and boat groundings appear to be significantly degrading portions of the Park's benthos.

INTRODUCTION

This study was conducted to map, using aerial photography, the distribution of seagrasses and benthic habitats westward of the patch reef system that demarcates the eastern boundary of Biscayne National Park, and to develop a digital geographic information system (GIS) map, in ARCView/ARCInfo format, of the habitat distributions in the study area. The east coast of Florida is a high-energy shoreline bounded by areas characterized as lagoons behind barrier islands (Seaman 1985) and typically affected by semi-diurnal tides that are 0.6 m or greater (Livingston 1990). Seagrass beds, tidal flats and marshes, soft sediments, hard substrates, shellfish beds, and a series of transition zones including those affected by people characterize the inshore marine system.

Biscayne Bay, located on the southern Atlantic coast of Florida, is a shallow lagoonal embayment comprised of estuarine seawater diluted by land runoff (Livingston 1990). The drainage area of the Biscayne Bay system is a watershed of approximately 480 km^2, located south of the Indian River watershed, which discharges to a series of canals and rivers with a combined average daily flow of 91 m^3/s, although much of the natural drainage pattern has been altered by urbanization and anthropogenic influences. Deterioration of seagrass in northern and southern Biscayne Bay was reported in the 1960s–1970s (McNulty 1961, Zieman 1970) and continues due to ongoing population growth in the Miami-Homestead, Florida area. The bay substrate is composed primarily of shell and coral ($CaCO_3$) fragments (Fernald 1981) which supports extensive beds of turtle grass (*Thalassia testudinum*), and other seagrass species including manatee grass (*Syringodium filiforme*), which occurs in mixed seagrass beds or monospecific patches (Jaap and Hallock 1990), stargrass (*Halophila* *engelmannii*), shoal grass *(Halodule wrightii)*, and widgeon grass (*Ruppia maritimae*) (Livingston 1990). The sediment depth affects the distribution of seagrass species (Fonseca et al. 1981, Scoffin 1970, Zieman 1972). Several types of calcareous green algae occur in subtropical seagrass communities including *Halimeda*, *Penicillus*, *Rhipocephalus*, and *Udotea* (Jaap and Hallock 1990). Shallow water (<6 m) live-bottom communities of sponges, soft coral and benthic organisms occur throughout the bay, and soft bottom (unvegetated) areas support microorganisms, meiobenthic organisms, and macrobenthos.

Biscayne National Park includes 708 km^2 (Jaap and Hallock 1990) within a marine embayment. Benthic habitats had been mapped and classified in this region of the Florida coast previously (Milano 1983, NOAA and FDEP/FMRI 1995), however, the U. S. National Park Service is responsible for the management of the park and determined in 1997 that an updated map of the benthic habitats of Biscayne National Park was needed to allow the agency to manage benthic resources.

METHODS AND MATERIALS

GIS Map Production

Aerial photography of the study area was flown on 10 November, 1997 along twelve parallel flight lines running north to south across the study area during low tide using Eastman Kodak Aerocolor HS S0358 film (Aerial Cartographics of America, Inc., Orlando, FL). Weather conditions during the flight were clear and surface winds were less than 5 mph. Photographs were scaled to 1":12,000’ and polygons were delineated under 10x magnification using a 2.0 rapidograph pen on clear acetate overlays registered to 147 unrectified color diapositives. Individually delineated polygons were uniquely labeled. Initial field checking was conducted by surveying 160 sampling stations with an underwater viewing scope or by diving on 3–4 April, and 8, 11–13 May 1998 to confirm the photographic signatures of the delineated benthic habitats. Features and habitats not identifiable in the diapositives were not mapped.

Acetate overlays were mosaiced into ‘tiles’ and matched. Polygons were digitized in ARCInfo Version 7.2.1 (Environmental System Research Institute, Redlands, CA). One ground control point (GCP) per photograph was collected from USGS Digital Ortho Quarter-Quads (DOQQs) and additional ground control points were synthesized from obvious features on the aerial photography and some ground control targets installed prior to the aerial photography. Tiles were rectified using the GCPs to a Universal Transverse Mercator (UTM) projection, edge-matched, merged, and labeled. A digital boundary file provided by Biscayne National Park and National Oceanic and Atmospheric Administration (NOAA) navigational charts for Biscayne Bay were used to establish the mapping boundary of the study area. A metadata file was attached to the digital mapping file (FGDC 1995). The draft seagrass and benthic habitat delineation map (scale: 1":10,742’) was overlaid with an ARCView-generated grid of stratified random sample points and mapping accuracy was sampled March 5–10, 1999 using a Polar research boat outfitted with a Garmin 75 DGPS (Garmin International, Olathe, KS, USA) to locate the pre-plotted sample points in the field. Direct observation, underwater viewers and subsurface diving were used to confirm the mapped delineations. In a statistical sample cumulative mapping accuracy was 85% ($p=0.10$).

Benthic Habitats Classification System

The study area included benthic habitats westward from the patch reef boundary in Biscayne National Park, Dade County, Florida, USA. Seagrasses and benthic habitats were classified according to their community characteristics using a modification of existing habitat classification systems (Florida Marine Research Institute 1998). Three major categories: seagrasses, hard bottom, and bare substrate, were mapped. Seagrass habitats were separated into classes based on seagrass density, distributional patchiness, and type of substrate present (i.e., "soft bottom" or "hard bottom"). The special class modifier "b" was used to indicate a polygon that occurred in a "bank" or periodically exposed condition where seagrasses were present as intertidal seagrass banks or "flats". The resulting system included eight habitat type classes, and one special class modifier. Patch reefs that occurred in the oceanside zone near the eastern project limit were mapped as hard bottom with some seagrass and appeared as distinctive features when compared to inshore hard bottom areas. Seagrass species were not differentiated during mapping. Certain areas were mapped and labeled as ""unmappable / uninterpretable" because the habitat type occurring in those areas could not be determined due to poor water clarity, water depth, or sun glare on the photographic images. Descriptions of the mapped habitat types are provided in the following paragraphs.

Continuous Seagrasses

Continuous seagrass is the dominant, and diverse, habitat type in Biscayne Bay; the distinguishing characteristic is continuous seagrass in monotypic or mixed beds of varying densities.

Moderate to dense, continuous beds (SD): Continuous *Thalassia testudinum*, *Syringodium filiforme*, and *Halodule wrightii* occurs individually or in mixed beds with 95100% cover from the intertidal zone to about 7 m (offshore). The seagrass beds have widespread occurrence and diverse signatures. *Halodule* was most common in the southwest and west side of the Bay, *Syringodium* was most common in the eastern Bay and offshore, and *Thalassia* was found throughout the Bay. Substrates underlying these beds are typically soft but may be only a thin layer over rock. Macroalgae of various species (e.g. *Halimeda* sp., *Panicles* sp., *Udotea* sp., *Avrainvillea* sp., and various red algae) are commonly found in these beds. Sparse seagrass in patches may be a component as well.

Sparse, continuous beds (SS): Seagrasses *(Thalassia sp.* or *Halodule* sp.) occur in these areas in low density (d50 shoots/m^2), typically in shallow protected bays where physical conditions or the substrate limits development. These areas were often hard to distinguish on aerial photographs from hard bottom or bare bottom habitats.

Patchy Seagrass

A common and diverse habitat type in Biscayne Bay, the distinguishing characteristic is some amount of identifiable seagrass in a matrix of other bottom types. These areas are often the result of depressional basins or creek-like features in the hard bottom seabed that accumulate sediment or contain organic deposits from mangrove communities occurring during lower sea level. Greater sediment depth allows seagrass development in an area, or allows denser seagrasses as compared to surrounding areas.

Dense patches of seagrass in a matrix of sparse seagrass (SPS): Patches occur as depressional features where deep sediment allows denser development of seagrasses than on surrounding bottoms where only a thin layer of sediment is present supporting sparse seagrass. Dense patches are difficult to discern on aerials from seagrass patches in hard bottom and may be more common in deeper parts of the Bay than mapped.

Moderate to dense, discontinuous beds (>50%) with blowouts and/or sand or

mud (SDB): Discontinuous *Thalassia*, *Syringodium*, or rarely *Halodule*, individually or in mixed beds with open, essentially unvegetated areas composed of sand or soft substrates is typically found on the east side of the Bay and offshore where the bottom substrates are other than rock. The occurrence of discontinuous beds ranges from the intertidal zone to about 6 m depths offshore in moderate to high tidal current or wave energy regimes. Natural blowouts or patches are dispersed as holes or bare areas in otherwise continuous seagrass beds, especially near the entrance to tidal channels and passes. Vessel impacts (groundings or scars) resulting in sizeable open, unvegetated areas in seagrass are also mapped as patchy on some intertidal banks.

Dense patches of seagrass (>50%) in a matrix of hard bottom (SPH): One of the most common categories in the western Bay and in channel passes to the east, open patches of hard bottom occur in areas where a thin sediment layer over flat natural rock precludes continuous development of seagrasses. Open hard bottom patches may include sparse seagrass, considerable macroalgae, sponges, and soft corals.

Dominantly sand/mud with small scattered seagrass patches (<50% cover) (SPP): Sand or soft bottom areas with recognizable seagrass patches are often found in high-energy locations but also in deeper regions of the north Bay. The substrates may vary from sand to mud, and macroalgae, either fixed or drifts, may be a significant component. The offshore features tend to be sandier and protected, and deeper inshore areas tend to be muddier substrates.

Largely macroalgal cover with scattered seagrass patches (SPA): This is an unusual category in South Florida where scattered seagrass patches are a significant habitat component, but the dominant background habitat is macroalgae. Algal cover may be a bank of *Halimeda* sp., *Penicillus* sp., *Udotea* sp., *Avrainvillea* sp., or *Caulerpa* sp., depending on the exposure and substrate. This habitat type is difficult to identify and delineate on photographic images.

Hard bottom (HS)

Hard bottom with perceptible seagrass (<50% cover) is one of the most common classes mapped in the Park. Seagrasses usually occur in patches, depressions and basins in hard bottom where adequate sediments have accumulated, but constitute <50% of visible bottom coverage and often much less. Hard bottom may be mostly unvegetated, and may include solitary hard corals and soft corals, but most often includes a variety of macroalgae as well as sponges. Large areas of HS in western and central portions of the Bay may also include sparse seagrass (usually shortbladed *Thalassia sp.)* or may include no seagrass over relatively large areas. These areas are considered a hard bottom class, as the rock substrate is the controlling influence.

Bare Substrate (BS)

This substrate consists of bottom areas composed usually of either sand or a thin layer of sand over rock, but in protected areas it may consist of mud or soft marl. The areas appear unvegetated but may include some macroalgae, and may include large blowouts in open water seagrass habitats. Bare bottom is most common in the northeast area of the Park.

Unmappable/Uninterpretable (U)

The classification ‘unmappable/ uninterpretable’ was used to designate areas where the water depth obscured the photointerpretation of the substrate and/or the substrate condition was uninterpretable due to turbidity, albedo, etc. These areas occurred in most narrow or deep tidal channels and the deepest portions of the

Bay, especially in the northeast Bay where turbid water is common.

Special Modifier – Intertidal Banks (Xb)
Banks are typically intertidal seagrass habitats, even if they are intertidal only on the lowest of spring low tides. Intertidal banks are present along most shorelines of Biscayne Bay and on large bank features in the northern Bay and mid-Bay. Banks possess a distinctive signature on aerial photographs as compared to the surrounding bottom. Sometimes thinned or "burned" off areas with reduced seagrass density are visible on bank tops. Intertidal banks may also include hard bottom habitats along island shorelines that appear to be intertidal.

RESULTS AND DISCUSSION

A total of 52,947.75 hectares were mapped; the area of land polygons was then subtracted resulting in a total of 49,811.36 hectares of all types of benthic habitats, which included eight habitat type classes, and the class "unmappable/uninterpretable" (Figure 1). A comparison of the areal extent of each habitat type by acres and hectares and percent of total mapped benthic habitat is summarized in Table 1. The dominant habitat was moderately dense to dense seagrass that was distributed over 25,445.93 ha, of which 9.76% occurred on banks. This category included all seagrass species, as the species were not differentiated during the delineation. Less common habitats were 4,936.85 ha (0.57% in banks) of dense patches of seagrass in a hard bottom matrix (SPH), 2,934.05 ha (30.26% in banks) of moderate to dense discontinuous seagrass with blowouts or sand or mud (SDB), and 1,339.51 ha (0.94% in banks) of sand or mud with small, scattered seagrass patches (SPP). Uncommon habitat was 53.11 ha of sparse continuous seagrass (SS) (none of this habitat occurred in banks), and 127.99 ha (89.76% occurred in banks) of dense patches of seagrass in a matrix of sparse seagrass (SPS). Hard bottom (HB) habitats covered 13,793 ha (2.07% in banks), and bare sand (BS) occurred on 441.10 ha. The majority of the bare sand occurred in the northeast corner of the Park in the cut north of and eastward of Soldier Key. Overall mapping was very accurate and the category 'unknown' (U) was assigned to only 1.46% (729.41 ha) of the mapped benthic area.

Some areas of the Bay were difficult to photo-interpret as a result of water quality conditions which obscured benthic features or as a result of indistinct feature signatures. These areas included portions of the west side of the Bay where dark water appeared in images, probably as the result of canal drainage, and portions of the southwest Bay where hard bottom and sparse seagrass classes were difficult to separate. In the northern and mid-Bay deep water and poor water clarity affected photointerpretation, especially in differentiating areas of sparse seagrass and hard bottom, and on the inside of the east Bay islands intertidal communities (including seagrass and hard bottom habitats) were difficult to delineate.

CONCLUSIONS

This study presents a novel classification of the seagrass and benthic habitat types in Biscayne National Park. Results of this mapping elucidate the spatial distribution of the marine landscape in the study area within Biscayne National Park. Benthic habitats are generally distributed as an oblong basin of hard bottom surrounded and interspersed with continuous and patchy seagrass of varying densities in microhabitats determined by water depth, light penetration, temperature and other factors that affect seagrass community distribution.

Physical alteration of peripheral watersheds, the release of agricultural and industrial wastes, the lack of control of stormwater runoff from municipal devel-

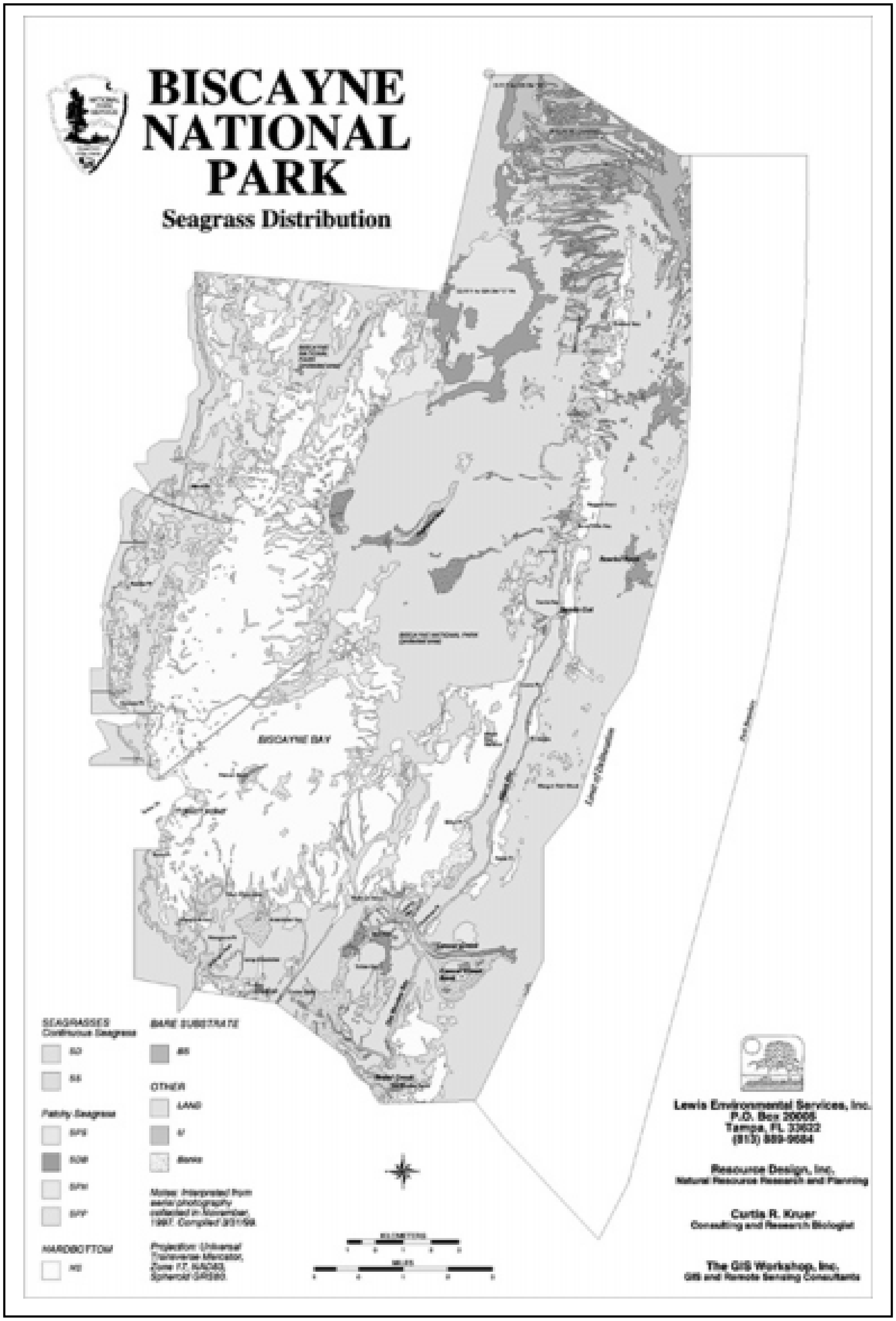

Figure 1. Map of benthic habitats westward of the patch reef boundary in Biscayne National Park, Florida, USA.

Table 1. Summary of acreages (hectares) of seagrass and benthic habitat types within the area mapped westward of the patch reef system boundary in Biscayne National Park as delineated in March 1997.

HABITAT CODE	ACRES	HECTARES	I
SD	56,739.01	22,961.49	
SDb	6,139.23	2,484.44	
SS	131.21	53.11	
SPS	32.36	13.10	
SPSb	283.89	114.89	
SDB	5,056.48	2,046.25	
SDBb	2,193.79	887.80	
SPH	12,153.89	4,918.63	
SPHb	69.72	28.22	
SPP	3,278.66	1,326.86	
SPPb	31.27	12.65	
HS	33,380.01	13,508.44	
HSb	704.19	284.97	
BS	1,089.97	441.10	
LAND	7,750.18	3,136.39	
U	1,802.34	729.41	
Total Mapped Area	130,837.19	52,947.75	
Total Mapped Benthic Area	123,087.01	49,811.36	

Legend:

SD	moderate to dense continuous seagrass
SDb	moderate to dense continuous seagrass on banks
SS	sparse continuous seagrass
SPS	patches in a matrix of sparse seagrass
SPSb	patches in a matrix of sparse seagrass on banks
SDB	moderate to dense continuous beds (>50%) with blowouts
SDBb	moderate to dense continuous beds (>50%) with blowouts on banks
SPP	dominantly sand/mud with small scattered seagrass patches
SPPb	dominantly sand/mud with small scattered seagrass patches on banks
SPH	dense patches of seagrass (>50%) in a hard bottom matrix
SPHb	dense patches of seagrass (>50%) in a hard bottom matrix on banks
HS	hard bottom
HSb	hard bottom on banks
BS	bare substrate
Land	land
U	unmappable/uninterpretable

opments and boating impacts due to propeller scarring and boat groundings remain the chief threats to the inshore marine systems of Florida (Livingston 1990, Sargent et al. 1995). The widespread occurrence of prop scar damage from increasing boating activities in shallow water has resulted in 11,220 acres of prop scarring in Dade County marine waters, much of it in Biscayne National Park (Sargent et al. 1995). Widespread decreases in water clarity in the northeast region of the Park in waters receiving urban runoff from the Miami-Homestead area were noted in this study. Strong enforcement and rapid restoration of propeller scars and grounding sites should be a priority management activity. It is anticipated that the production of this habitat classification map will provide a baseline against which the effectiveness of future marine resource management actions can be quantitatively measured.

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RESPONSES OF SUWANNEE RIVER TIDAL SAV TO ENSO-CONTROLLED CLIMATE VARIABILITY

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ABSTRACT

Submerged aquatic vegetation (SAV) of the tidal Suwannee River, Florida, was studied from January 1998 to January 1999, and in June 2000. Beds of SAV occurred in association with tidal freshwater and brackish marshes. Fifteen species of vascular plants were found. Delta shoals near the river's mouth contained fewest species and species richness was greatest near the intersection of East Pass. Species number increased at an overall rate of about 2 species per upriver kilometer. Downstream limits of species corresponded to patterns of mean salinity and salinity variability. Upstream limits were set by the availability of shallow substrata suitable for SAV. Within the range of SAV, maximum bed widths alternated banks with proximity to the Gulf, possibly reflecting geomorphic control. Based on percent cover data at 15 sites, species with greatest frequencies, densities and abundances were tape grass, *Vallisneria americana*, strapleaf sag, *Sagittaria kurziana*, and Eurasian water milfoil, *Myriophyllum spicatum*. Tape grass and strapleaf sag had highest cover values in middle-river reaches whereas highest *Myriophyllum* cover was upstream. Plant cover was high at the onset of the study, at a time of record river flows associated with El Niño floods. Plant cover decreased from June to September 1998 as decreasing river flows and Hurricane Earl's storm surge elevated salinity levels in the lower river. Biomass was measured at 6 stations from Alligator Pass toward Gopher River. Total above and below-ground biomass for all species combined was lowest at stations closest to the Gulf. Combined above-ground biomass was greatest in the middle river whereas maximum below-ground biomass included middle and upper river stations. Above-ground biomass for the dominant species generally corresponded to spatial patterns in cover, and declined during September's high salinity episode. A La Niña drought persisting throughout 1999 and the first half of 2000 led to a follow-up sampling trip in June 2000, a period of record low flows. Species richness, plant cover, and biomass were lower than previously measured, although one new species, *Ruppia maritima*, was collected. Responses were consistent with elevated salinities caused by the drought. Overall, geological, hydrological and chemical factors control the downstream limit and abundances of SAV in the tidal river. Salinity variation may also be a controlling factor. Permanently reduced flows of the river will increase salinity in the lower river and could cause an upstream retreat and overall reduction of SAV beds. Upstream SAV beds of the Suwannee estuary have the greatest species richness owing to their tidal freshwater situation. A loss of species in this community might not reduce total bed area, but cover, biomass, and habitat-level diversity could be affected adversely.

INTRODUCTION

Tidal fresh water environments are common across the coastal plain of the southeastern United States (Odum *et al*., 1984) but are comparatively rare along the Florida peninsula because many rivers are small and have low flows and long upstream tidal excursions. The general effect of gradual sea-level rise has compressed tidal fresh water reaches into small areas of all but Florida's largest rivers, such as the Suwannee River. Scant references existed regarding the submerged aquatic vegetation of the Suwannee River's tidal reach (Mattson, 2000), prompting a reconnaissance in January 1998. The preliminary survey established that the tidal Suwannee River harbored more submerged aquatic vegetation than previously known. Patterns observed in SAV dispersion and species abundance suggested that SAV were valuable as primary producers and faunal habitat in the lower river. SAV patterns may also help establish minimum flows and levels for the river because altered flow regimes are often manifested first, and most, in low-salinity and tidal-fresh reaches of rivers (Sklar and Browder, 1998).

THE STUDY AREA

The Suwannee River heads in the Okeefenokee Swamp and flows 385 km to the Gulf of Mexico, draining a watershed of 25570 km^2 . The tidal river study area is 54 km downstream of a USGS gage which captures a drainage area of 24968 km^2; annual mean and median flows past this reach are 298 m^3/s and 239 m^3/s, respectively. Springs comprise approximately one-half of average annual flows here. The study reach is tidally influenced. Tides at Cedar Key are mixed with a spring range of 1.15 m. The bed of the river is below sea-level nearly to its confluence with the Santa Fe River 106 km upstream. During periods of low river flow, tidal variation in river stage can be measured 77 km upriver. Tidally based reversals of flow do not occur much upstream of Gopher River, and when the criterion of salt is used as an indicator of tidal action, only the passes and river below Gopher River may be considered tidal. The river passes through dense mixed wetland forests comprised of baldcypress *(Taxodium distichum)*, pumpkin ash *(Fraxinus profunda)*, swamp tupelo *(Nyssa biflora)*, water tupelo *(Nyssa aquatica)*, sweet bay *(Magnolia virginiana)* and southern red cedar *(Juniperus silicicola)*. The passes dissect deltaic lowlands covered by rushes, grasses and sedges of tidal wetlands, interspersed with hammocks of cabbage palm *(Sabal palmetto)*. All passes but Alligator Pass have been dredged in the 1960s, 1970s, and 1980s (Wright, 1995). The shallow Gulf at the river's mouths is semi-enclosed by oyster reefs and is called Suwannee Sound. The Sound does not support extensive SAV.

CLIMATE

River flow was high at the beginning of the study period and generally declined for the remainder of the year. Three significant hydrological events related to the El Niño-Southern Oscillation (ENSO) occurred during the study period. An El Niño winter spanning late 1997 and early 1998 brought several months of above-average rainfall to Florida, resulting in above-average river flows and a maximum historic flow at Branford, relative to 1931-1999, coinciding with the first quarterly SAV sampling trip in March 1998. Flooding was extensive throughout the watershed and the river was out of its banks through most reaches. Discharge was so great that the river was fresh to its mouth. River flows declined steadily through spring and summer but were high in October, in part because of rainfall in the watershed associated with Hurricane Earl. Earl was a category 2 hurricane that crossed from the Gulf of Mexico to the Atlantic Ocean during the first few days of September 1998. It made landfall near Panama City, Florida as a category 1 hurricane but caused the largest storm surges along the Big Bend coast. According to the National Hurricane Center, storm surge was estimated to be near 8 ft in Franklin, Wakulla, Jefferson and Taylor counties and approximately 6 to 7 ft in Dixie County, where the town of Suwannee is located (http://www.nhc.noaa.gov/1998earl.htm). Finally, a drought throughout the southeastern United States began approximately in January 1999, the result of an ENSO cold-phase called La Niña. The La Niña drought persisted through the latter part of 2000, with record low flows of the Suwannee River at Branford during June 2000 (Figure 1). Penetration of saline Gulf waters into the tidal river matched hurricane and drought-induced low-flow events (Figure 2).

METHODS

A general SAV census was made during 1998–99. At 0.25 km intervals, the bank having the most SAV was visited. SAV species were noted and the bank-normal width of the SAV bed was estimated at each site shown with a circle on Figure 3. Sixteen stations were selected for study (squares in Figure 3). Intensive stations were sampled on five occasions—March,

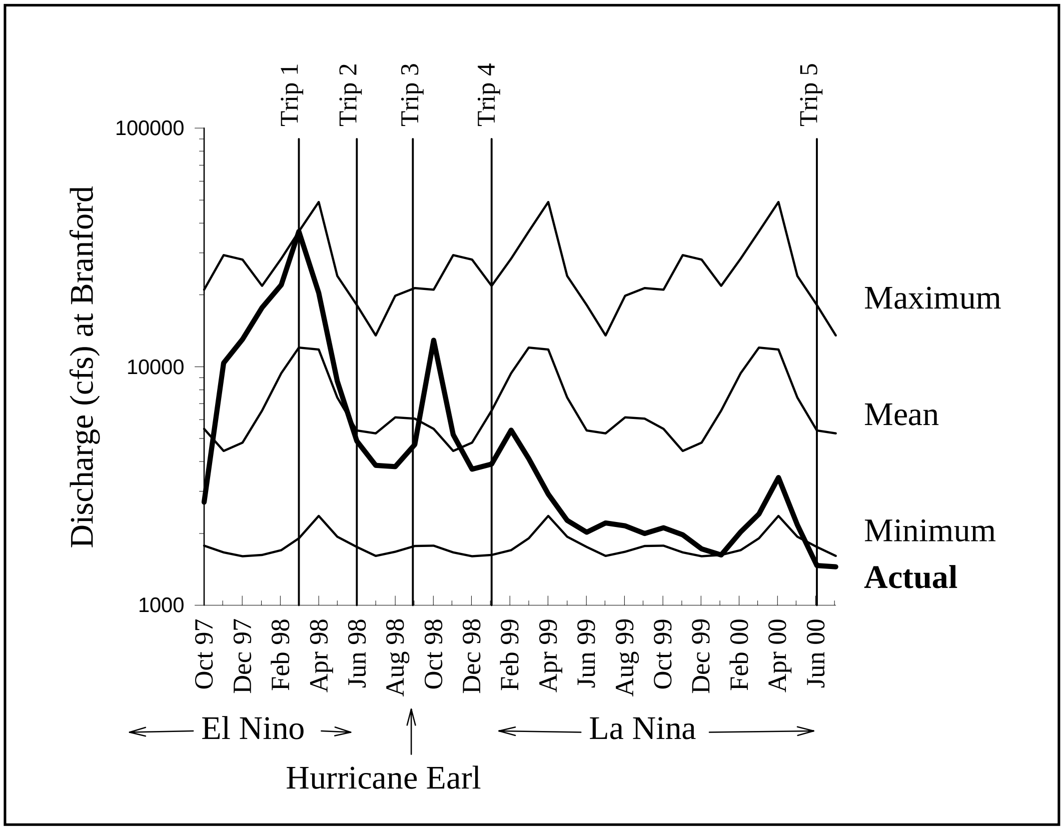

Figure 1. Historic versus actual discharges of the Suwannee River at Branford, 120 km from the Gulf of Mexico.

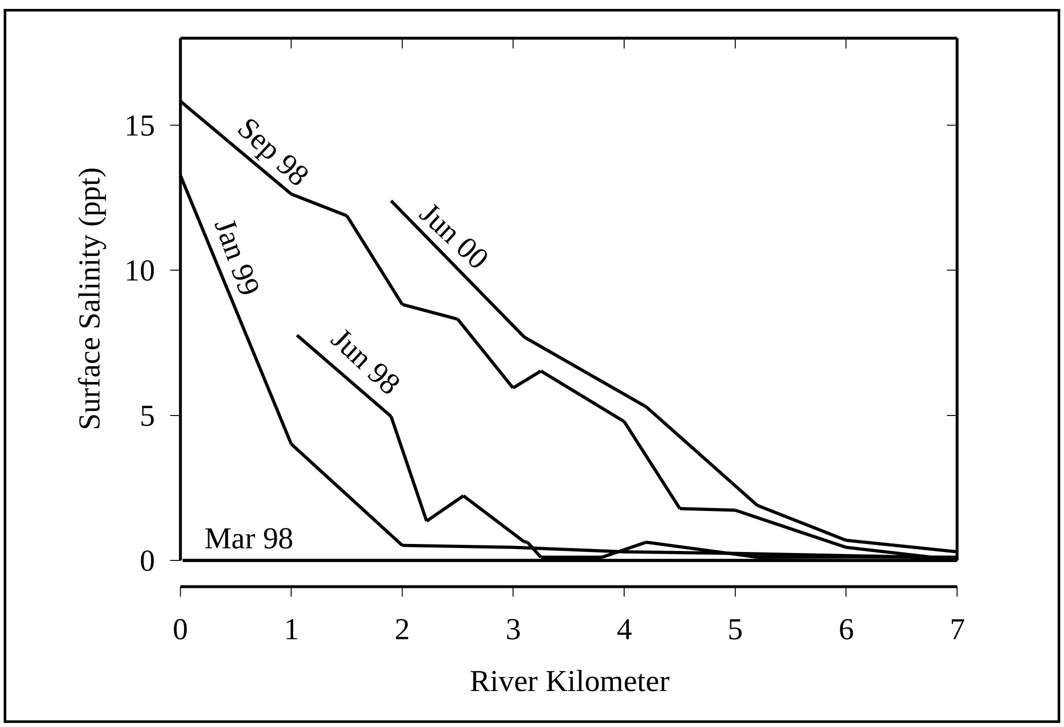

Figure 2. Mid-channel surface salinity (ppt) measured near slack high water. June 2000 data provided by U.S. Geological Survey.

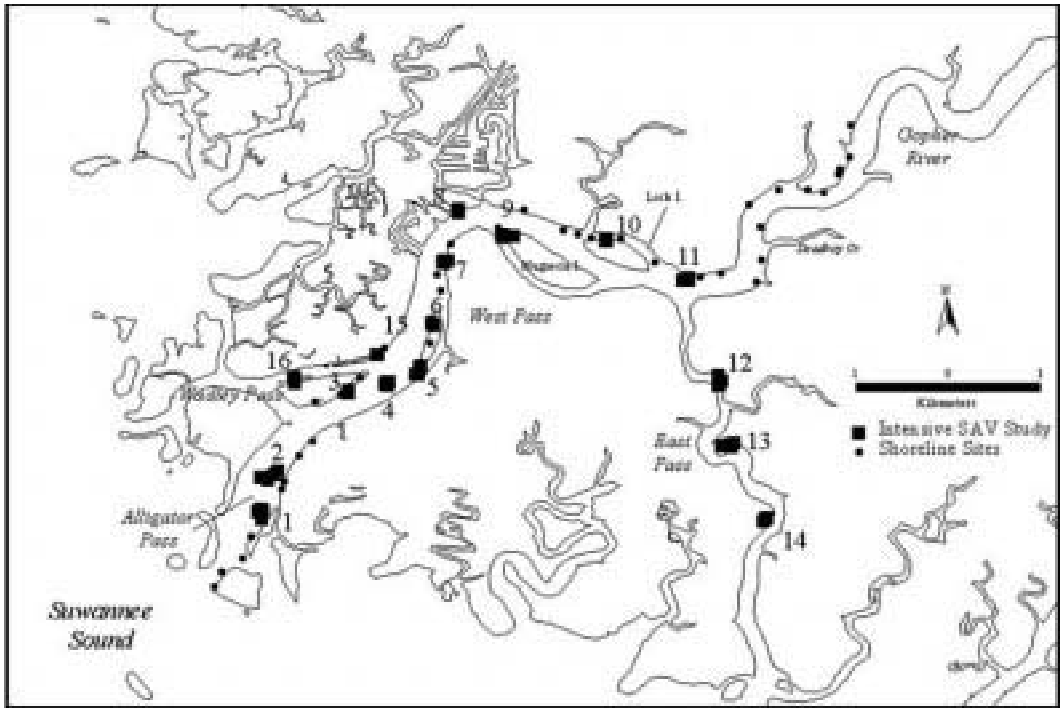

Figure 3. Lower Suwannee River; East, West and Alligator Passes, and Suwannee Sound. Station 17 at the mouth of Gopher River is not shown, and was sampled only in June 2000.

June, and September 1998, January 1999, and June 2000. Percent cover estimates were made following Braun-Blanquet (1932). Four quadrats, (each 0.25 m^2 in area) were deployed at random over SAV. Percent cover was scored for each species within a quadrat using the following scale: BB1= <5%; BB2= 5–25%; BB3= 25–50%; BB4=50–75%; BB5= 5–100%. There were 64 quadrats deployed per trip and 320 quadrats were deployed overall. At Stations 2,4,6,8,9 and 10, above- and below-ground biomass were determined by harvesting all plant material from five 25 cm by 25 cm (0.0625 m^2) quadrats. Samples were taken where SAV was present rather than at random. All plant material from a quadrat was washed in river water; bagged, and stored on ice. At the laboratory, contents were re-washed, sorted by species and fraction, and air dried for 24 hours. Material was dried to constant weight at 75-80 ^{o}C in tared foil boats and weighed with a Mettler PE 3600 top-loading electronic balance calibrated with NBS standards. Weights were converted to grams dry weight per m^2. Slack high-tide salinity profiles were made along the centerline of the channel with samples and measurements made at points corresponding to the location of bank stations, or 1 km intervals from the mouth of each pass. Near surface and near bottom salinity measurements were made with a bench-tested YSI Model 33 SCT meter.

RESULTS AND DISCUSSION

Dispersion: SAV was more abundant in the middle reach of the study area than at extremes (Figure 4). Near the Gulf, SAV is rare despite large shallow areas of the delta shoal. Near Gopher River banks are narrow and steep, offering little SAV habitat. Largest local areas of SAV grew on alternate sides of the river according to sediment availability; wide, shallow flats occurred at and downstream of river bends

Species Diversity: Thirteen species of submerged aquatic macrophytes were observed or collected, and two others are known (Table 1). Widgeon grass occurs in

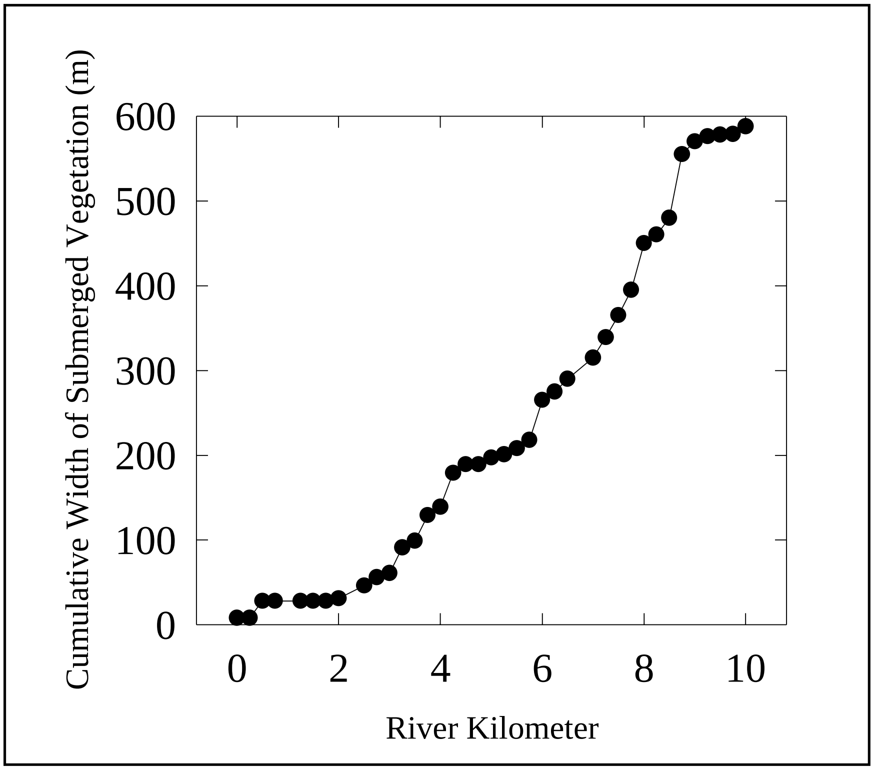

Figure 4. Bank-normal width of SAV beds at 0.25 km intervals, in meters, excluding bands of barren bottom internal to SAV beds. Data collected from January 1998 to January 1999.

upper estuaries and back bays but was encountered only in June 2000 when river salinity was high. Florida tape grass tolerates salinities up to 6–12 ppt (Doering and Chamberlain, 2000); the majority of species were associated with oligohaline and tidal-fresh waters.

Species Distribution: No species occurred throughout the length of the study area (Figure 5), though tape grass, strap-leaf sag and Eurasian water milfoil nearly did. Species had differential downstream limits, and overall diversity was greatest below Gopher River. Species loss was greatest downstream of the town of Suwannee, especially with the onset of drought conditions.

SAV Cover: Dominant species survived the first winter period of high flows with generally high cover values which induring the summer growing season (Figure 6). Elevated salinities caused by Hurricane Earl, and decreasing flows associated with the onset of the drought, contracted the range and depressed cover of Eurasian water milfoil. Prolonged drought resulted in June 2000 cover values much lower than values from June 1998.

Dominance, Abundance, and Density: Braun-Blanquet cover data were used to compute dominance (number of quadrats with species/total number of quadrats); abundance (sum of cover values/number of quadrats with species), and density (sum of cover values/total number of quadrats). For the three dominant SAV species (tape grass, strap-leaf sag and Eurasian water milfoil), computed attributes depicted similar patterns of response to increased salinity. Using abundance as the example, abundance values computed for Eurasian water milfoil followed patterns of raw cover data (Figure 6). Abundance values of strap-leaf sag were moderate to high at mid-river stations in March 1998. Higher

Table 1. Oligohaline Submerged Aquatic Vegetation of the Tidal Suwannee River.

SCIENTIFIC NAME	COMMON NAME	SOURCE(S)
Cabomba caroliniana	Cabomba	1
Ceratophyllum demersum	Hornwort	1
Cladophora sp.	a green alga	2
Chara sp.	Musk-grass, a green alga	1
Egeria densa	Brazilian elodea	1
Enteromorpha compressa	a green alga	1
Hydrilla verticillata	Hydrilla	1
Hygrophila sp.	Hygrophila	3
Isoetes flaccida	Riverbank quillwort	1
Ludwigia repens	Red ludwigia	1,2
Myriophyllum braziliense	Parrot’s feather	1
Myriophyllum spicatum	Eurasian water milfoil	1,2
Najas guadalupensis	Common water nymph	1,2
Nymphaea mexicana	Mexican water lily	1,2
Ruppia maritima	Widgeon grass	4
Sagittaria kurziana	Strap-leaf sag	1,2
Sagittaria subulata	Dwarf arrowhead	1,2
Vallisneria americana	Tapegrass	1,2
Zannichellia palustris	Horned pondweed	5

Sources: 1, shoreline survey; 2, noted during 1998-99 cover study; 3, R. Mattson, SRWMD, personal communication; 4, noted during June 2000 cover study; 5, Clewell et al. (1999).

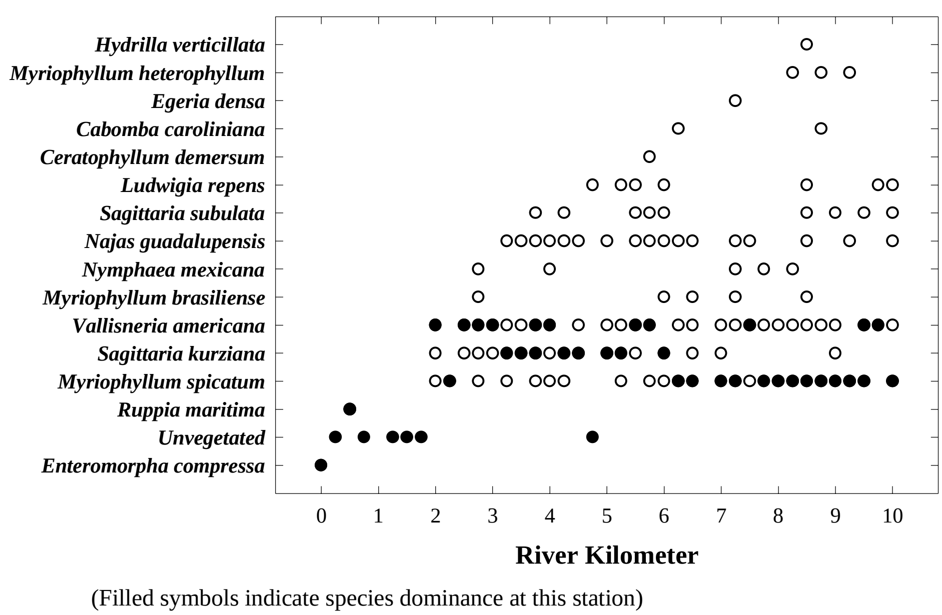

Figure 5. Longitudinal dispersion of SAV species in the tidal river, sorted by downstream range limit. Dominance based on cover data. Data collected from January 1998 to January 1999.

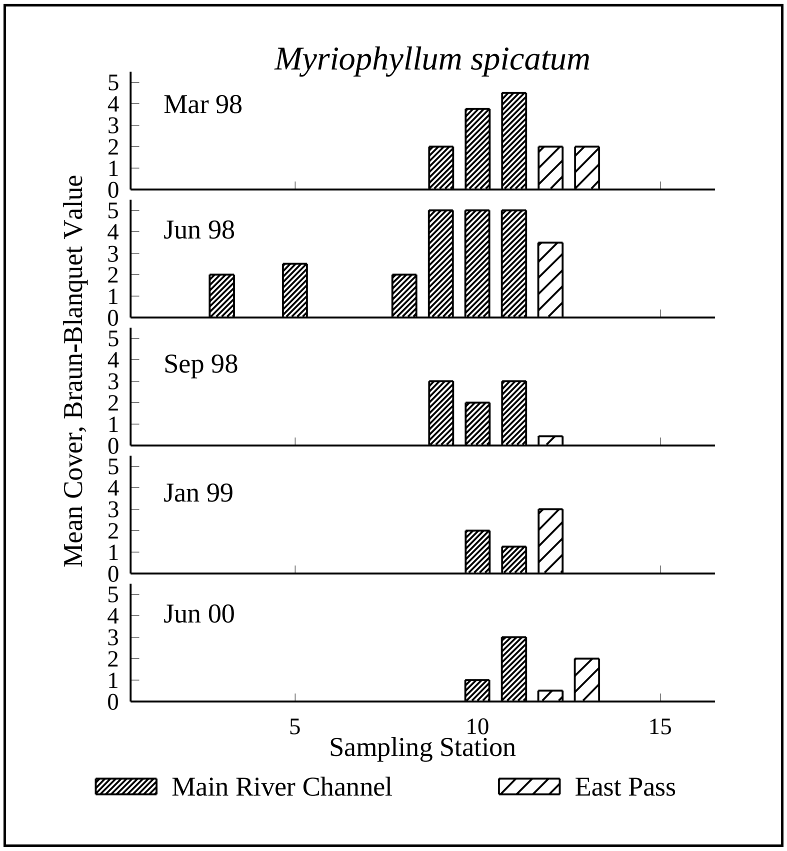

Figure 6. Mean cover of Eurasian water milfoil relative to river position and sampling date. See text for explanation of cover values.

abundances were calculated in June and September 1998 when strap-leaf sag actually extended down-river to Station 4. In January 99 abundance values were low at the only two stations (8,11) where strap-leaf sag was found. In June 2000 abundance values were very high but only at Stations 9–11, and 15. By contrast, abundance values for tape grass were moderate to high at all stations along the tidal river, during all sampling periods but June 2000. Then, abundance values were high where tape grass was found (Stations 7–11, and 17) but tape grass could not be found at all at Stations 2–6. Results from June 2000 showed that tape grass was very abundant at surface salinities lower than 5.0 ppt.

Above-Ground Biomass: Highest tape grass and strap-leaf sag biomasses occurred at Station 6 and 8, respectively, whereas Stations 9 and 10 had highest biomass of Eurasian water milfoil. All species reached greatest biomass during June 1998 and in general all species followed a seasonal pattern depicted in Figure 7, in which increasing salinities depressed above-ground biomass. The contrast is most evident comparing June 1998 to June 2000.

SUMMARY AND CONCLUSIONS

The tidal Suwannee River supports a large and productive SAV community dominated by 3 species but including a dozen other species with oligohaline and tidal-fresh affinities. Standard biomass (kg/m^2) of all Suwannee species combined compares favorably to published values elsewhere in Florida and the world (Table 2). Within the Suwannee drainage, tidal river biomass is comparable to that measured at Ginnie Spring, but lower than

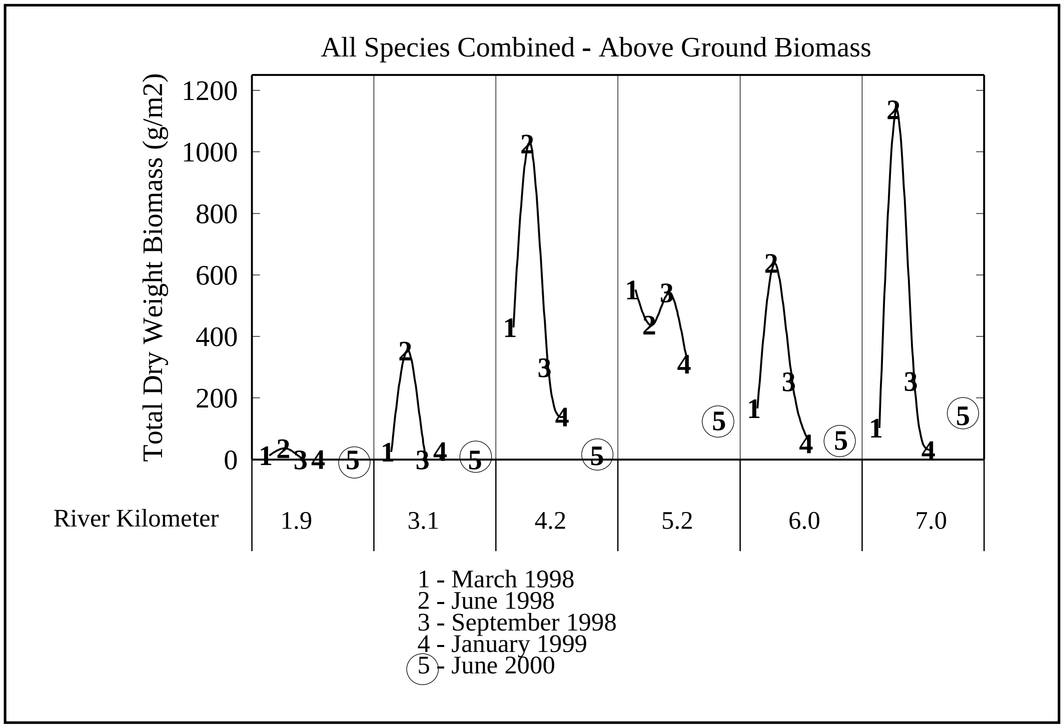

Figure 7. Total dry-weight, above-ground biomass values for tape grass, strap-leaf sag, and Eurasian water milfoil, combined, relative to river position and sampling date. Contrast June 1998 to June 2000.

Table 2. Mean above-ground biomass of combined SAV species, kg/m^2.

REGION	RIVER SYSTEM	FRESH	OVEN-DRY	SOURCE
Suwannee	Blue Spring	0.35		Duarte & Canfield 1990
Drainage	Ginnie Spring	7.20		same
	Ichetucknee	10.78		same
	Fanin Spring	15.73		same
	Manatee Spring	11.43		same
	ESTUARY	7.3*	0.72	THIS STUDY**
Florida	Wacissa	11.0		Canfield & Hoyer 1988
	Wekiva	2.6		same
	Alafia	0.2		same
	St. Marks	3.9		same
	Hillsborough	3.7		same
	Lake Okeechobee			
			0.45	Hopson & Zimba 1992
Others	Mobile Bay		0.14	Stout & Heck 1991
	Mississippi River		0.22	Donnermeyer & Smart 1985
	Ontario		0.25	Crowder et al. 1997
	China	1.54		Guan et al. 1997
	New South Wales		0.57	Royle & King 1991

*Based on a conversion provided by T.K. Frazier, University of Florida (pers. commun.)
**Calculated for biomass stations 4,6,8,9,10 in June 1998

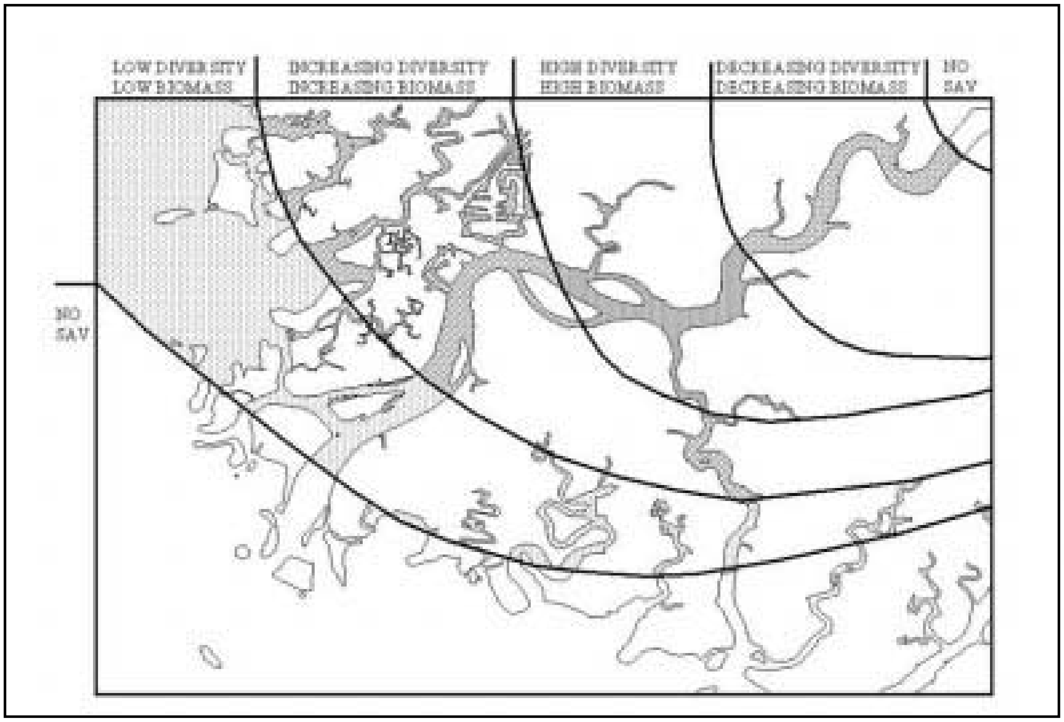

Figure 8. Schematic depiction of spatial patterns in species diversity and above-ground biomass for SAV in the tidal River. SAV grows upstream of the study area as small discontinuous patches of various species. Patterns apply only to main river and distributary channels.

biomass measured at other springs.

Through studies during a period of high climate variability, we have discovered the tidal river's SAV community to possess strong gradients in species richness and plant abundance (Figure 8). A river reach near the divergence of East and West Passes contained the highest persistent species richness and biomass values. Hurricane and drought effects were evident in the reach but not as extensive or continuous as observed in river reaches closer to the Gulf of Mexico. From evidence including record high and record low flows, and a hurricane, salinity appears to a stronger factor affecting SAV dispersion and abundance than sediment type or water clarity. In addition to average salinity conditions, salinity variation and extremes may also be controlling factors.

Other studies of natural variation and storm effects demonstrate that introduction of brackish water to reaches that historically have been tidal freshwater has the potential to simplify the SAV community by eliminating sensitive species. Purcell (1977) found that seasonal elevations of salinity retarded *Vallisneria* in East Bay, Florida, and it is clear from other studies (Woodward-Clyde,1998) that seasonal salinity increases are sufficient to regulate the dispersion and abundance of tape grass. When the "Storm of the Century" drove salt water into Kings Bay, Florida, tape grass and several other species of freshwater aquatic plants were greatly reduced in abundance, but later returned (Bishop and Canfield 1994).

Natural responses signify that permanently reduced flows of the river will increase salinity in the lower river and could cause an upstream retreat and overall reduction of SAV beds. Retreat could result as the seaward limit of SAV moves upstream as a result of salinity stress. Reduction could result because Suwannee Sound lacks beds of estuarine or marine species to recruit into the lower river if salinity rises—the barren area of the Sound and delta shoals could penetrate into the river, instead.

Introduction of brackish water to reaches that historically have been tidal freshwater has the potential to simplify the SAV community by eliminating sensitive species. The upstream SAV beds of the Suwannee estuary have the greatest species richness owing to their tidal freshwater situation. A loss of species in this community might not reduce total bed area, but cover, biomass, and habitat-level diversity could be affected adversely. In worst cases of flow reduction from natural or anthropogenic causes, the habitat type of tidal freshwater SAV bed could possibly cease to exist altogether. To the extent that it elevates salinity in the lower river, dredging of navigation channels could also cause SAV retreat and reduction. With or without flow reductions or channel deepening, sea-level rise may pose special problems for the maintenance of SAV in the deltaic ecosystem of the lower Suwannee River (Day et al., 1997).

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SEAGRASS RECOVERY IN WEST GALVESTON BAY

John Huffman

BACKGROUND

Submerged aquatic vegetation (SAV) beds have been all but extirpated from the Galveston Bay system. Estimates of their aerial coverage during the 1950s range from 2,500 to 5,000 acres. West Bay contained the most extensive seagrass meadows with approximately 2,200 acres of SAV beds, comprised primarily of shoalgrass, *Halodule wrightii.* By 1989 seagrasses in West Bay were completely eliminated and by 1993 only about 700 acres remained in the Galveston Bay system primarily in Christmas Bay (Figure 1). Their loss has been attributed to numerous causes, including subsidence, increased water column turbidity from dredging, shrimp trawling, contaminant discharges, chemical spills, or other factors that reduced light attenuation or otherwise inhibited SAV growth.

In 1995 numerous sites were identified along the south shoreline of West Bay that were colonized naturally by wigeongrass, *Ruppia maritima.* The growth of this ephemeral SAV indicated that water clarity or other environmental conditions may have recovered to conditions suitable for the restoration of permanent seagrasses in West Bay.

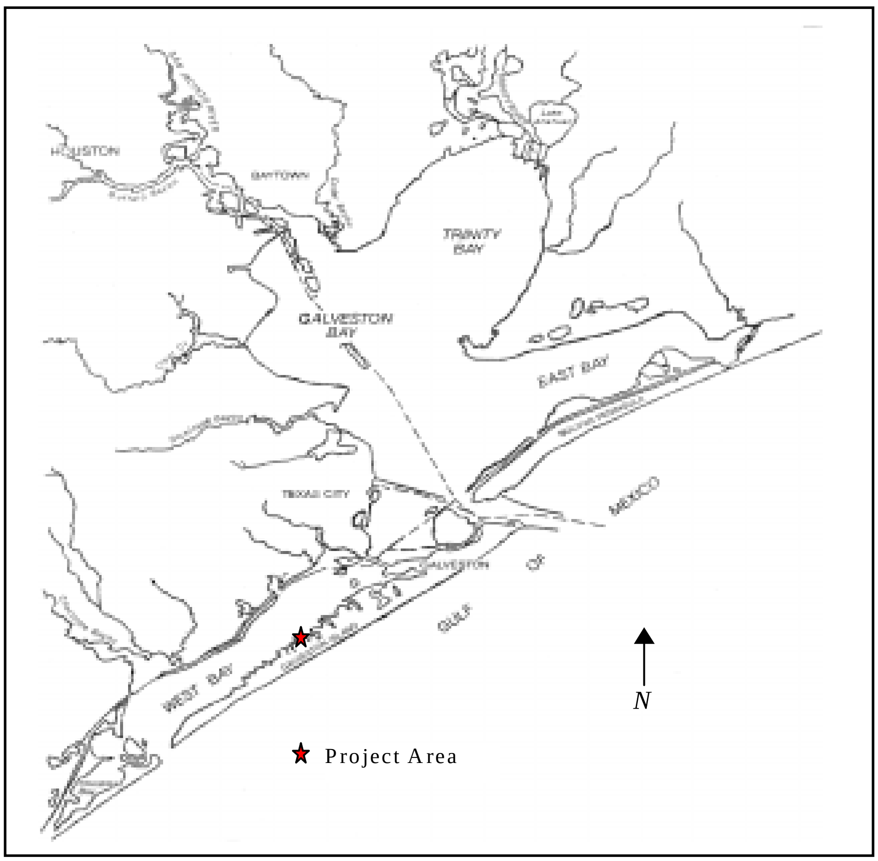

Figure 1. Galveston Bay system. In 1991 all remaining SAV beds were found in Christmas Bay in extreme SW corner of the system.

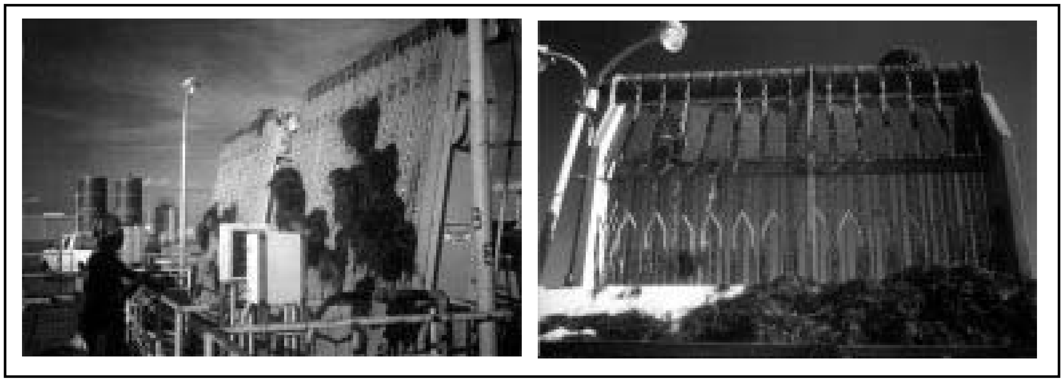

Figure 2. Mechanical rakes at the power plant remove seagrass debris off the intake screens (left). Seagrass wrack is continuously removed and landfilled (right).

OBJECTIVE

Seagrass wrack from the Laguna Madre (largest areas of seagrasses on the Texas south coast) composed of turtle grass *Thalassia testudinum*, manatee grass *Syringodium filiforme*, clovergrass *Halophila engelmanni* and shoalgrass collects in large amounts on the water intake screens at the Central Power and Light Plant in Corpus Christi, Texas. The seagrass that collects on the screens consists primarily of dead leaf litter throughout the year and is removed from the screens and landfilled by the truckload (refer to Figure 2). When feeding ducks and typically stronger waves arrive in late fall, the volume of wrack increases and more viable seagrass material is trapped on the screens. The objective of the project was to determine if seagrass wrack, collected during these high load periods, could be used to successfully restore seagrass beds. The successful use of the seagrass wrack would provide a very inexpensive and simple method of SAV restoration in Galveston Bay.

Broadcasting Seagrass Wrack

Two enclosures, 60 meters in diameter, were constructed to detain the seagrass wrack collected from the power plant. The enclosures were built in areas where widgeongrass had been found at the Galveston Island State Park (Figure 3) in West Bay. In October 1996, the Texas Coastal Program collected seagrass wrack material and with help from National Marine Fisheries Service personnel (NMFS), distributed the seagrass wrack into each of the enclosures (Figure 4). It was anticipated that the enclosures would retain the material for approximately six months. Small wooden stakes were also placed within the enclosures to catch drifting seagrass material and facilitate establishment of viable pieces.

RESULTS

The enclosures were first inspected two months after seagrass wrack was distributed. Portions of the fence were found cut or damaged on site inspections in February and March 1997 and most wooden stakes were missing. The fences were repaired and some material was found remaining in the enclosures. One plant was found growing during the March visit. No evidence of the introduced material was subsequently seen and the fence enclosures were removed about a year after their construction.

In 1998, in preparation for a marsh restoration project at the Galveston Island State Park, several small patches of clovergrass were first discovered in an area slated for marsh terracing approximately 200 meters south of the one enclosure. As a result of this discovery, the project was revised to avoid impacts to these seagrasses.

Figure 3. Galveston Island State Park, December 1995, prior to seagrass and marsh restoration projects.

Further surveys found more SAV beds throughout the West Bay cove where the two enclosures were located. Four years after initial broadcasting of the seagrass wrack, over forty acres total of clovergrass and shoalgrass beds have recovered in the Galveston Island State Park area as seen in December 2000 aerial photograph (Figure 5). Efforts to quantify the extent of SAV in the Galveston Island State Park are continuing.

DISCUSSION

The rapid expansion of clovergrass and shoalgrass from this project may be attributed, at least in part, to an abundance of seeds in the wrack material. Prior to this project, no beds of stargrass were present in West Bay and only small (less than 1 acre) planted beds of shoalgrass exist. Reviews of recent aerial photographs and field investigations have not identified the colonization by clovergrass in any West Bay coves other than the area planted by this project. It is highly unlikely that any material from existing SAV beds caused such rapid colonization in the project area. New projects underway will test this and other methods to restore seagrass meadows to West Galveston Bay.

New Restoration Efforts

The presence of healthy seagrass beds expanding in the area has encouraged new efforts in seagrass restoration in West Galveston Bay. Several projects have been initiated through the FWS' Texas Coastal Program such as; a test of planting techniques (the use of seagrass wrack, peat pot plugs and mechanical injection methods) by the NMFS, planting an additional 2 acres with mechanical injec-

Figure 4. Broadcasting seagrass wrack collected from the power plant intake screen into an enclosure in West Galveston Bay. Fragments were allowed to drift within the enclosure freely and settle to the substrate naturally.

Figure 5. Aerial photo of the seagrasses taken December 2000. Preliminary estimates indicate that more than 40 acres of seagrass (*Halophila engelmanni* and *Halodule wrightii*) have been re-established.

tion in cooperation with Texas Parks and Wildlife, and new aerial surveys and monitoring. The recovery of SAVs at the Galveston Island State Park has renewed efforts and hopes of restoring some of the 2,200 acres of seagrasses that were lost in West Galveston Bay. The results of these projects will provide guidance for expanded restoration efforts in the future.

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WATER DEPTH (MTL) AT THE DEEP EDGE OF SEAGRASS MEADOWS IN TAMPA BAY MEASURED BY GPS CARRIER-PHASE PROCESSING: EVALUATION OF THE TECHNIQUE

J.O.R. Johansson

ABSTRACT

The Tampa Bay Estuary Program (TBEP) has selected seagrass restoration target depths for each major bay segment at which adequate light conditions (20.5% of subsurface PAR irradiance) shall be maintained to ensure seagrass growth and the long-term Tampa Bay seagrass restoration goal of 15,400 ha. To evaluate the progress towards the goal, information on today's seagrass depth distribution is needed. Specifically, a need exists to accurately determine the water depth at the deep edge of the meadows for each seagrass species in different sections of the bay.

A relatively simple technique that provides elevation measurements, related to the mean tide level (MTL), of Tampa Bay seagrass meadows is described and evaluated. The technique uses mapping grade differential Global Positioning System (GPS) carrier-phase processing equipment that is currently owned by several TBEP partners.

The elevation of a specific seagrass location is determined by placing one GPS instrument as a base station at a surveyed benchmark with a known elevation above MTL and a second instrument at the seagrass site to be surveyed. Tests of measurement errors indicate that the technique yields elevation measurements with an error that is less than ±10 cm for survey sites located up to 10 km from bench mark sites.

Field evaluations of the technique that included measurements in the four major bay segments and the deep edge of the three major Tampa Bay seagrass species, *Halodule wrightii*, *Thalassia testudinum,* and *Syringodium filiforme,* were conducted at ten Tampa Bay seagrass study sites.

The depth of the measured deep edges ranged from about -0.30 m MTL for *H. wrightii* meadows in the upper section of Hillsborough Bay to near -2.0 m MTL for *S. filiforme* meadows on the southwestern side of Middle Tampa Bay. All sites surveyed had deep edge elevations shallower than the TBEP seagrass restoration target depth for the respective bay segment.

The estimated average percent of subsurface incident light available at the deep edges of the surveyed seagrass meadow ranged from 59.8% to 28.9% for *H. wrightii*, from 19.0% to 16.9% for *T. testudinum,* and from 16.7% to16.2% for *S. filiforme*.

The differential GPS carrier-phase processing technique was field practicable and measured seagrass elevations with acceptable quality. The field measurements provided an important first step in understanding the current depth distribution of the major Tampa Bay seagrass species. However, many more elevation measurements should be obtained to yield a more complete understanding of the seagrass depth distribution in the bay.

INTRODUCTION

The Tampa Bay Estuary Program (TBEP) has adopted a long-term Tampa Bay seagrass restoration goal of 15,400 ha, which is approximately 95% of the estimated Tampa Bay seagrass cover present in 1950. Protection of the 10,400 ha existing in 1994 and the restoration of an additional 5,000 ha will be accomplished primarily through management of external nitrogen loadings and bay water quality.

The Tampa Bay seagrass restoration goal was established through a multistep process that included the identification of specific seagrass restoration areas from comparisons of ca. 1950 and 1990 high altitude aerial photography. Areas that had lost seagrass over the 40-year period and that had not been physically altered to prevent future seagrass recolonization were selected for restoration (Janicki and Wade 1996). It was further determined, through field studies conducted in Lower Tampa Bay, that *Thalassia testudinum* required a minimum of 20.5% of sub-surface irradiance to ensure healthy growth (Dixon 2000). This finding was adopted by the TBEP as an overall Tampa Bay seagrass light requirement target. Subsequently, water quality conditions and external nitrogen loading rates required to sustain a minimum of 20.5% of subsurface irradiance at the seagrass restoration areas in the major bay segments were determined from empirical models (Janicki and Wade 1996).

To link the seagrass restoration areas with the water quality and nitrogen loading based light target, it was necessary to determine to what depth seagrass grew in 1950. The 1950 seagrass depth distribution was estimated from apparent seagrass areas visible on ca.1950 high altitude vertical photographs that were overlaid on NOAA National Ocean Survey (NOS) sounding data collected between 1947 and 1958. The soundings were corrected to mean tide level (MTL) (Janicki and Wade 1996).

Estimates of the 1950 seagrass depth distribution were then used to develop bay segment specific seagrass target depths for Tampa Bay (Janicki and Wade 1996). The adopted approximate target depths were: -1.0 m (MTL) for Hillsborough Bay, -2.0 m (MTL) for Old Tampa Bay, -1.6 to -2.4 m (MTL) for Middle Tampa Bay (depending on sub-segment), and -2.5 m (MTL) for Lower Tampa Bay (see Figure 1 for location of bay segments). The Tampa Bay seagrass restoration goal will be accomplished when the deep edges of the seagrass meadows, delineated from the Southwest Florida Water Management District (SWFWMD) high altitude aerial photography, eventually extend to these depths in the respective bay segments.

The estimated 1950 Tampa Bay seagrass depth distribution was important for the development of the TBEP seagrass restoration and protection goal. Likewise, information on today's seagrass depth distribution is needed to evaluate the progress of the seagrass restoration process. Present-day depth information would yield a comparison to the estimated seagrass depth distribution in 1950. However, more importantly, the present seagrass depth information combined with light attenuation data from routinely conducted water quality monitoring programs could be used to calculate the percentage of subsurface irradiance available for different seagrass species found in the different bay segments. This information would relate current water quality conditions to the TBEP seagrass restoration goal and serve as a check on the Tampa Bay resource-based management plan (Johansson and Greening 2000). Also, seagrass depth measurements could be used to estimate specific seagrass species light requirements in the major bay segments, and therefore, complement the *T. testudinum* light requirement studies in Lower Tampa Bay (Dixon 2000). Finally, seagrass elevation measurements would also complement the cooperative Tampa Bay permanent seagrass transect monitoring program by providing elevation reference points on the transects (see City of Tampa 2000 and Avery et al. in prep). These reference points could be used for detailed measurements of seagrass elevations and also to measure potential sediment losses or gains along the transects.

The present study evaluated a relatively simple and practical field technique to

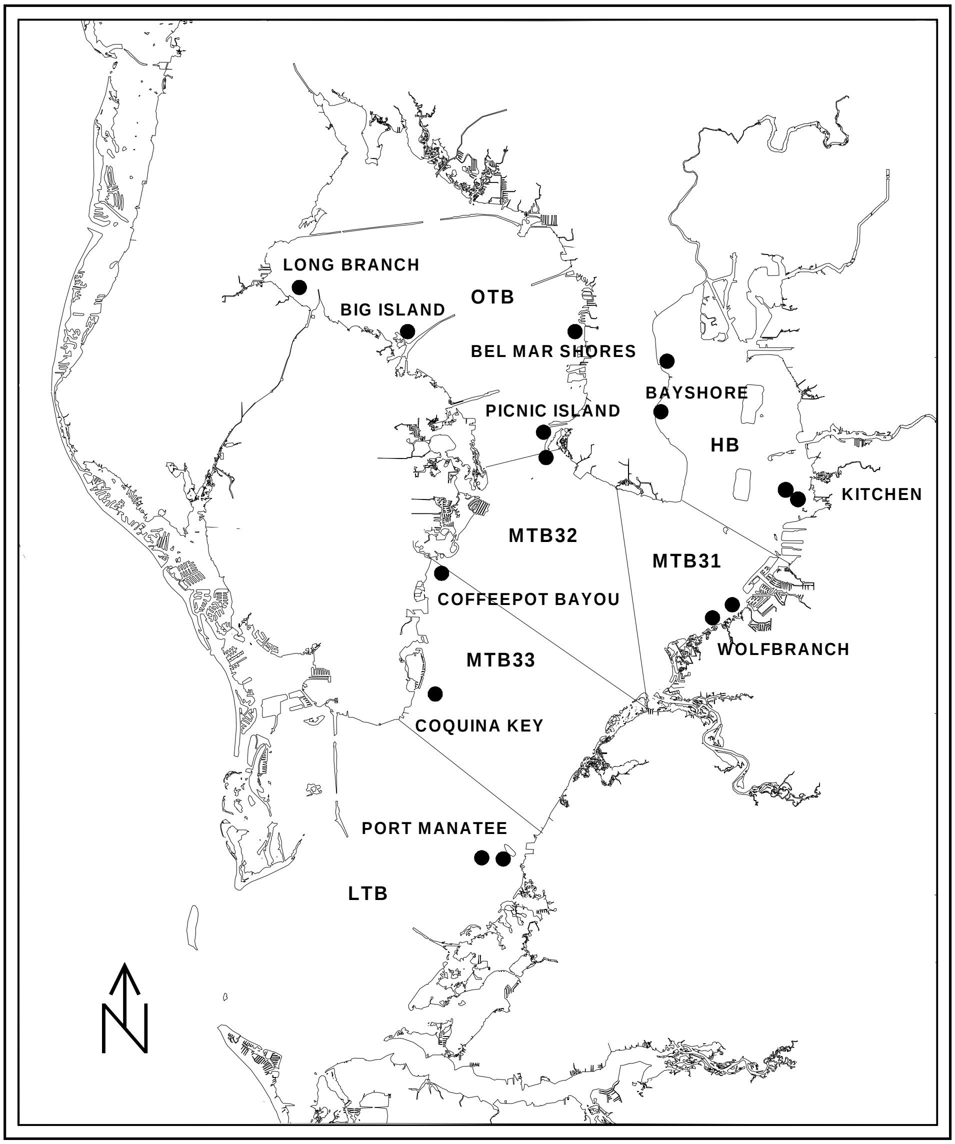

Figure 1. Locations of seagrass elevation survey sites in Tampa Bay. Also shown are major bay segments (HB=Hillsborough Bay; OTB=Old Tampa Bay; MTB=Middle Tampa Bay [including sub-segments]; and LTB=Lower Tampa Bay).

measure the depth to which seagrass meadows extend in Tampa Bay. The technique uses mapping grade differential Global Positioning System (GPS) equipment (Trimble Pathfinder PRO XR) to measure elevations related to a defined tidal datum (MTL). The current cost of the system is approximately $11,000 and several TBEP partners have purchased the system. The study included evaluations of measurement errors and numerous field surveys that measured seagrass elevations in the four major bay segments and for the three major Tampa Bay seagrass species, *Halodule wrightii*, *T. testudinum*, and *Syringodium filiforme.* Further, numerous benchmark locations were inspected and evaluated near the periphery of the bay for suitability as GPS base station locations.

METHODS

Determination of Measurement Errors

Trimble specifications for the GPS Pathfinder PRO XR system with carrier-phase processing reports the accuracy of position determinations, expressed as root mean square error (RMS), as 10cm + 5 ppm with 20 minutes of satellite tracking (occupation time). The 5 ppm error is caused by the distance between the base and the rover stations (baseline) and equals 0.5 cm of error for each kilometer of separation. To achieve 10 cm + 5 ppm accuracy, a minimum of 5 satellites should be tracked. PDOP (position dilution of precision), which is a measure of the current satellite geometry, should be less or equal to 6; the signal to noise ratio, which is a measure of the of the strength of the satellite signal relative to the background noise, should be less or equal to 6; and the satellite elevation mask, which excludes satellites low on the horizon, should be set at 15 degrees. Further, optimal accuracy is obtained by collecting data in an environment that has a clear view of the sky and that is devoid of large reflective surfaces, such as buildings, that extend above the satellite elevation mask.

The Trimble specifications do not differentiate between horizontal and vertical accuracy levels for carrier-phase processing. However, a report that characterizes the accuracy of the Trimble PRO XR receiver (Trimble 1997) states that the vertical error for carrier-phase processing solutions is similar to the horizontal error. The report also shows that the accuracy increases with increasing occupation time. As shown in Figure 2, modified from Trimble (1997), an error of less than 5 cm RMS can be expected with an occupation time of 30 min. For these tests, Trimble used a relatively short baseline (less than 1 km), 5 or more satellites, a maximum PDOP of 4, and the satellite elevation mask set at 15 degrees for the rover station and at 10 degrees for the base station.

Thirty-five tests were conducted over several days on the roof of the City of Tampa Bay Study Group (COT) laboratory to specifically test the vertical measurement performance of the PRO XR system (Fig. 3). This location provided a clear view of the sky and lacked potentially interfering reflective surfaces. Two PRO XR instruments were placed on the roof at a location with a known MTL elevation. The phase centers (the location within the antenna where the receiver detects the GPS signal) of the two antennas were located at near identical elevations and separated less than 1.0 m horizontally. One instrument was used as a base station and the other as a

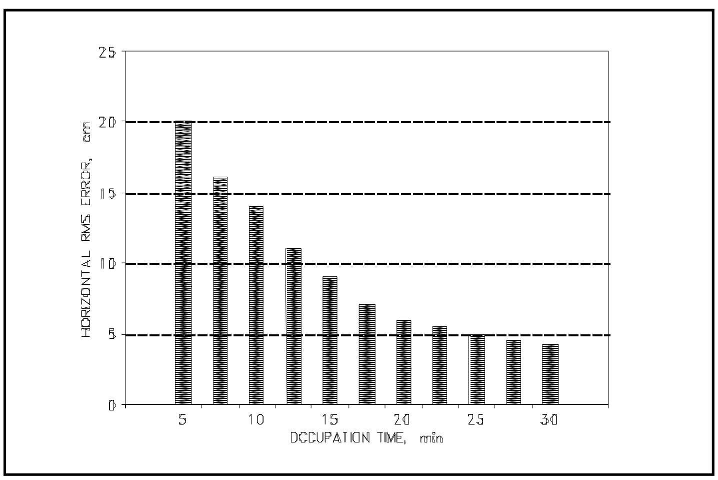

Figure 2. Performance of the Trimble Phase Processor v.2 software with the GPS Pathfinder PRO XR system according to Trimble (1997). Figure modified from Trimble (1997). Horizontal errors are shown; however, Trimble (1997) states that vertical and horizontal errors are similar for phase processed solutions.

Figure 3. Trimble Pathfinder PRO XR instruments located on the roof of the City of Tampa Bay Study Group laboratory during tests of vertical measurement errors.

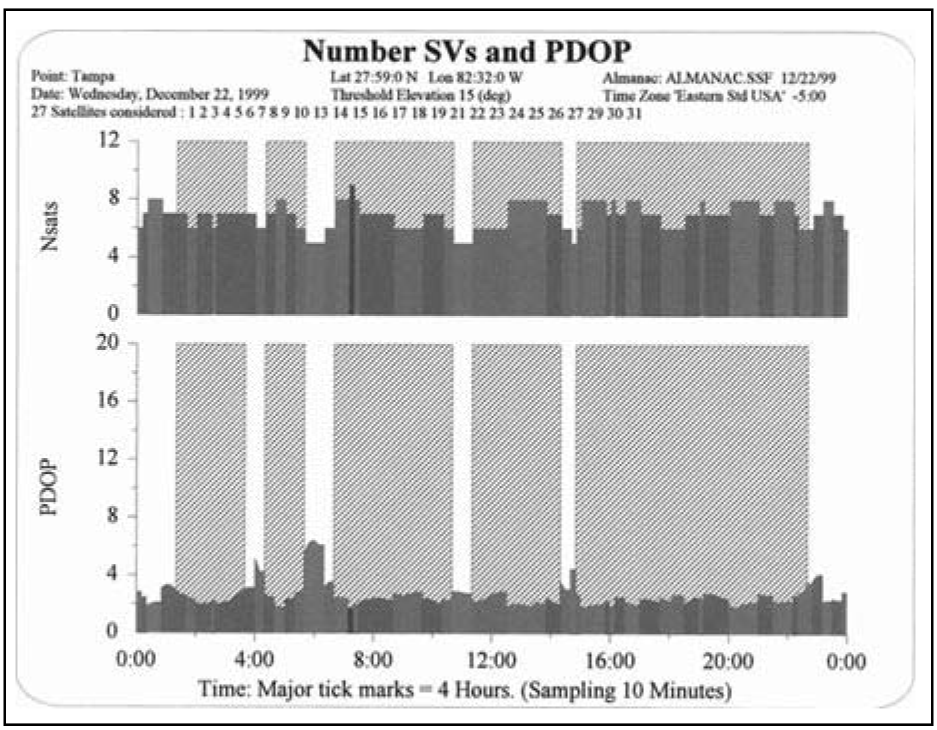

Figure 4 Example of quick plan graph from Trimble Pathfinder Office v.2.1. Excellent data collection windows (minimum of 6 satellites [SV] and maximum PDOP of 3) are shown as hatched bars.

rover station. The instruments were configured to the Trimble recommendations (see above and Trimble 1996). As recommended in the Trimble manual, the base station instrument had the satellite elevation mask set at 10 degrees. Further, predicted daily satellite schedules were examined prior to testing to ensure optimum data collection periods (Fig. 4). Generally, periods with a minimum of 6 available satellites and a PDOP of less than 3 were selected for data collections. These requirements should provide measurements with an accuracy comparable to that reported by Trimble (1997). The satellite data collection period for the 35 tests ranged from 30 to 41 minutes.

The potential baseline errors affecting the seagrass elevation measurements in the current study were not tested specifically (see below). This error was assumed to be 5 ppm, or 0.5 cm for each kilometer of separation between the base and rover stations, as specified by Trimble (1997).

Seagrass elevation measurements were replicated with n>2 at four specific seagrass sites to estimate the variability of field measurements, including variations caused by GPS errors and other errors, such as antenna height measurements. At one of these sites, measurements were repeated on two separate dates with the base station located at two different benchmark locations.

Measurements of Seagrass Elevations

A total of 38 seagrass elevation measurements were performed in Tampa Bay between October 1999 and February 2000 (Table 1). Measurements were conducted at ten general areas in the four major bay segments (Fig. 1). Most study areas were located at, or close to, an established Tampa Bay fixed seagrass transect (see Avery 2000; Avery et al. in prep; City of Tampa 2000) and included different seagrass species when present. Two study areas were located in Hillsborough Bay, four in Old Tampa Bay, three in Middle Tampa Bay, and one in Lower Tampa Bay. Of the 38 measurements, 21 were conducted on *H. wrightii*, 8 on *T. testudinum*, and 9 on *S. filiforme*. Twenty-nine measurements were conducted at distinctive deep edges of either large seagrass areas (meadows) or isolated smaller areas (patches) that were visible on recent, most often 1999, aerial photographs. The remaining nine measurements were done in seagrass areas other than the defined deep edge. These included measurements near the center of *H. wrightii* and *T.*

Table 1. Location of GPS seagrass elevation survey sites in Tampa Bay, including the number of measurements conducted for each surveyed seagrass species (see Fig. 1 for locations of study sites).

LOCATION	BAY SEGMENT	SEAGRASS SPECIES (number of surveys)
Bayshore	HB	*H. wrightii* (4)
Kitchen	HB	*H. wrightii* (9)
Long Branch	OTB	*H. wrightii* (1)
Big Island	OTB	*H. wrightii* (2)
Bel Mar Shores	OTB	*S. filiforme* (1); *T. testudinum* (1)
Picnic Island	OTB	*S. filiforme* (2); *T. testudinum* (1)
Wolf Branch	MTB	*H. wrightii* (4); *T. testudinum* (4)
Coffeepot Bayou	MTB	*S. filiforme* (mixed with sparse *T. testudinum)* (2)
Coquina Key	MTB	*S. filiforme* (2)
Port Manatee	LTB	*H. wrightii* (1); *S. filiforme* (2); *T. testudinum* (2)

testudinum patches; and at the shallow edge of a *H. wrightii* meadow in Hillsborough Bay.

Prior to conducting the field measurements at the selected seagrass areas, suitable benchmarks had to be located, preferably within 5 km of the survey sites in order to minimize the baseline error. Several publications and sources of benchmarks were examined; however, NOS tidal benchmarks were the primary type used (see www.opsd.nos.noaa.gov). The NOS benchmarks are referenced to mean lower low water and mean high water; however, the MTL elevation can easily be calculated from the tide station data provided for each set of benchmarks. The NOS benchmarks are not directly referenced to the National Geodetic Vertical Datum (NGVD)-29 datum, although, several tide stations (e.g. St. Petersburg and Ballast Point) have been tied to NGVD-29. The lack of a direct reference to NGVD-29 for some of the tidal benchmarks used was not of concern since the purpose of the study was to estimate the depth of the water above the seagrass meadows at the MTL.

After the selection of a suitable benchmark location, it was necessary to visit the benchmark site and locate (recover) the specific marker to be used and also to determine that the location was suitable for GPS observations (i.e. a relatively open area with a clear view of the sky and with no large reflective surfaces nearby). Most benchmark locations were not directly useable for GPS observations and a suitable location for the base station had to be marked and offset from the benchmark by using standard level (Carl Zeiss Ni2) and rod surveying techniques. All offset distances were relatively short (<200m) and all level readings were duplicated.

Elevation measurements at the ten selected seagrass study sites (Fig. 1) followed the establishment of base stations. Figure 5 is an aerial photograph of the Kitchen area of Hillsborough Bay that is shown as an example of the seagrass study sites. The photo shows the approximate locations of the seagrass elevation measurements. The specific locations to be measured within each seagrass study site (most often the deep edge of the meadow), were determined in the field by comparing aerial photographs of the area with on-site observations. The majority of the deep edge seagrass study sites had a very distinct and easily defined deep edge of the meadow, but several sites had sparse (low shoot density) seagrass coverage that extended from the edge of the meadow into deeper waters. This sparse seagrass coverage was not considered to be part of the defined meadow.

Typical set-ups of the GPS instruments for measurements of the deep edge of seagrass meadows are illustrated in Figures 6 and 7. The base station was placed with its antenna vertically above the benchmark and the rover station was placed on a tripod above the sea surface with its antenna vertically above the seagrass edge to be measured. As illustrated in Figure 6, the base station antenna height (A), i.e. the distance between the antenna phase center and the center of the benchmark, was measured using a weighted metric tape measure, and recorded. Similarly, the rover station antenna height (C), i.e., the distance between the antenna phase center and the top of the sediment at the seagrass site, was also measured and recorded. The instruments were configured to Trimble recommendations (Trimble 1996) and the daily satellite schedule was examined prior to data collections to ensure optimum data collection periods (see above). Static satellite observations were conducted for a period sufficiently long to ensure that the two stations collected at least 30 minutes of overlapping data (see addendum for an updated and more efficient method of satellite data collections).

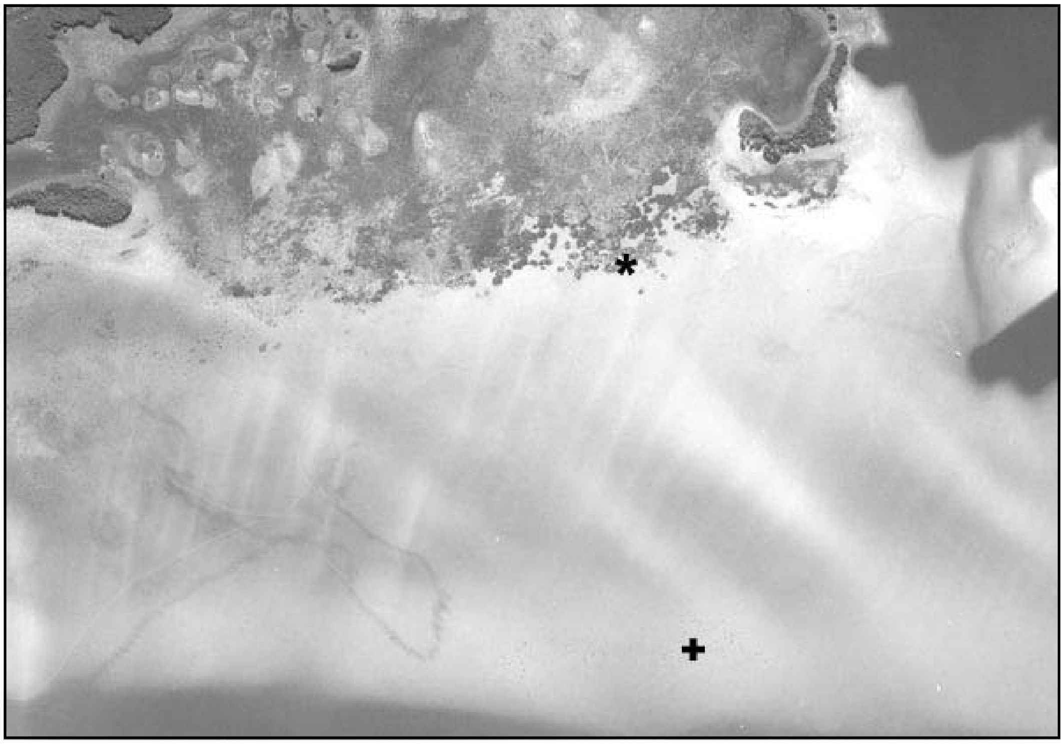

Figure 5. Vertical photograph of the Kitchen area in Hillsborough Bay taken on October 26, 1999. The symbols show the approximate locations of the GPS seagrass elevation survey sites.

The collected satellite data was analyzed using the Trimble software products Pathfinder Office v.2.1 and Phase Processor v.2. The software calculated the relative elevation difference between the two antennas (D in Fig. 6). Since the MTL elevation of the benchmark (B) was known and the antenna heights (A and C) had been measured in the field, the MTL elevation of the deep edge of the seagrass meadow (X) could easily be calculated using the equation shown in Figure 6.

RESULTS

Measurement Errors

Results from the 35 tests conducted on the roof of the laboratory to determine elevation measurement errors of the PRO XR system are shown in Figure 8. As previously discussed, the two instruments were assumed to be at identical elevation during all tests, i.e. the true elevation difference was 0 m. Measured elevation differences ranged between +6.0 cm to -2.7 cm. The average difference of the 35 tests was 0.2 cm (STD 2.1 cm). The 95% confidence interval ranged from 0 to 0.9 cm, suggesting that the confidence interval contains the actual elevation 95% of the time.

The baseline error introduced during these tests was near zero since the two antennas were separated by less than 1.0 m. However, the potential baseline error must be considered during field measurements. Trimble reports this error to be 0.5 cm for each kilometer of separation between the base and rover stations. The baseline distance should, therefore, be kept as short as possible. Baseline distances used during the seagrass elevation study ranged from 0.12 km to 10.6 km, resulting in potential baseline errors ranging from 0.1 cm to 5.3 cm. The average baseline distance of the 38 field measurements was 3.5 km.

Results from the replicated seagrass elevation measurements with n>2 are discussed below.

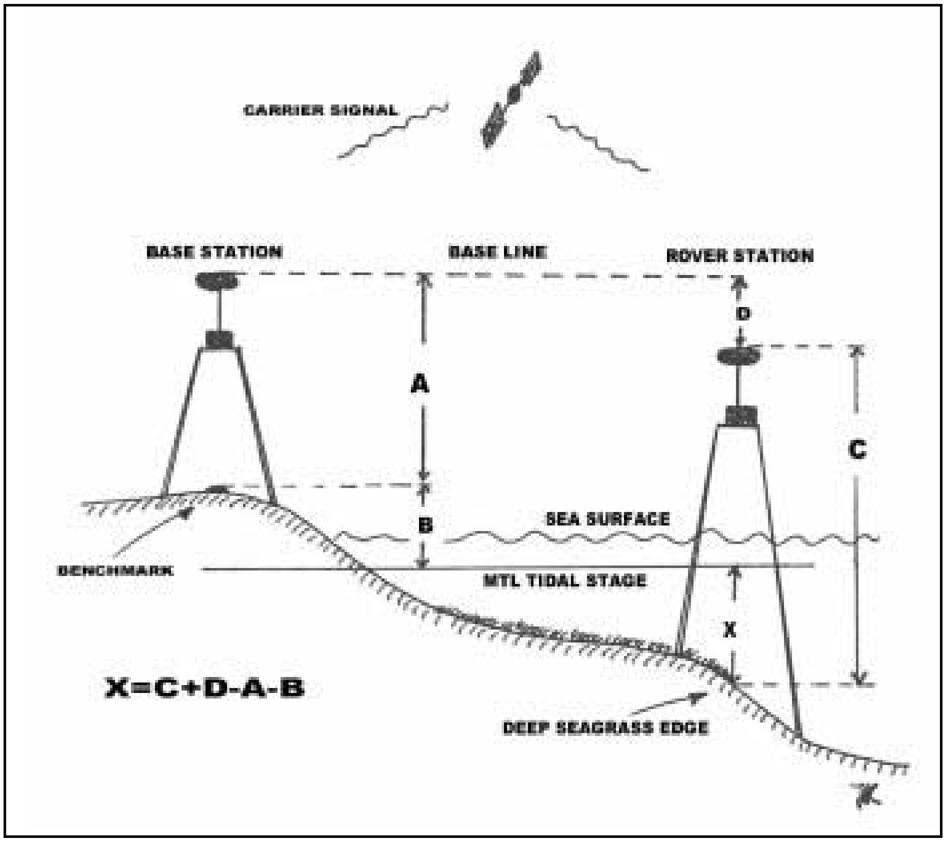

Figure 6. Schematic of typical GPS stations set-up during elevation measurements. A = base station antenna height; B = benchmark elevation above MTL tidal stage; C = rover station antenna height; D = Relative elevation difference between base station and rover station antennas; X = Calculated elevation of the deep seagrass edge.

A B

Figure 7. Field set-ups of base (A) and rover (B) stations during GPS seagrass elevation measurements in Tampa Bay.

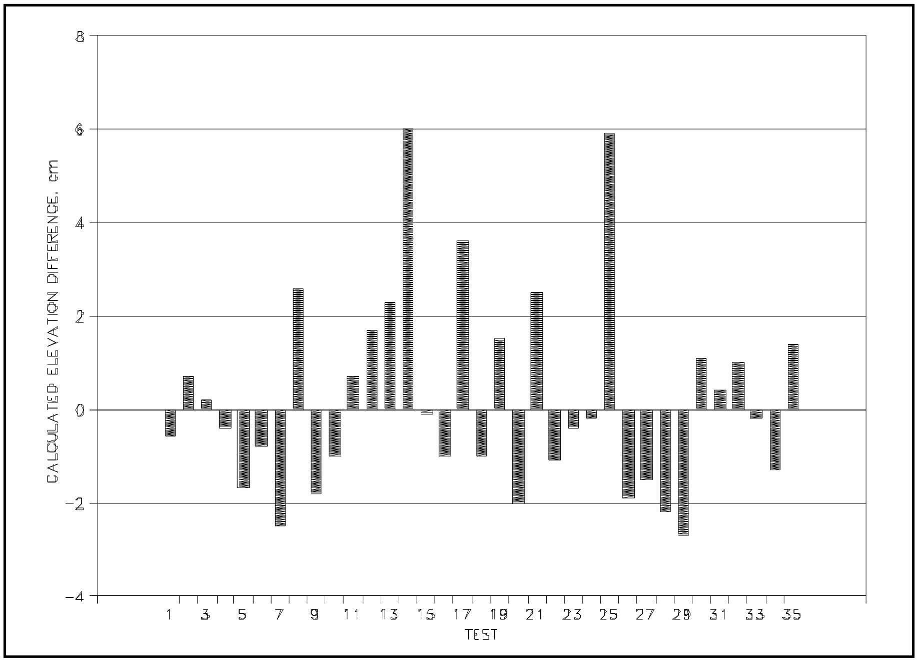

Figure 8. Results from 35 tests of vertical measurement errors conducted on the roof of the City of Tampa Bay Study Group laboratory using two Pathfinder PRO XR instruments. The true elevation difference between the instruments was 0 cm.

Seagrass Elevations

Results from the 38 seagrass elevation measurements at the ten selected seagrass study sites are shown in Table 2. The shallowest deep edge of the *H. wrightii* meadows was in the upper section of Hillsborough Bay (-0.30 to -0.34 m MTL) and at intermediate depths in the lower Hillsborough Bay and the northeastern area of Middle Tampa Bay, just south of Hillsborough Bay (-0.48 to -0.58 m MTL). The deepest *H. wrightii* surveyed was found at similar depths at Big Island in Old Tampa Bay and at Port Manatee in Lower Tampa Bay (-0.71 to -0.76 m MTL). Deep edges of *T. testudinum* meadows occurred at similar depths at Bel Mar Shores in eastern Old Tampa Bay and at Port Manatee. Depths for these edges ranged from -1.53 m MTL at Bel Mar Shores to -1.73 m MTL at Port Manatee. Isolated patches of *T. testudinum* located on the shallow sandbar at Picnic Island and the Wolf Branch area were found at considerably shallower elevations (-0.53 to -0.90 m MTL). Deep edges of *S. filiforme* meadows also occurred at similar depths at the sites in eastern Old Tampa Bay and at Port Manatee. Depths of these edges ranged from -1.19 to -1.46 m MTL. However, the deepest *S. filiforme* edges were measured at the two sites on the western side of Middle Tampa Bay. At Coffeepot Bayou the deep edge was between -1.79 and -1.81 m MTL and at Coquina Key between -1.93 and -1.96 m MTL. The latter depths were the deepest seagrass elevations measured in this study.

Results from the four seagrass sites with replicated (n>2) elevation measurements (Table 2) show that the standard deviation of the determined elevations ranged from 3 to 4 cm. The coefficient of variation for these measurements ranged from 4.1% to 7.7%. At one of these sites, the offshore bar in the Kitchen in southeastern Hillsborough Bay, four measurements were conducted in the center of different *H. wrightii* patches. Two of these measurements were performed with the base station located in Simmons Park, approximately 10.6 km

Table 2. Results of GPS seagrass elevation measurements conducted for different seagrass species in different sections of Tampa Bay. Elevation expressed as mean tide level (MTL).

SEAGRASS SPECIES LOCATION	BAY SEGMENT	NUMBER of SURVEYS	ELEVATION (MTL) (m) Average	Range	STD
***H. WRIGHTII* :**					
Bayshore North; Deep Edge of Meadow	HB	1	-0.30		
Bayshore South; Deep Edge of Meadow	HB	1	-0.34		
Bayshore South; Patch Offshore Bar	HB	1	-0.65		
Bayshore South; Shallow Edge of Meadow	HB	1	-0.20		
Kitchen; Deep Edge of Meadow	HB	5	-0.52	-0.48 to -0.58	0.04
Kitchen; Patches Offshore Bar	HB	4	-0.73	-0.69 to -0.77	0.03
Long Branch; Deep Edge of Meadow	OTB	1	-0.65		
Big Island; Deep Edge of Meadow	OTB	2	-0.74	-0.71 to -0.76	
Wolf Branch; Deep Edge of Meadow	MTB	4	-0.52	-0.50 to -0.57	0.03
Port Manatee; Deep Edge of Meadow	LTB	1	-0.72		
T. TESTUDINUM:					
Bel Mar Shores; Deep Edge of Meadow	OTB	1	-1.53		
Picnic Island; Deep Edge of Patch	OTB	1	-0.90		
Wolf Branch; Deep Edge of Patch	MTB	1	-0.76		
Wolf Branch; Center of Patches	MTB	3	-0.57	-0.53 to -0.59	0.03
Port Manatee; Deep Edge of Meadow	LTB	2	-1.6	-1.54 to -1.73	
S. FILIFORME:					
Bel Mar Shores; Deep Edge of Meadow	OTB	1	-1.42		
Picnic Island; Deep Edge of Meadow	OTB	2	-1.33	-1.19 to -1.46	
Coffeepot Bayou; Deep Edge of Meadow	MTB	2	-1.80	-1.79 to -1.81	
Coquina Key; Deep Edge of Meadow	MTB	2	-1.95	-1.93 to -1.96	
Port Manatee; Deep Edge of Meadow)	LTB	2	-1.2	-1.14 to -1.27	

from the seagrass site. The other two measurements were performed on a different date and with the base station located on Hillsborough Bay spoil island 3-D, approximately 2.7 km from the seagrass site. The seagrass patch elevations based on the Simmons Park benchmark were -0.73 and -0.77 m MTL; elevations based on the 3-D benchmark were -0.69 and -0.73 m MTL.

DISCUSSION

Technique Evaluation

Results from tests of measurement errors conducted by Trimble (Trimble 1997) and the present study suggest that the technique using PRO XR instruments and Phase Processor software will yield seagrass elevation measurements with an error less than ±10 cm for survey sites located up to 10km from benchmark sites.

Further, the field evaluation of the technique, which included measurements of the deep edge of the three major Tampa Bay seagrass species, *H. wrightii*, *T. testudinum*, and *S. filiforme* in the four major bay segments, found the method to be practical. Excellent replication of elevations was obtained when several measurements were taken in the same general area and also when different benchmarks were used.

Seagrass Elevations

First, it should be recognized that the present study was primarily designed to evaluate the GPS carrier-phase processing technique and that seagrass elevation measurements were conducted at a limited number of Tampa Bay seagrass sites. Although deep edge elevation measurements were conducted in all four major Tampa Bay segments and measurements included the three major seagrass species, a much more intensive effort is required before comprehensive conclusions should be formulated about the Tampa Bay seagrass depth distribution. Elevation measurements should be conducted at most, if not all, of the nearly 60 seagrass monitoring transects included in the Tampa Bay cooperative seagrass monitoring program. However, recognizing the limitations of the present study, several interesting findings warrant further discussion.

The deep edge elevations of the measured seagrass meadows ranged from -0.30 m MTL for *H. wrightii* in the upper portion of Hillsborough Bay to -1.96 m MTL for *S. filiforme* near Pinellas Point in Middle Tampa Bay. Further, all sites visited in the present study had deep edge elevations shallower than the TBEP seagrass restoration target depth for the respective bay segment. The greatest deviation from the target depth was found at the Long Branch and Big Island sites in western Old Tampa Bay, where the deep edges of the *H. wrightii* meadows were about 1.30 m shallower than the -2.0 m MTL target depth selected for this bay segment. The least deviation was found at three sites: the *H. wrightii* meadow in the Kitchen in southeastern Hillsborough Bay; the *T. testudinum* meadow at Bel Mar Shores in eastern Old Tampa Bay; and the *S. filiforme* meadow at Coquina Key in southwestern Middle Tampa Bay. These three areas had deep edges that were approximately 0.50 m shallower than the respective bay segment targets.

Similar deep edge depths were found for all three seagrass species at the Old Tampa Bay sites and the Port Manatee site in Lower Tampa Bay. This was surprising, considering the distance of these areas from the mouth of Tampa Bay. The Old Tampa Bay sites are approximately 50 km from the mouth of the bay, while the corresponding distance for the Port Manatee site is only about 20 km. It could be expected that water quality and light attenuation at the Port Manatee site would be superior due to its relative closeness to the Gulf of Mexico, and therefore, would allow seagrass to grow deeper at this site. Analysis of Hillsborough County Environmental Protection Commission (HCEPC) water quality monitoring data, averaged over the last six years, generally supports this hypothesis. Light extinction (Secchi Disk depth), chlorophyll *a* concentrations, and water color were all considerably lower near the Port Manatee site as compared to the Old Tampa Bay sites. However, turbidity was slightly higher near the Port Manatee site.

Additional elevation measurements of Lower Tampa Bay seagrass meadows may find deeper seagrass edges in this bay segment. Dixon (2000) conducted light requirement studies at *T. testudinum* sites in Lower Tampa Bay that ranged in depth from -1.98 to -2.37 m MTL. These depths, which were estimated from sea surface observations, are approximately 0.3 to 0.6 m deeper than the *T. testudinum* meadows surveyed at the Port Manatee site.

Light Availability

Light attenuation measurements of the water column directly above the deep edges of seagrass meadows in Tampa Bay are scarce. Light measurements are most often collected at deeper Tampa Bay sites during routine water quality monitoring. Light attenuation at the seagrass survey sites was therefore estimated from deeper site data. This method was previously used by the TBEP to establish the Tampa Bay seagrass restoration target (Janicki and Wade 1996; also see Giesen et al. 1990). In our study, monthly HCEPC Secchi Disk depths for the period 1994–99 collected near the seagrass elevation survey sites were converted to light attenuation (K_{dPAR}) values using bay segment-specific factors derived from concurrent Secchi Disk depth and PAR measurements by the COT at deep sites for the same six-year period (Table 3 and Fig. 9).

Table 3. HCEPC and COT water quality monitoring stations that were used to estimate the average water column light attenuation at the seagrass elevation survey sites in Tampa Bay for the six year period 1994–99.

HCEPC WATER QUALITY STATIONS	COT SECCHI DEPTH and PAR STATIONS	BAY SEGMENT	SEAGRASS SURVEY LOCATION
6 and 7	4, 12, 17, 18, 19, and 20	HB	Bayshore
73	4, 12, 17, 18, 19, and 20	HB	Kitchen
65	40	OTB	Long Branch
66	40	OTB	Big Island
50 and 51	40	OTB	Bel Mar Shores
33 and 36	40	OTB	Picnic Island
81	13 and 23	MTB	Wolf Branch
32	13 and 23	MTB	Coffeepot Bayou
28	13 and 23	MTB	Coquina Key
90	95	LTB	Port Manatee

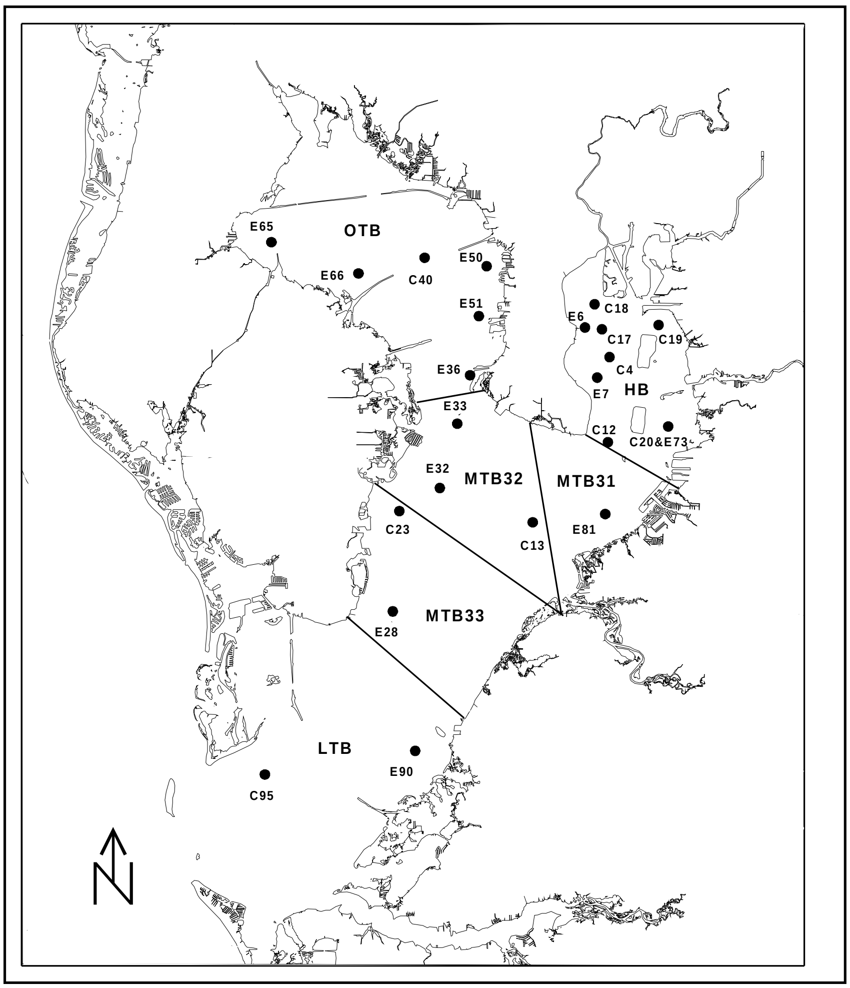

Figure 9. Location of HCEPC water quality monitoring stations (E) located near seagrass elevation survey sites (see Fig. 1). Also shown are COT stations (C) that, since 1994, have concurrently measured PAR attenuation coefficient and Secchi disk depth information.

The percentage of subsurface light remaining at the sediment surface at the deep edge of the seagrass meadows can be estimated from K_{dPAR} and the seagrass elevation measurements using the Lambert-Beer equation:

$$I_Z = I_0 * e^{-kz}$$

where I_Z = the incident light at depth z; I_O = the incident light just below the surface; k = the diffuse PAR light attenuation coefficient; and z = the depth (as m MTL) at I_Z.

The estimated average percent of subsurface incident light available at the deep edges of the seagrass meadows over the six-year period 1994–99 for the different seagrass survey sites and seagrass species is shown in Table 4. The available light at the deep edges of *H. wrightii* meadows in all four bay segments ranged from 59.8% to 28.9% of subsurface incident light and was substantially above the adopted TBEP seagrass restoration light target of 20.5%. Deep edges of *T. testudinum* at Bel Mar Shores in Old Tampa Bay and Port Manatee in Lower Tampa Bay appeared to receive less light than the target, 19.0% to 16.9%. The deep edges of *S. filiforme* meadows at Coquina Key and Coffeepot Bayou in Middle Tampa Bay received the least amount of light of all study sites, 16.7% and 16.2%, respectively.

As discussed above, the estimated light availability at the deep edge of the seagrass meadows was calculated from Secchi disk depth and PAR light attenuation data from the HCEPC and COT routinely conducted water quality monitoring programs at deep

Table 4. Estimated percentage of annual average subsurface irradiance (PAR) remaining at the sediment surface at the deep edge of seagrass meadows (or patches in areas lacking larger meadows) in Tampa Bay. The light attenuation coefficient (K_{DPAR}), used to calculate subsurface irradiance (%PAR), was estimated from the average 1994–99 Secchi disk depth at HCEPC water quality monitoring stations located near seagrass study sites and the COT bay segment specific light attenuation measurements for the same period.

SEAGRASS SPECIES LOCATION	BAY SEGMENT	ANNUAL AVERAGE ATTENUATION COEFFICIENT (K_{PAR}) (m^{-1})	PERCENT OF SUBSURFACE PAR REMAINING AT SEDIMENT SURFACE (%PAR) Average	Range
***H. WRIGHTII*:**				
Bayshore North	HB	-1.72	59.8	
Bayshore South	HB	-1.72	55.4	
Kitchen	HB	-1.59	44.0	39.9 to 46.8
Long Branch	OTB	-1.91	28.9	
Big Island	OTB	-1.54	32.3	30.9 to 33.7
Wolf Branch	MTB	-1.13	55.6	52.4 to 57.1
Port Manatee	LTB	-1.02	47.8	
T. TESTUDINUM:				
Bel Mar Shores	OTB	-1.16	16.9	
Picnic Island (patch)	OTB	-1.10	37.1	
Wolf Branch (patch)	MTB	-1.13	42.5	
Port Manatee	LTB	-1.02	19.0	17.1 to 20.9
S. FILIFORME:				
Bel Mar Shores	OTB	-1.16	19.2	
Picnic Island	OTB	-1.10	23.6	20.0 to 27.1
Coffeepot Bayou	MTB	-1.01	16.2	16.0 to 16.4
Coquina Key	MTB	-0.92	16.7	16.5 to 16.9
Port Manatee	LTB	-1.02	29.4	27.3 to 31.4

water sites. A limited amount of water quality information is available for the shallow nearshore areas in Tampa Bay that can be used to evaluate the assumption that water quality of the shallow areas is similar to the deep areas. The COT has measured chlorophyll *a* and turbidity at five sites located on the nearshore sandbars in Hillsborough Bay on a monthly schedule since 1995. Three of these sites are located near deeper water quality monitoring stations. A comparison between the shallow and deeper sites showed no consistent difference in chlorophyll *a* concentrations. Turbidity, on the other hand, was often higher and more variable at the shallow sites. Turbidity peaks in the shallow areas were often associated with strong wind events. The limited comparison from Hillsborough Bay suggests that the shallow and deeper water column light climate may at times be substantially different. Therefore, the use of water quality data from deep sites for estimating water column light attenuation at the seagrass meadows needs to be evaluated further by additional deep and shallow water quality comparisons.

The average percent of subsurface incident light available at the deep edges of the seagrass meadows shown in Table 4 may not correspond to the minimum light requirement for maintaining sustained growth of the different Tampa Bay seagrass species. Determination of minimum light requirements for Tampa Bay seagrass species was beyond the scope of this study. Additional work is required to resolve uncertainties about extrapolating light availability data to seagrass light requirements. These uncertainties include, but are not limited to:

1. Light attenuation of the water column over the seagrass meadows may be different than that estimated from deep water data (see above).
2. The time period (six years) selected for calculating the average light attenuation of the water column above the seagrass meadow in this study may not properly reflect the lag-time of seagrass growth response to changes in light availability. The time-lag may be shorter or longer.
3. Seasonal light availability, specifically during the active seagrass growing season, may be more appropriate for estimating minimum seagrass light requirements than annual averaged values.
4. Epiphytic growth on the seagrass blades may have caused additional reductions in light availability.

Recommendations for Future Studies

Recently, seagrass recovery has stagnated in several areas of Tampa Bay, despite ambient water quality and light availability conditions that appear adequate to support continued seagrass expansion. As shown above, the deep edges of the *H. wrightii* meadows in the Kitchen in southeastern Hillsborough Bay and the Wolf Branch area in eastern Middle Tampa Bay were estimated to receive an average 44% and 57% of the incident light, respectively. These light levels are considerably greater than the 20.5% light target adopted by the TBEP; however, no expansion of these meadows into deeper water have occurred over the last three to four years.

Many factors may limit seagrass expansion in Tampa Bay in addition to water quality. Lewis et al. (1985) discussed the importance of an offshore unvegetated sandbar that separates the main seagrass meadow from the open bay waters, to protect the seagrass meadow by reducing wave impacts from storms and ship traffic. Destabilization and the ultimate loss of the bar may result in the shoreward migration of the seagrass meadow. However, studies examining the dynamics of the shallow sand bars and their interaction with the development of seagrass meadows are lacking for Tampa Bay.

Additional elevation measurements are recommended to learn more about the seagrass depth distribution and the dynamics of the shallow sandbars in Tampa Bay. The GPS carrier-phase processing technique could be used at most, if not all, of the 60+ baywide seagrass monitoring transects included in the cooperative Tampa Bay seagrass monitoring program to accurately and quickly determine the transect depth profiles (see addendum). Further, deep edge elevation measurements for the different seagrass species found on each transect could easily be included during the depth profile measurements.

The proposed periodically conducted elevation measurements will provide important information to complement the biennial high altitude aerial seagrass photography conducted by SWFWMD and the annual cooperative Tampa Bay seagrass transect monitoring program. Combined, the three programs would become a powerful tool for evaluating the progress of the Tampa Bay water quality and seagrass restoration effort.

CONCLUSIONS

Evaluations of measurement errors suggest that the GPS carrier-phase processing technique will yield seagrass elevation measurements with an error less than ±10 cm for survey sites located up to 10 km from benchmark sites. Further, repetitive elevation measurements (n>2) conducted at four specific seagrass areas resulted in a standard deviation of the determined elevations that ranged from 3 to 4 cm and a coefficient of variation that ranged from 4.1% to 7.7%.

Elevation measurements at ten Tampa Bay seagrass study sites found relatively shallow deep edges of *H. wrightii* meadows in the upper section of Hillsborough Bay (-0.30 to -0.34 m MTL) and at intermediate depths in the lower Hillsborough Bay and at the Wolf Branch area in northeastern Middle Tampa Bay, just south of Hillsborough Bay (-0.48 to -0.58 m MTL). The deepest *H. wrightii* surveyed was at Big Island in western Old Tampa Bay and at Port Manatee in eastern Lower Tampa Bay (-0.71 to -0.76 m MTL). Deep edges of *T. testudinum* meadows ranged from -1.53 m MTL at Bel Mar Shores in eastern Old Tampa Bay to -1.73 m MTL at Port Manatee. Isolated patches of *T. testudinum* located on the shallow sandbar at Picnic Island in southeastern Old Tampa Bay and the Wolf Branch area were at considerably shallower elevations (-0.53 to -0.90 m MTL). Deep edges of *S. filiforme* meadows in eastern Old Tampa Bay and Port Manatee ranged from -1.19 to -1.46 m MTL. However, the deepest *S. filiforme* edges were found outside the well-developed offshore sandbars at Coffeepot Bayou and Coquina Key on the western side of Middle Tampa Bay. The depth of these edges ranged between -1.79 and -1.81 m MTL at Coffeepot Bayou and between -1.93 and -1.96 m MTL at Coquina Key. The latter measurements were the deepest seagrass elevations recorded in this study.

All survey sites had deep edge elevations shallower than the TBEP seagrass restoration target depth for the respective bay segment. The greatest deviation from the target depth was found in western Old Tampa Bay, where the deep edges of the *H. wrightii* meadows were about 1.30 m shallower than the -2.0 m MTL target depth selected for this bay segment. The least deviation was found at three sites: the *H. wrightii* meadow in southeastern Hillsborough Bay; the *T. testudinum* meadow at Bel Mar Shores in eastern Old Tampa Bay; and the *S. filiforme* meadow at Coquina Key in southwestern Middle Tampa Bay. These three areas had deep edges that were approximately 0.50 m shallower than the respective bay segment targets.

The average percent of subsurface incident light available at the deep edges of *H. wrightii* meadows ranged from 59.8% to 28.9% and was substantially above the adopted TBEP seagrass restoration light target of 20.5%. Deep edges of *T. testudinum* at Bel Mar Shores and Port Manatee appeared to receive less light than the target (19.0% to 16.9%). The deep edges of *S. filiforme* meadows at Coquina Key and Coffeepot Bayou received the least amount of light of all study sites, 16.7% and 16.2%, respectively.

The field evaluation of the GPS carrier-phase processing technique provided an important first step in understanding the current depth distribution of the major Tampa Bay seagrass species. However, many more elevation measurements should be conducted to yield a more complete understanding of the seagrass depth distribution in the bay.

Recently, seagrass recovery has stagnated in several areas of Tampa Bay, despite ambient water quality and light availability conditions that appear adequate to support continued seagrass expansion. One theory proposed for the poor expansion focuses on the importance of the offshore unvegetated sandbar to protect the main seagrass meadow from wave action and to allow seagrass to expand into deeper waters. However, studies examining the dynamics of the shallow sand bars and their interaction with the development of seagrass meadows are lacking for Tampa Bay.

Additional elevation measurements are recommended to learn more about the seagrass depth distribution and the dynamics of the shallow sand bars in Tampa Bay. The GPS carrier-phase processing technique could be used at most, if not all, of the 60+ bay-wide seagrass monitoring transects included in the cooperative Tampa Bay seagrass monitoring program to accurately and quickly determine the transect depth profiles. Further, deep edge elevation measurements for the different seagrass species found on each transect could easily be included during the depth profile measurements.

The proposed elevation measurements will provide important information to compliment the biennial high altitude aerial seagrass photography conducted by SWFWMD and the annual cooperative bay-wide seagrass transect monitoring program conducted by the TBEP partners. Combined, the three programs would become a powerful tool for evaluating the progress of the Tampa Bay water quality and seagrass restoration effort.

ADDENDUM

Trimble recently distributed an upgraded version of the Pathfinder Office software (version 2.70), which includes software that calculates “Post-Processed Kinematic GPS” solutions. The upgraded software, in contrast to that used for processing the data in the current study (Phase Processor v.2), does not require that the GPS rover receiver remains static during the satellite data recording period. Horizontal and vertical positions can therefore be collected “on-the-fly”, which allows for much more productive field surveys. For example, instead of obtaining a single elevation measurement during a static 30-minute data collection period (as used in the current study), the new technique can provide 360 measurements (with a sampling interval of 5 s) during the same time period.

The “on-the-fly” technique is currently being tested for measurement errors by the COT. However, preliminary results agree with Trimble specifications, which state that the kinematic method is as accurate as the static method.

The greatly increased number of data points that can be collected in the field with the "on-the-fly" method will allow for much more efficient and productive seagrass elevation studies, as well as for other studies requiring highly accurate vertical and/or horizontal position information. For example, seagrass species zonation with depth and elevation profiles of the permanent seagrass transects can easily and quickly be determined.

ACKNOWLEDGMENTS

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ON DEFINING THE "EDGE" OF A SEAGRASS BED

R. Virnstein, W. Avery, J.O.R. Johansson

Need for a definition of the "edge" of a seagrass bed:

The edge is the basis for:

1. Total acreage of seagrass.
2. Change detection, both by ground-based transects and mapping from aerial photos.
3. Seagrass restoration targets; assessments of progress may be based on a comparison of the deep edge of grass beds to the target depth.

Two guidelines emerge for determining the edge:

1. Determining the edge should be definable and repeatable.
2. An operational definition is OK (as opposed to an absolute or ecological definition).

The edge definition ideally should be the same whether determined: (a) from swimming in the water, or, assuming good water clarity; (b) as seen from a boat, or; (c) from a plane — by aerial photos. The edge definition should also correspond to edges of target coverages.

Even though very sparse grass may indicate that grass can grow in that area, three principles suggest that areas of very sparse grass can be excluded from the operational definition of a bed:

1. Very sparse grass probably offers little habitat and ecological value. (But it sure would be valuable to have hard data.)
2. Very sparse grass doesn't fit the image of a "bed." We ought to be able to convince others that density is sufficient to call it a "bed." Eventually, we should have some photos to illustrate edges, both where beds become patchy and where they become sparse.
3. Unless we keep track of individual patches, change detection is difficult in very sparse grass.

In order to include areas (a) that meet the definition of a "bed" and (b) that include areas with some grass but not enough to be classified as a "bed" (many biologists can't stand to categorize an area with even a small amount of grass as zero), two categories of "edge" are suggested:

1. **Bed:** The edge of a definite bed, about >10% visual cover that is mapable. This category of "bed" would include what most of us think of as a bed that has enough grass to provide a reasonable level of ecological value. Perhaps it would usually be easy to recognize, if not at a single point along a line transect, but rather as part of a smoothed contour, perhaps looking 10 m or so to the side of a line transect and 10–30 m past this point. Yes, this category would exclude very sparse grass or patches scattered here and there (thus category #2).

2. **Zones of seagrass occurrence:** The zones of wherever some seagrass is present (perhaps down to some limit of 1 shoot per m^2 for at least 10 m, or something like that). This edge would be fuzzy and more difficult to quantify, but at least it would give some idea of the zone in which some seagrass is present. We might think that is important in trying to predict and understand bed expansion or contraction. A little bit could expand into a denser bed the next year. At least we would know then that it did not come from zero. Conversely, it may indicate the last remains of a declining bed that next year might disappear completely.

Thus the two definitions of edge:

"Bed" = ≥10%, visual estimate, within 10–30 m along transect line.

"Zones of seagrass occurrence" = <"bed" but >1 shoot m^2 for at least 10 m. There may be a second zone of widely-spaced patches (but a lower limit would still be needed).

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USE OF A WAVE EXPOSURE TECHNIQUE FOR PREDICTING DISTRIBUTION AND ECOLOGICAL CHARACTERISTICS OF SEAGRASS ECOSYSTEMS

Bradley D. Robbins, Mark S. Fonseca, Paula Whitfield, Pat Clinton

INTRODUCTION

The development of seagrass patches and subsequently, the landscape that they define, is typically thought to be dependent on the phenotypic response of seagrass to environmental factors such as light/ nutrients (Dennison et al. 1993), sedimentation (Harlin and Thorne-Miller 1981), and disturbance (Fletcher and Fletcher 1995). Hydrodynamic forces have also been shown to have an impact on the structure and function of aquatic macrophyte habitats (see review by Fonseca 1996). Within estuaries, the marked developmental differences in seagrass patches have been linked to water movement either via tidal currents (Fonseca *et al.*1983) or wind-generated waves (Fonseca and Bell 1998) with the recognition that while hydrodynamics modify seagrass, seagrasses in turn, modify hydrodynamics (Fonseca and Cahalan 1992). Therefore, by modeling hydrodynamics our understanding of seagrass distribution within estuaries should be enhanced.

MODEL HISTORY AND DEVELOPMENT

In 1996, Fonseca began to develop a model (*sensu* Keddy 1982) that describes the impact of wind-generated waves on the distribution of subtidal seagrass patches within an estuarine landscape. The model, developed using the advanced macro language (AML) associated with ARC/ INFO, first calculates effective fetch (direct fetch weighted by shoreline shape using 9 cardinal directions in 11.25^o increments) from the shoreline to each point defined by a regularly spaced grid or a user supplied point coverage for each of the 8 major compass headings. These data are then combined with meteorological data to estimate a Relative Exposure Index (REI) using measures of effective fetch (*F*), wind speed (*V*; exceedance winds defined as the top 5% of wind events; measured 10 m above mean sea level), and wind duration (*P*):

$$REI = \sum_{i=1}^{8} (V_i \times P_i \times F_i)$$

The model may be applied to an entire water body using an existing shoreline coverage or a subset of the water body using any smaller study area (*e.g.*, point sample) within the water body.

Phase I—Model Validation

Using data from Core and Back Sounds, Carteret County, NC (Figure 1) Fonseca and Bell (1998) conducted a validation exercise to test whether the model could predict seagrass cover. Eighteen 2500 m^2 areas were mapped *in situ* by recording the presence and absence of seagrass (*i.e.*, *Zostera marina* and *Halodule wrightii*) on each 1 m^2 intersection across a 2500 m^2 site. Water depths (corrected for tide height) were recorded across each site using a coarser grid consisting of 1 x 3 m cells. Seagrass physical characteristics (shoot density and biomass, above- and below-ground) were determined by species from random cores within each site. Current velocity was recorded at each site over a rising spring tide by tracking the movement of an introduced dye across a fixed distance. The REI at each study site was then computed using the REI algorithm. Stepwise multiple regression and Principal Components Analysis were used to relate the ecological attributes (seagrass biomass, shoot density, and sediment characteristics) of each site with the site's physical

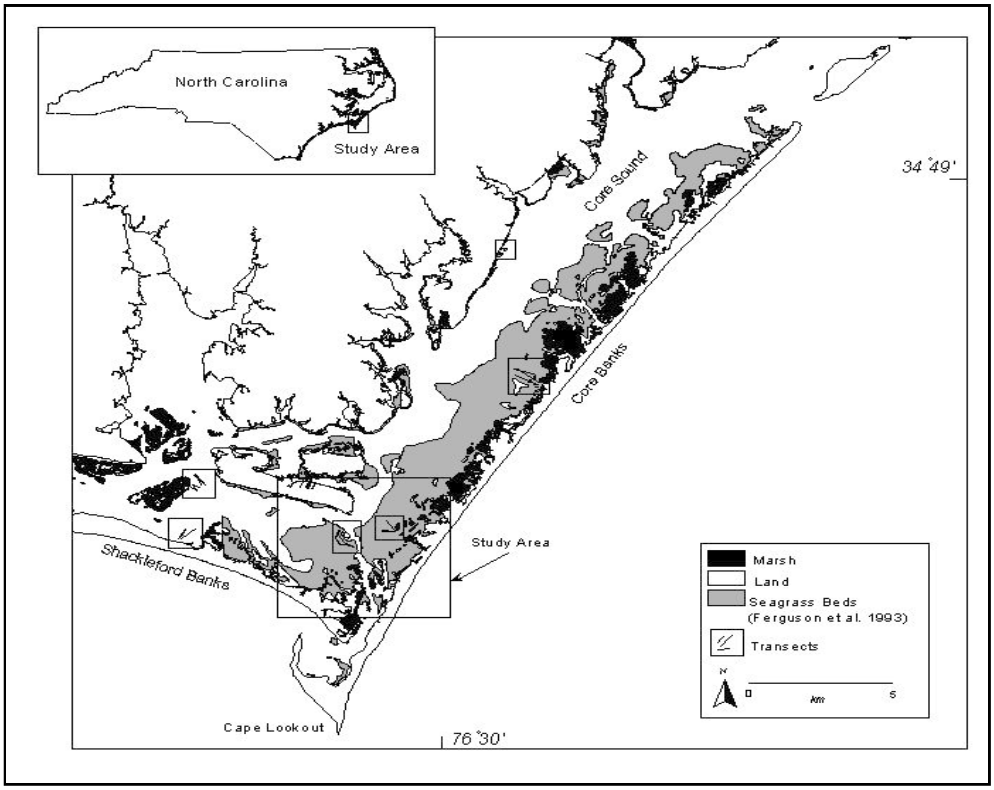

Figure 1. Site map of Core and Back Sounds, Carteret County, NC (34°40' to 34°50'N; 76°20' to 76°40'W). Map indicates the 18 study areas.

setting (REI, tidal current speed, and water depth). These physical factors explained substantial amounts of the variation of the dependent variables (see Fonseca and Bell 1998). The success of this study prompted a second validation study.

Phase II—Model Validation Part 2

In order to determine the utility of the model, seagrass and environmental data were collected from both Chesapeake and Chincoteague Bays (Figure 2). A similar analysis as that performed on the North Carolina data accounted for little of the variation among sites. This was attributed to upland erosion changing the characteristics of the adjacent study sites and the presence of large shoals that diminished wave effects (and thus, REI) more than what would be predicted using simple fetch measurements (Koch et al. *in press*). These results illustrated the need to develop a process with spatial sensitivity to the benthic topology upwind of our sites.

Phase III—A New Iteration

To develop and test the model further, we have applied it to Yaquina Bay, a small drowned river estuary located on the outer coast of Oregon (Figure 3). This estuary differs from the eastern estuaries previously modeled in that it is dominated by winter floods and by summer oceanic tides. The estuary is also much smaller than the eastern estuaries with a maximum fetch of less than 6000 m. Two species of seagrass, the native *Zostera marina* and its exotic congener *Z. japonica* are found within the estuary with *Z. japonica* typically limited to the upper intertidal zone and *Z. marina* found from the mid-intertidal to the subtidal zone—a situation very much like that found in the North Carolina studies where *H. wrightii* occupies shallower areas and *Z. marina* has a more subtidal distribution.

To address the problems found during the second validation exercise, the model has

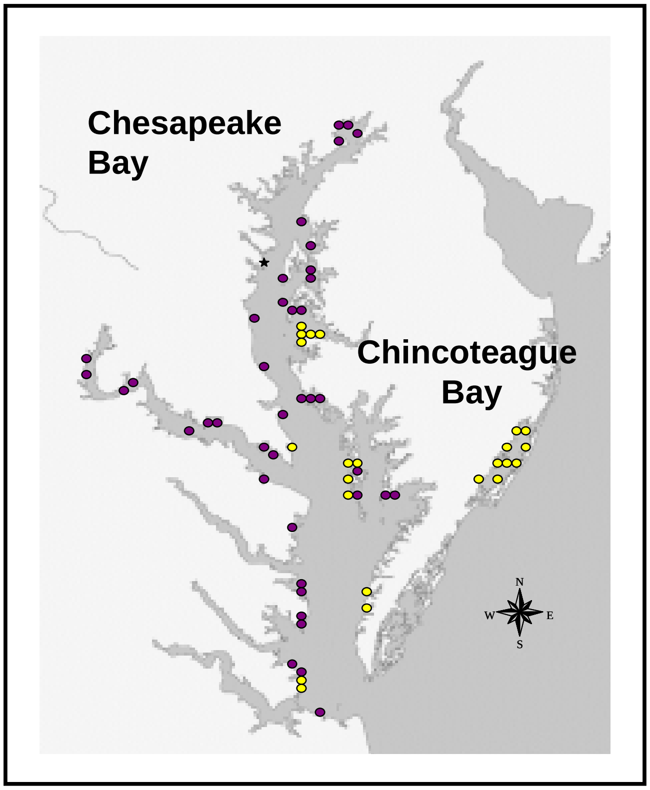

Figure 2. Site map of Chesapeake and Chincoteague Bays, Maryland (37°00' to 40°00'N; 76°00' to 77°30'W).

gone through several iterations to incorporate not only effective fetch, wind speed and wind duration, but also bathymetry (Figure 4). To achieve this an inverse distance weighted procedure (Davis 1996) incorporating bathymetric topology is used to calculate bathymetrically-weighted effective fetch (*idwF*). A further refinement has been achieved by weighting idwF using tidal emersion (*T*) duration at each sampling site.

$$REI = \sum_{i=1}^{8} [(V_i \times P_i \times (idwF_i \times T_i)]$$

Although the process of statistically comparing the modeled output with the digitized image has not yet been completed, a visual comparison of submerged aquatic vegetation (SAV) distribution digitized from 1997 orthorectified color infrared aerial photographs (see Young et al. 1999) and SAV distribution portrayed by our model (Figure 5) suggests that this iteration of the model holds promise.

CONCLUSIONS

We have developed a spatial model that incorporates the environmental factors that may influence the development and maintenance of seagrass landscapes. This model provides a parsimonious vehicle that will enable researchers to both hindcast and forecast trends in seagrass landscape structure and function. Our goal was to develop products that predict: 1) the probability of seagrass habitat cover; 2) the probability of seagrass habitat lost to acute storm events; and 3) probable sites for regrowth given some level of disturbance (*e.g.* restoration). Each of these products contains explicit information required for managers that could not be derived from a traditional mapping exerciese. With this analytical tool we can now begin to predict the kind of seagrass habitats that may

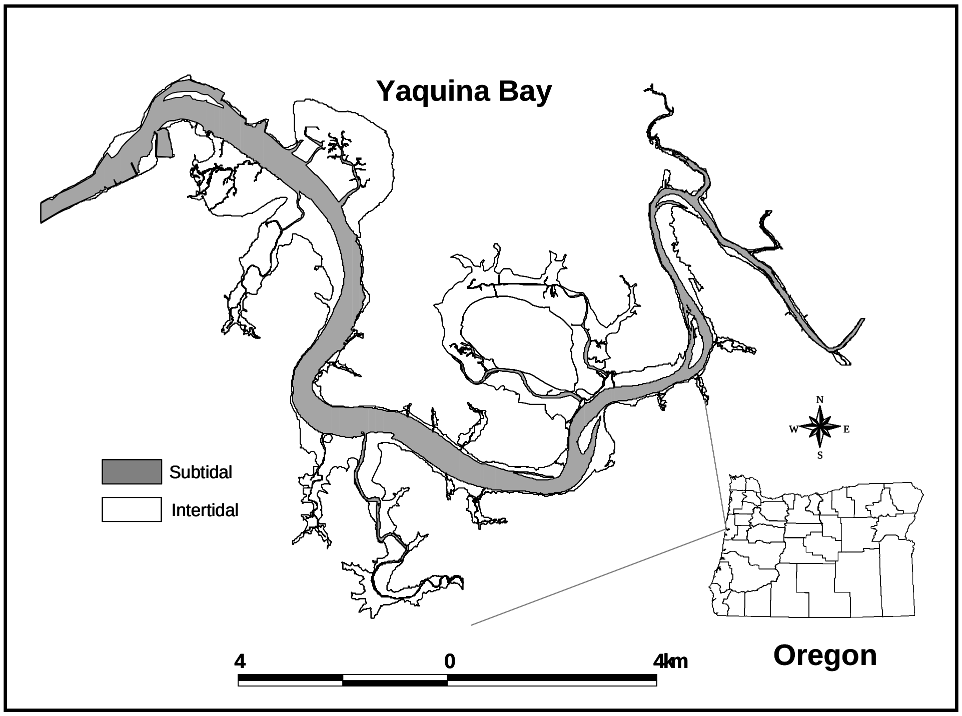

Figure 3. Yaquina Bay, a small estuary located on the outer coast of Oregon (44°34' to 44°38'N; 123°54' to 124°04'W).

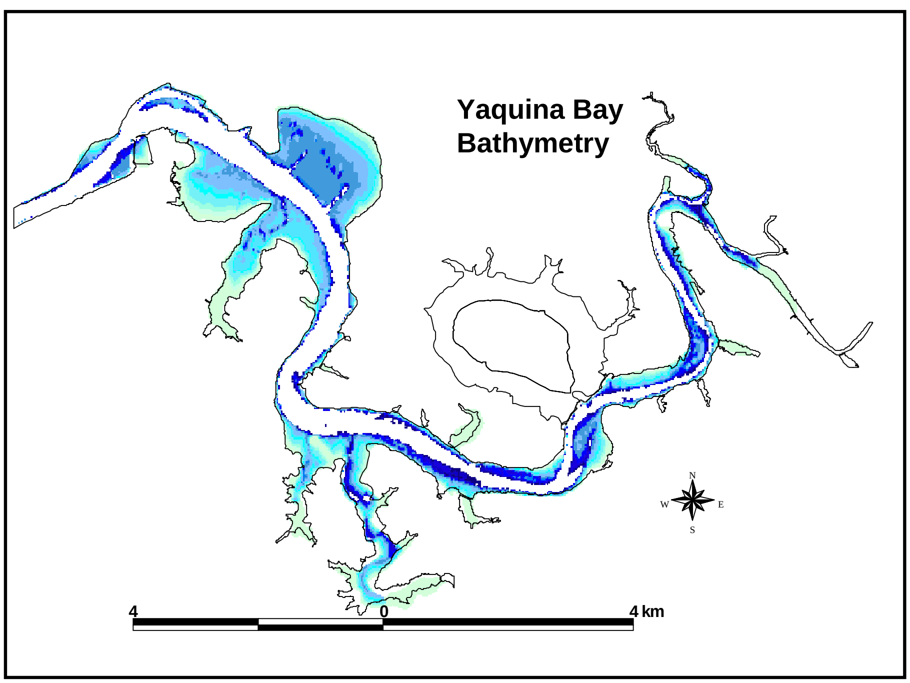

Figure 4. Yaquina Bay bathymetry with depths of the intertidal region shown in shades of gray (dark = deep; light = shallow).

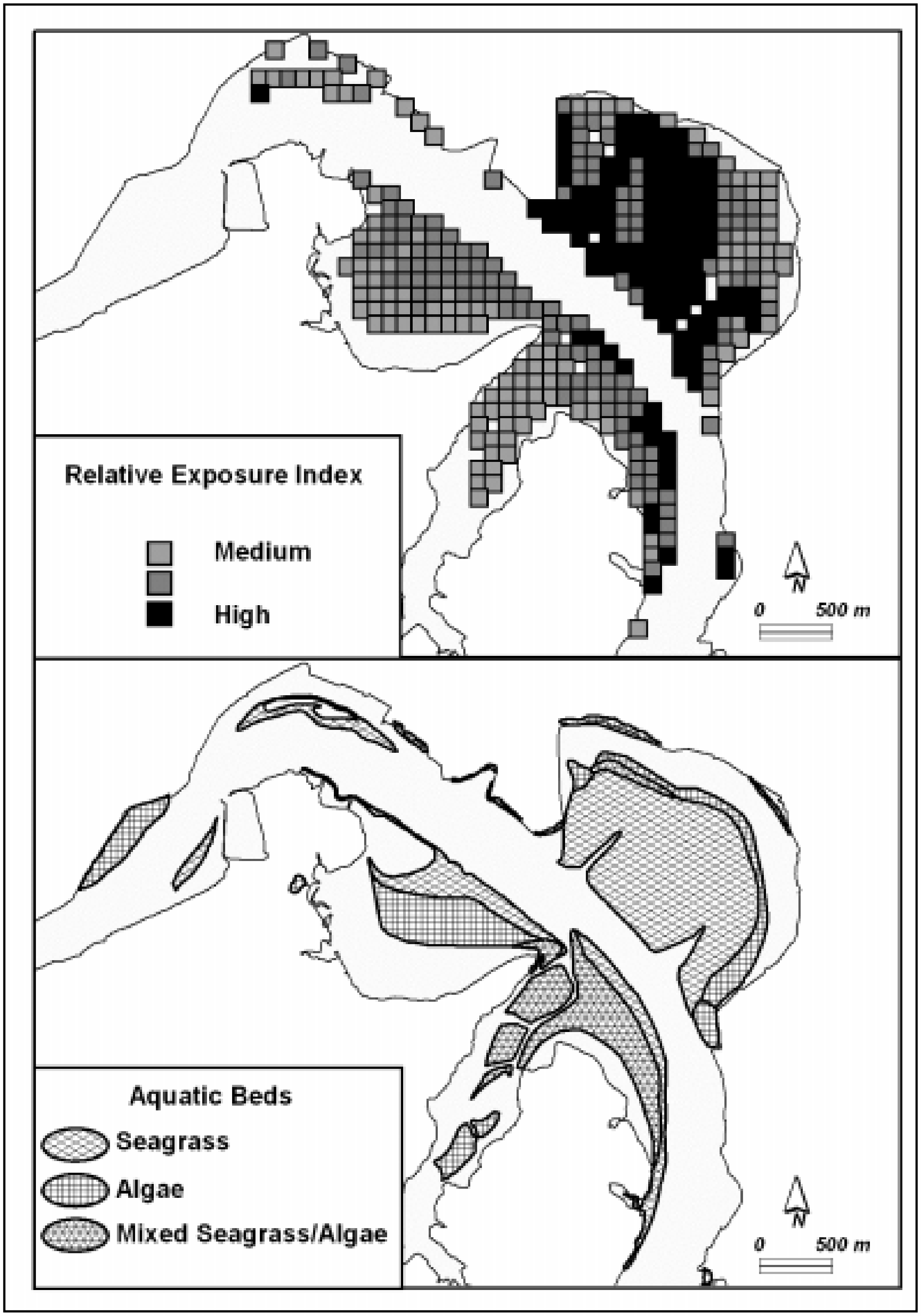

Figure 5. A visual comparison of Yaquina Bay SAV (submerged aquatic vegetation) distribution digitized from 1997 orthorectified color infrared aerial photographs (see Young et al. 1999) and SAV distribution portrayed by the REI model.

develop in the area as the result of restoration, their faunal components, and through hindcasting of storm event data, the susceptibility of these beds and the shoreline property they protect to storm events, particularly hurricanes.

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THE POTENTIAL IMPORTANCE OF THE LONGSHORE BAR SYSTEM TO THE PERSISTENCE AND RESTORATION OF TAMPA BAY SEAGRASS MEADOWS

Roy R. (Robin) Lewis III

ABSTRACT

Tampa Bay seagrass meadows exhibit bathymetric relief that normally includes an unvegetated offshore sandbar, flanked by seagrasses and oriented parallel to the shoreline in most cases. This is called a longshore bar. Historical changes in the distribution and coverage of seagrass in the bay may be associated with changes in the longshore bar system. Recent management initiatives related to increasing seagrass cover in the bay have not recognized the potential importance of this physical feature. It is suggested that maintenance and restoration of these bars in locations where they currently exist or existed in the past may be essential to meet the currently established management goal of maintaining and restoring seagrasses in the bay to 15,358 ha.

INTRODUCTION

The recent increase in seagrass meadow coverage in Tampa Bay, first documented by Johansson and Lewis (1992; see also Lewis et al. 1998), has in recent years slowed its rate of increase, peaking at 10,893 ha in 1996, and then declining to 10,053 ha in 1999 (Tomasko, personal communication). While this decline may fall within the normal year-to-year variation in seagrass cover, it does signal the cessation of what was hoped to be a rapid recovery of seagrass coverage to 1950s levels (15,358 ha). This leveling-off may indicate that natural recovery due to improved water quality *alone* may not be able to achieve the goals established by the Tampa Bay Estuary Program (Janicki et al. 1995; Tampa Bay National Estuary Program 1996).

Physical stress (i.e., "the hydrodynamic setting" *sensu* Fonseca and Bell 1998) on seagrasses and the resulting patterns of patchy seagrass versus dense meadows, or the complete absence of seagrass, were not factored into the original model for predicted seagrass recovery in Tampa Bay (Janicki et al. 1995). The relative wave-exposure index (REI) described by Fonseca and Bell (1998) may have important implications for Tampa Bay seagrasses. The presence or absence of the previously described offshore bar system in Tampa Bay seagrasses (Lewis et al. 1985) may be one factor needing greater consideration when projecting seagrass recovery rates, in combination with the results of an REI study.

The bar system is correctly referred to as a "longshore bar" following the nomenclature of Shepard (1952). Fonseca et al. (1998; Figure 1.3, page 11) include an oblique aerial photograph of a seagrass bed in Tampa Bay showing the longshore bar system and note that "Reduction in wave energy from both the shelving shoal and the grass itself is thought to be responsible for the resultant seagrass bed landscape pattern."

The Longshore Bar System

Lewis et al. (1985) described the seagrass meadows of Tampa Bay as consisting of five types (Fig. 1). These include four perennial types (present year round) and one ephemeral type (reappearing annually and often changing specific location and extent). These are:

1. Mid-bay shoal, perennial — MBS(P)
2. Healthy fringe perennial — HF(P)
3. Stressed fringe perennial — SF(P)
4. Ephemeral — E
5. Colonizing perennial — C(P)

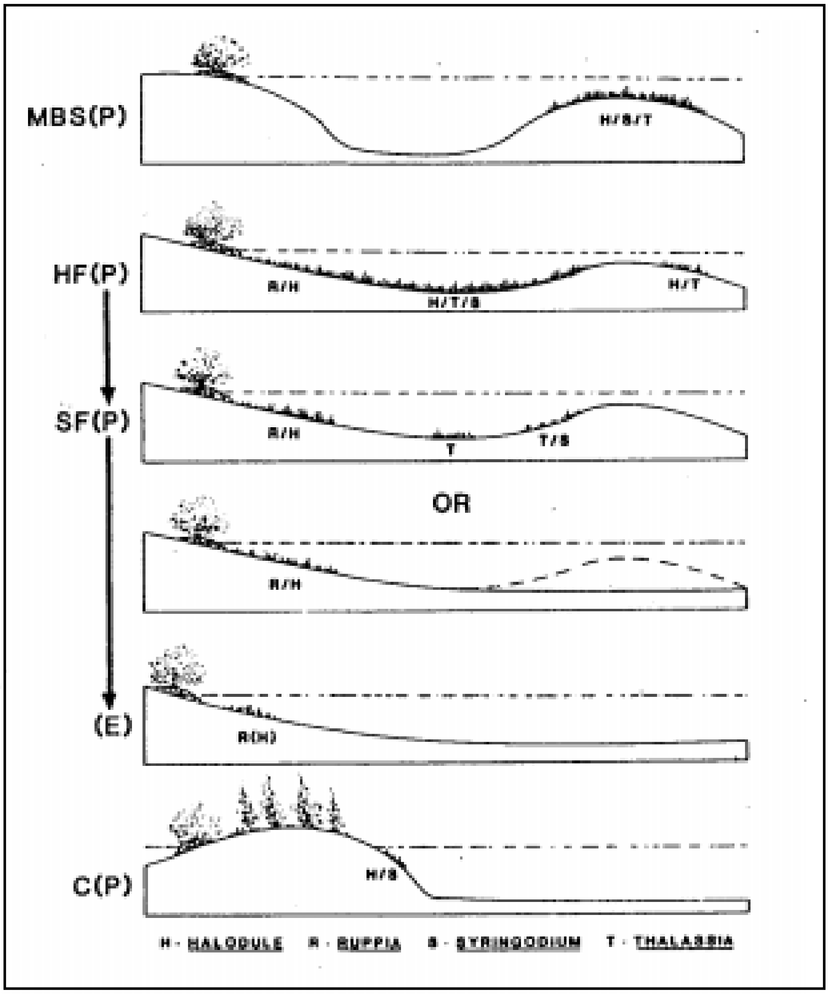

Figure 1. Seagrass meadow types in Tampa Bay (from Lewis et al. 1985).

It was theorized in that paper that types 2–4 represent stages in the eventual disappearance of a particular meadow due to man-induced stress described as "Destabilization of the offshore bar…" which "…apparently leads to inshore migration of the bar and its eventual disappearance…" (page 220 in Lewis et al. 1985). A careful examination of Figure 2 from the same paper, illustrating the "healthy fringe perennial" type of seagrass meadow, shows the presence of an emergent and often unvegetated bar between bayward seagrass areas and landward seagrass areas. Figures 3 and 4 are photographs of a particular bar system located in Lower Tampa Bay east of one of the filled causeways for the Sunshine Skyway Bridge.

THE HYPOTHESIS

As illustrated in Figure 1, it was hypothesized that the loss of the deeper seagrasses on the outside (bayward) side of the bar might be the first step in destabilization. This was linked with the widespread observations in many seagrass meadows around the world that reductions in water clarity due to phytoplankton blooms or turbidity would first stress those seagrasses living closest to their compensation point.

The hypothesis by Lewis et al. (1985) was that the bar in some way protected the existing seagrasses from wave or current energy, and that its loss removed that protection, and resulted over time in the

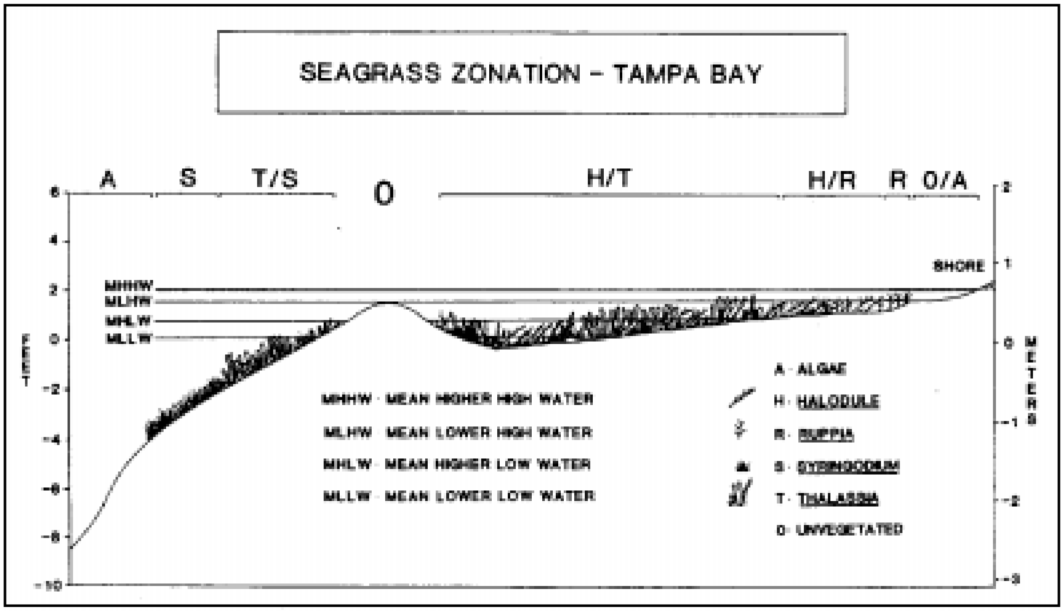

Figure 2. Seagrass zonation in Tampa Bay (from Lewis et al. 1985).

reduction in the size of the seagrass meadow.

It is also significant to this hypothesis that existing seagrass meadows showing recovery towards their historical distribution appear to be stymied from further progress or have shown declines in recent years. The seagrass meadow first studied in detail by Johannson and Lewis (1992) offshore of Wolf Branch Creek in Lower Tampa Bay (Figs. 5, 6 and 7) historically had a nearly continuous longshore bar system extending north from the mouth of the Little Manatee River for a distance of approximately 8 km. That entire system has disappeared in the last 40 years. I hypothesize that until the bar system is replaced, natural colonization that might expand the existing meadow will be limited to just a fraction of the potential area available with suitable water quality due to hydrodynamic stress. Similar examples exist in the area north of the Howard Frankland Bridge in Pinellas County (Old Tampa Bay).

Potential support for this hypothesis is illustrated by the volunteer colonization of seagrasses behind a breakwater constructed by the Florida Department of Transportation at the south end of the Sunshine Skyway Bridge in 1995 as part of Permit Number 411352329 issued on February 2, 1988 (Figure 8). Intended as mitigation for seagrass losses associated with the causeway expansion, the project was never planted as planned, and instead became essentially 100% colonized by volunteer seagrass within four years.

The Next Step

All of the above information is largely anecdotal and observational. However, a more detailed study of the entire issue is warranted, beginning with more detailed historical mapping of the bar system and documentation of what has happened to it. Largely intact examples of the system are present in the southern portions of Tampa Bay. With the current commitment on the part of the Tampa Bay Estuary Program to provide matching funding to allow a relative wave-exposure index (REI) study of Tampa Bay to be conducted by the National Marine Fisheries Study, additional important information could be gathered by combining the two studies.

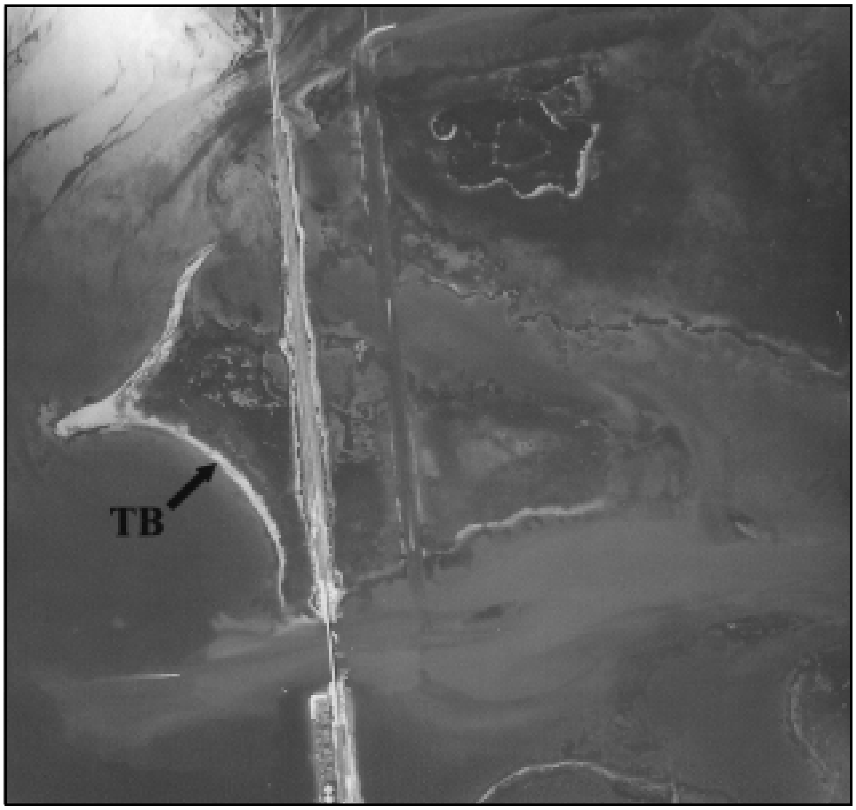

Figure 3. Vertical aerial photograph of the north approaches to the Sunshine Skyway Bridge with seagrass beds and the distinct longshore bar (LB) shown at the arrow. Photo date February 20, 1993.

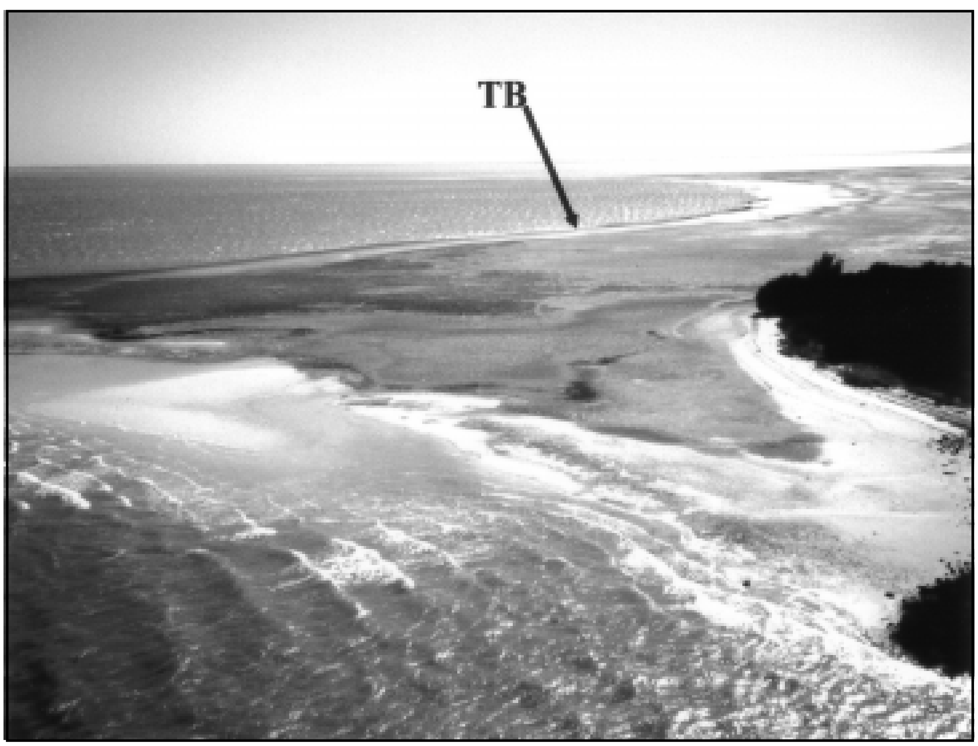

Figure 4. Low level oblique photograph taken from the top of one of the Sunshine Skyway Bridge approaches, February 2000, on an extreme winter low tide. The longshore bar system is visible at the arrow (TB). Same location as that shown in Figure 3.

Figure 5. Composite vertical aerial photograph of the Wolf Branch seagrass meadows north of the mouth of the Little Manatee River in Middle Tampa Bay, March 23, 1957. The longshore bar system is clearly visible at the arrow.

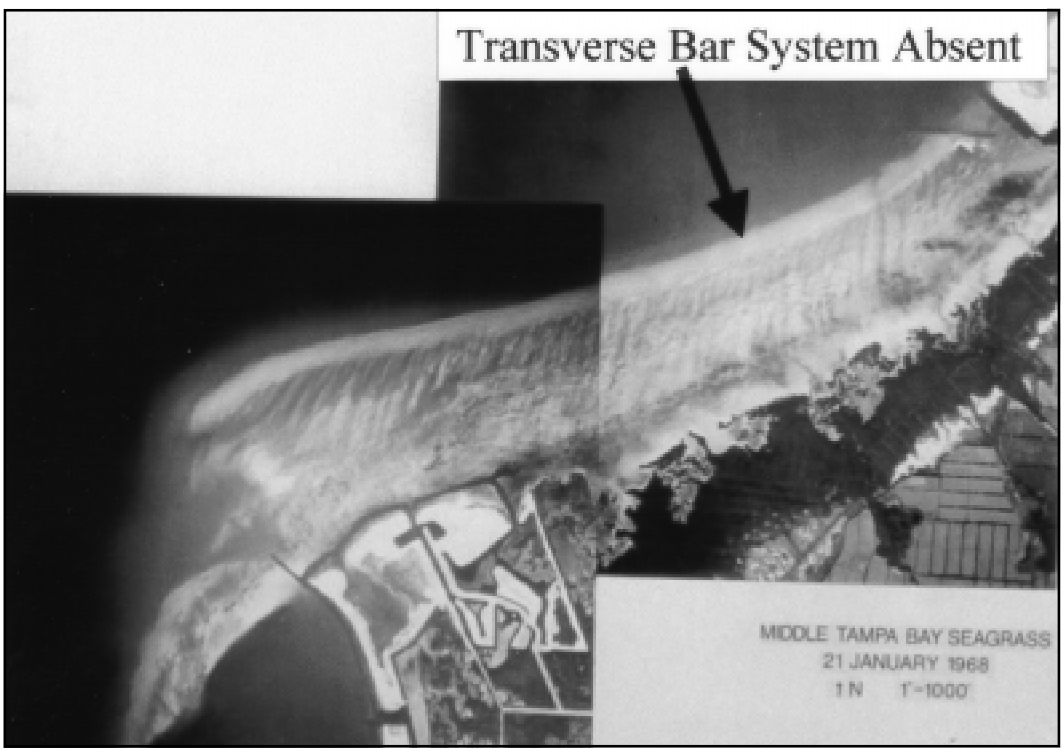

Figure 6. Composite vertical aerial photograph of the Wolf Branch seagrass meadows north of the mouth of the Little Manatee River in Middle Tampa Bay, January 21, 1968. The longshore bar appears degraded and the seagrass meadows much thinner.

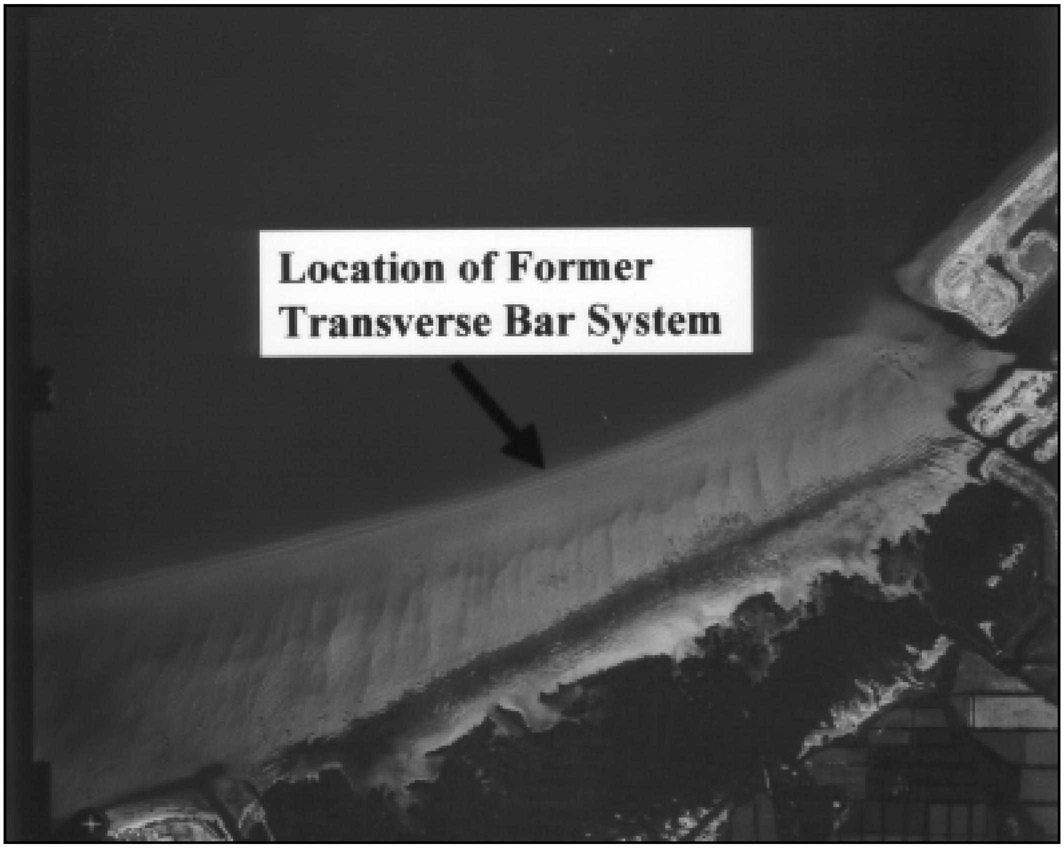

Figure 7. Vertical aerial photograph of the Wolf Branch seagrass meadows north of Simmons Park, December 10, 1990. Seagrass meadows are restricted to shallow water. The historical longshore bar system is absent.

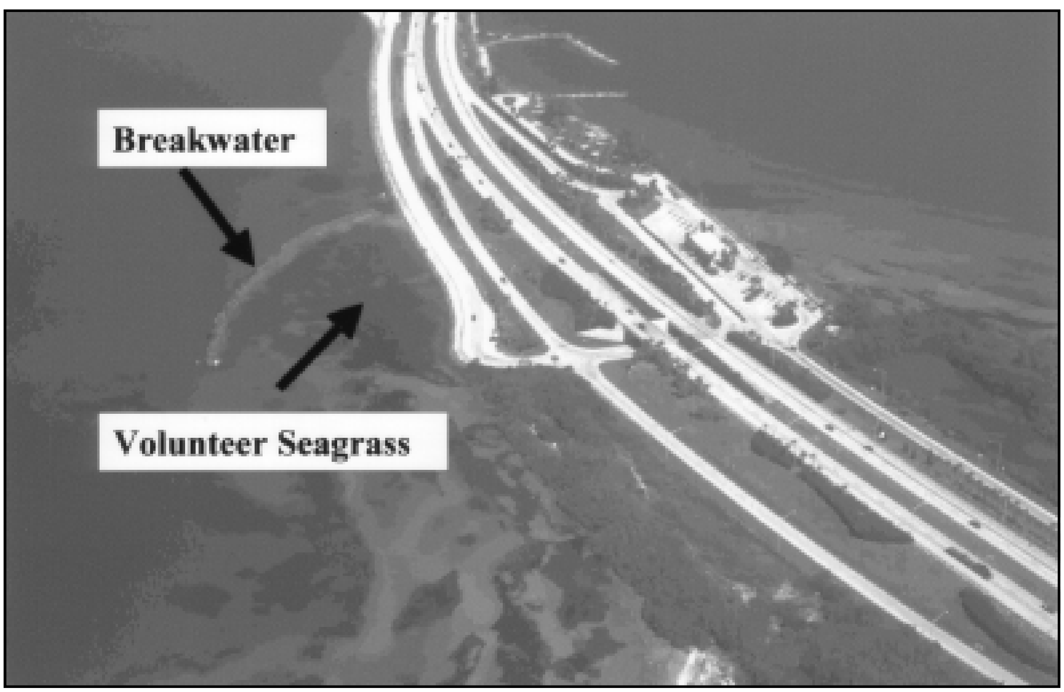

Figure 8. Oblique aerial photograph looking north from the Manatee County side of the Sunshine Skyway Bridge approach causeway. The FDOT constructed breakwater and volunteer seagrasses are shown. Photo date: August 20, 2000.

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PRODUCTION OF RHIZOME MERISTEMS BY *THALASSIA TESTUDINUM*

Clinton Dawes, John Andorfer

ABSTRACT

It was hypothesized that the average of 7.5 years required for regrowth of the tropical seagrass *Thalassia testudinum* into propeller cuts is due the slow production of rhizome meristems. Rhizome transplants with double short shoots were subjected to different levels of plant growth regulators, fertilizer treatments and planting techniques in experimental field and tank nurseries. Field experiments demonstrated that the presence of intact apical meristems prevented the formation of new lateral branch meristems. Transplant survivorship in the field varied widely (29%–83%) after 7 to 9 months. In contrast, plants in tank culture showed 82%–98% survivorship with new blade, root and rhizome growth after 8–19 weeks. Use of various fertilizers and plant growth regulators had no observable effect on rhizome production. Regardless of experimental design, new rhizome apices were produced only from existing short shoots, never from the rhizome or the basal portion of short shoots. Further, short shoots from young double units (120–180 days old) lacking a rhizome meristem produced few, if any rhizome tips, even after 4 months of growth. In contrast, older double units (300–375 days old) exhibited significantly higher production of new rhizome tips over the same period. *Thalassia testudinum* displays strong apical dominance of the rhizome. Thus, there is a long-term (years) delay in regrowth into propeller cuts after the rhizome is damaged. Further, production of new tips is primarily by older short shoots, suggesting that formation of turtle grass nurseries should include older transplants without rhizome apices.

INTRODUCTION

Declines of coastal seagrass meadows are evident throughout the world and have been linked to natural and human-induced disturbances (Short and Wyllie-Echeverria, 1996). Early reports of seagrass declines began with the widespread reduction of eel grass (*Zostera marina* L.) communities in northern Europe and the northeastern United States in the 1930s due to the wasting disease (Dexter, 1985; Den Hartog, 1996). Studies on seagrass Declines include western Australia (Cambridge et al., 1986), Chesapeake Bay, Maryland (Orth and Moore, 1983), and the Great Bay estuary in New Hampshire (Short, 1992). Most studies on seagrass damage have focused on human affairs (Thayer et al., 1975; Short and Wyllie-Echeverria, 1996; Dawes, 1998) because they can be controlled and are important to management decisions regarding estuaries and coastal habitats.

Seagrass losses have been most severe in estuaries and coastal communities where Florida seagrass beds are most abundant. Studies in Florida have involved Tampa Bay (Johansson, 1991), Indian River (Haddad and Harris, 1985), and Florida Bay (Roblee et al., 1991; Durako, 1994). Losses in the Indian River estuary on Florida's east coast have resulted in a 30% decline (Haddad and Harris, 1985). Livingston (1984) reviewed the status of seagrass beds along Florida coasts and reported a decline in 7 of 12 bay systems studied. Tampa Bay, with 1,036 km^2 of surface water, may have supported 30,970 ha of seagrass meadows in 1870 (Johansson, 1991); however, by 1982, there were only 8,763 ha of seagrass beds remaining.

Damage from propeller scars of power boats has long been recognized as a serious mechanical impact on seagrass beds in Florida Bay (Zieman, 1976), the Florida Keys (Matthews et al., 1991, Sarasota Bay (Folit and Morris, 1992) and Tampa Bay (Durako et al., 1992; Dawes et al., 1997). Aerial surveys of Florida's coasts indicated that more than 70,000 ha of the 1.1 million ha of seagrass beds showed some level of scarring (Sargent et al., 1995). Regrowth into propeller scars by *Thalassia testudinum* Banks ex König (turtle grass) is slow. Recovery rates are estimated to take 2–5 years in the Florida Keys (Zieman, 1976), 3.6 to 6.4 years in upper Tampa Bay (Durako et al., 1992), and 7.6 years in middle Tampa Bay (Dawes et al., 1997).

The slow recovery and the probability of continued mechanical impacts on turtle grass suggest a need for the development of nursery stock and techniques to increase growth and survival. In addition to the use of seeds, other possibilities include creation of nurseries as sources for mitigation planting and induction of new rhizome meristems *in situ* within damaged beds. Presently, the only source for turtle grass transplants are existing beds with survival being about 30%, for single short shoots (Tomasko et al., 1991). Further, morphological studies suggest that *Thalassia testudinum* rhizomes do not proliferate after being damaged (Tomlinson, 1974). The present report includes studies on inducing vegetative expansion of *T. testudinum* through production of rhizome tips.

MATERIALS AND METHODS

Site Selection

Field experiments and collection of *Thalassia testudinum* specimens for laboratory work were carried out in Cockroach Bay (CRB), a 760 ha estuary on the east side of Tampa Bay Florida, U.S.A. (27°41’ N, 82°30’ W). The shallow water (0.5–4 m depth) habitat is designated as a State Aquatic Preserve consisting of mangrove (Dawes et al. 1999) and seagrass (Dawes et al. 1997) communities.

Plant Material

Rhizome units with two short shoots (double short-shoot units) of *Thalassia testudinum* from Greater CRB were used in the tank and field experiments. Although use of single short shoots is the goal in transplantation studies, double short-shoot units were selected because of their higher (75%–85%) survivorship compared to 30% survival of single units (Tomasko et al. 1991). The units were easily removed from the edge of a tidal channel where erosion had reduced sediment coverage to less than 5 cm. The units were placed into a cooler in seawater and brought to the transplant site in CRB or back to the laboratory for tank culture. Young double short-shoot units with a rhizome meristem were frequently uncovered; these had the youngest shoot removed except when needed for an experiment.

Field Experiments

Field studies were performed at two sites inside two Recovery Areas in the CRB Preserve (RA 2, 4) and one site in Tampa Bay, outside of CRB between May 1997 and May 2000. All transplant sites were unvegetated patches surrounded by monospecific beds of *Thalassia testudinum*. The bare areas were a maximum of 1.5 m wide on a side and were of similar depth as the surrounding seagrass vegetation. Field studies tested the use of sediments (silica sand, commercial top soil, *in situ* sediment), containers (cardboard boxes, plastic pots), and nutrients (Forest Tablets®: NPK= 20-8-2, A.M. Leonard Inc., Piqua Ohio). The plant growth regulators tested included naphthaleneacetic acid (NAA), kinetin, and a rooting powder, Rootone® (0.2% NAA + 4% Thiram; Green Light Inc., San Antonio Texas).

1998 Nursery. Three plots were established in June 1998 in the three sites mentioned above and harvested 9 months later with *Thalassia testudinum* arranged in groups of 8 double short-shoot units for each treatment. The units were either planted directly in a 20-cm deep pit filled with silica sand or in rectangular plastic pots with silica sand. The seven treatments at each site included: (1) no pot or fertilizer (control); (2) no pot and fertilizer added after 1 month; (3) no pot and fertilizer added at time of planting; (4) in pots and fertilized at time of planting; (5) in pots without fertilizer; (6) no pot and fertilized at time of planting and the rhizome having an apical meristem; and (7) no pots and not fertilizer with the rhizome having an apical meristem. Fertilization was by the addition of two Forest Tablets® every 8 weeks to the designated treatments throughout the growing period (June– October; see Dawes et al., 1997).

1999 Nursery. Cardboard boxes were used to avoid the increased anaerobic nature of the sediment and restriction to horizontal expansion of the rhizome units that resulted with plastic pots. The same three sites were used as in the previous field studies with planting occurring in May and harvesting 7 months later by which time the boxes had disintegrated. Double short-shoot units of *Thalassia testudinum* were planted in the boxes with 20 cm of silica sand or commercial topsoil. Half of the boxes of each sediment type were fertilized with 2 Forest Tablets®; the other boxes received no additional nutrients. Each box contained four "young" and four "older" units and were embedded in the sediment in a haphazard pattern in clear areas within seagrass beds. The relative age of these plants was determined by taking into account their position on a rhizome along with the length of their vertical axes. Young short shoots taken from the first few shoots on a clone (1st and 2nd), had very short vertical axes (<1 cm), and their rhizomes were soft and flexuous. Older short shoots were the 4th and 5th or higher numbered ramets of a rhizome, had vertical stems that were elongated (1–3 cm), and a rigid horizontal rhizome due to development of fiber bundles.

Culture Experiments

Double short-shoot units of *Thalassia testudinum* were planted into glass aquaria the same day as collection (37 or 74 L tanks) after removal of old blade material. The tanks had been previously filled with seawater collected from the Gulf of Mexico and sand at least 1 week before use. The tanks were evenly spaced and illuminated by ten 2.5 m (8ft) Philips cool white fluorescent bulbs providing about 300 M photons m^{-2} s^{-1} at the water surface using a 14/10 h light/dark cycle. The seawater (30 ppt salinity, 20–22^{o}C) was filtered using a hanging filter (Whisper Inc.) on each tank. An additional circulating pump (Penguin Inc.) was added to the larger 74 L tanks. The double short-shoot units were planted in silica sand (~10–15 cm deep) directly in the aquaria or in shallow trays placed in the tanks. The double short-shoot rhizomes were aligned perpendicular to the long axis of each tank with either 8 (37 L) or 18–24 (74 L) units per tank. If possible, the experiment was begun after all double short-shoot units had produced a new set of blades and roots (ca 4 weeks) to allow for new growth after replanting.

Experiment 1. A 9-week experiment used 72 double short shoot units of *Thalassia testudinum* planted in three 74 L aquaria. Each tank was divided into three sections containing low, medium and high levels of nutrients in the sediment that had been divided into three equal parts with acrylic dividers and sealed with silicon caulk. The sediment consisted of 15 cm washed silica sand. Each sediment chamber was randomly chosen to receive 1, 2, or 3 Forestry Tablets® that were split and spread evenly in the sand to a depth of 15 cm.

The units were placed into each sediment nutrient regime and assigned haphazardly to be treated with an application of lanolin paste containing one of four possible naphthaleneacetic acid (NAA, Sigma Inc.) concentrations: 0 M, 10^{-7} M, 10^{-5} M, 10^{-3} M. The NAA was dissolved in warmed, liquid lanolin. After cooling to room temperature, the paste was applied to the cut ends of the rhizome as well as an area of the short-shoot just below its apical meristem. The short shoots had been scraped to remove the outer leaf bases with a razor blade. Two double short-shoot units from each hormone treatment were placed into each of the sediment nutrient treatments in all three tanks. Experimental design included 2 x 4 hormone treatments x 3 sediment nutrient levels x 3 tanks. The above- and below-ground biomass were separated into leaves, short shoots, roots and rhizomes with the number of new apical meristems and short shoots recorded. To minimize the influence of pseudo-replication, each tank was treated as a replicate (n=3), and the similarly treated plants in each tank were analyzed as subsamples.

Experiment 2. A similar, 8-week experiment, was initiated without sediment nutrients. Double short-shoot units were planted in three rows of 8 units in each of the four 74 L tanks. Prior to planting, the short shoots were treated with lanolin paste alone (control) or one of three hormone mixtures after removal of the leaf bases by scraping. These included 10^{-5} M kinetin in lanolin, 10^{-5} M NAA + 10^{-6} M kinetin (10:1 ratio NAA:kinetin) in lanolin, or Rootone® powder that was applied to the plants 5 min. prior to planting.

Experiment 3. Fourteen double short-shoot units were grown in four 74 L aquaria for 19 weeks. The rhizome and short shoots of seven units in each tank were coated with Rootone® powder and planted with the seven untreated units. Two of the aquaria received 4 Forest Tablets® split and distributed in the silica sand; the other tanks received no added nutrients. After 1 month, one half of a tablet was added to the high nutrient tanks.

Experiment 4. Determination of the age class (young, old short shoots) of *Thalassia testudinum* transplants was based upon physical appearance and position on the rhizome (Witz and Dawes, 1995). Three "young" and three "older" double short-shoot units of *Thalassia testudinum* were grown in plastic trays filled with silica sand for 17.5 weeks after 4 weeks of acclimation. Three trays were placed into each of four 74 L aquaria for a total of 72 units. All the trays from two aquaria were removed weekly and transferred to a container filled with a 2 mM nitrate ($NaNO_3$) of the same salinity as the aquaria. After 4 hours in the enriched seawater, the trays were rinsed in fresh seawater prior to their return to the aquaria. A third tray from each aquarium was also treated weekly by soaking in a 0.05 mg L^{-1} solution of gibberellic acid (GA_3) similar to the nitrate treatment. The fourth tray from each tank was similarly soaked in a 5 mg L^{-1} solution of GA_3. One of the fertilized trays from each tank received no plant growth regulators.

Data Analysis

All data were tested for normality and homogeneity of variance and transformed if necessary to satisfy the assumptions of parametric statistics. Analyses of variance (ANOVA) were conducted to test for significant differences between treatments. Multiple comparisons were made using Dunn-Bonferroni and the Student-Newman-Keuls methods. Significance was determine *a priori* at the 99% ($p \leq 0.01$) and 95% ($p \leq 0.05$) probability levels.

RESULTS

Field Experiments

The double short-shoot units of *Thalassia testudinum* transplanted to all three sites in CRB showed overall survival rates of 29% (1997; data not shown), 54% (1998), and 75% (1999). Survivorship was consistently lowest in the most exposed site (Tampa Bay) and highest in the most protected one within CRB (RA 4). The use of pots and boxes resulted in higher survival rates in the 1998 and 1999 experiments.

1998 Nursery. Survival of the double short-shoot units differed between transplant sites. Lowest survivorship (36%) occurred in Tampa Bay while recovery of transplants after 9 months was greater within Cockroach Bay with 41% and 84% of transplants recovered in RA 2 and RA 4, respectively. Regardless of treatment, few rhizome meristems were produced if the original rhizome apical meristem was present (Fig. 1a). In contrast, a significantly higher number of short shoots were produced on plants with rhizome meristems compared to ones lacking the apex (Fig. 1b). New rhizome meristems were produced, but only from short shoots and not the horizontal rhizomes. Production of new apical meristems and new short shoots showed no relationship with addition of fertilizer or the use of plastic pots.

1999 Nursery. Survival of the double short-shoot units transplanted in cardboard boxes with silica sand was higher compared with the previous experiments with 63% for Tampa Bay and RA2 and 83% for RA 4. Use of silica sand, sand and fertilizer, topsoil, or topsoil plus fertilizer showed no difference in the number of new apical meristems initiated or in the number of new short shoots produced (Table 1). However, apical meristem and short-shoot production was significantly higher ($p=0.05$) in the old short shoots compared to young ones. The trend was also independent of sediment type or fertilizer treatment (Table 1).

Culture Experiments

The double short-shoot units of *Thalassia testudinum* in tank culture showed a high level of survivorship in all experiments as well as production of new blades and roots. In the four tank experiments, 98% survived over 9 weeks (Expt. 1), 95% over 8 weeks (Expt. 2), 85% over 19 weeks (Expt. 3) and 82% over 17.5 weeks (Expt. 4).

Experiment 1. Application of NAA or levels of Forest Tablets® did not significantly ($p \leq 0.05$) alter root biomass (Figure 2a) or production of new rhizome apices in *Thalassia testudinum* (Figure 2b). Although the number of replicates was low, there were a significant number ($p \leq 0.05$) of new rhizome apices (Figure 2b) within 9 weeks.

Experiment 2. There were no significant differences between use of the plant growth regulators kinetin, auxin plus kinetin, or Rootone® and the control double short-shoot units in terms of leaf biomass, root biomass, or the number of new roots ($p \leq 0.05$). In contrast to the Tank Experiment 1, production of rhizome meristems was much lower possibly reflecting the lack of nutrient addition.

Experiment 3. Application of Rootone® powder to double short-shoot units did not have any significant effect on production of rhizome meristems (Figure 3a, $p= 0.5$) or on the number of new short shoots (Figure 3b, $p=0.18$) produced over the 19 week period. The commercial rooting agent, Rootone®, significantly influenced root biomass in one of the two unfertilized tanks (Tank 1; Figure 3c, $p< 0.001$). The greatest influence on production of root mass was the addition of nutrients where fertilized tanks (Fig. 3c, Tanks 2 and 4) contained significantly less root biomass than did unfertilized ones (Figure 3c, Tanks 1 and 2; $p \leq 0.01$). Root production in fertilized tanks was low and independent of the application of rooting hormone. However, the lower amount of root mass of the fertilized plants did not appear to inhibit their production of new rhizomes or short shoots (Figures 3a, 3b).

Experiment 4. The young and older short-shoot age classes were significantly different based on the number of leaf scars (Figure 4a, $p \leq 0.05$). Young short shoots had significantly fewer leaf scars (mean =

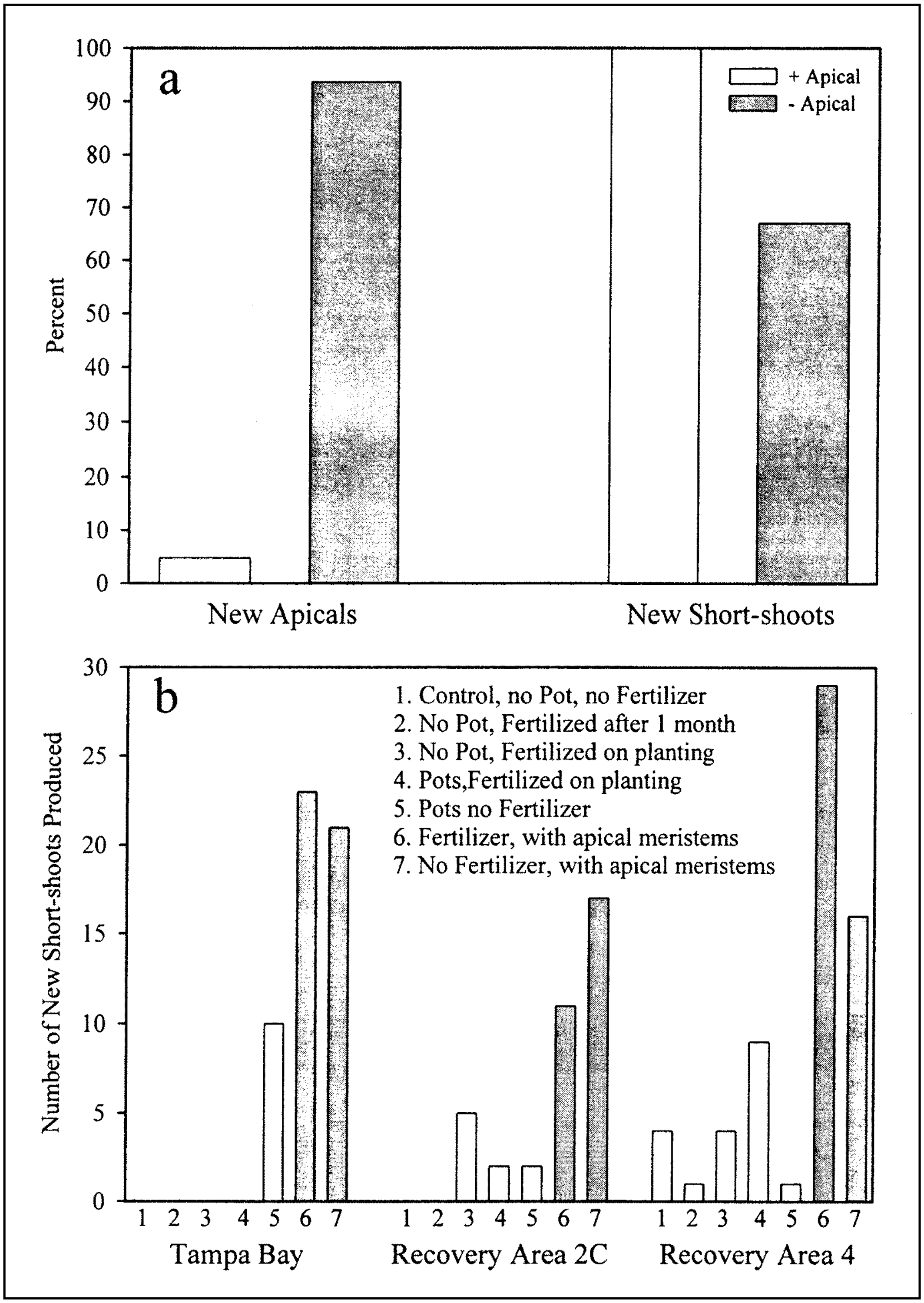

Figure 1. The percent of transplanted double short shoots of *Thalassia testudinum* having new rhizome apical meristems and short shoots. Transplants planted with (open bars) and without (solid bars) rhizome meristems are shown in Figure 1a. Figure 1b shows the number of new short shoots produced by transplants in each treatment at each of the three nursery sites. Presence of apical meristems at planting are shown by solid bars.

Table 1. The production of apical meristems and new short shoots by *Thalassia testudinum* double short-shoot transplants. Plants were transplanted into 3 sites in and around Cockroach Bay, Florida in May of 1999. The plants were placed into cardboard boxes containing different types of sediment with fertilized transplants receiving 16 g of a 22 N-8 P-2 K fertilizer (Fert.). After 6 months of growth the transplants were harvested; the number of new apical meristems and short shoots were significantly higher ($p \leq 0.01$) on old when compared to young short shoots.

		Number of New Components			
		Apicals		Short shoots	
Site	Treatment	Young	Old	Young	Old
Recovery Area 4	Sand	0	1	0	2
	Sand + Fert.	0	2	0	0
	Top Soil	0	3	0	5
	Top Soil + Fert.	0	1	1	4
Recovery Area 2	Sand	0	1	0	4
	Sand + Fert.	0	3	4	7
	Top Soil	0	0	0	1
	Top Soil + Fert.	0	2	0	0
Tampa Bay	Sand	1	5	4	1
	Sand + Fert.	0	4	0	5
	Top Soil	0	0	0	2
	Top Soil + Fert.	0	3	0	5

10.5 scars) and were determined to be 120–180 days old, while older ones (mean = 21.9 scars) were 300-375 days (Figure 4a, $p < 0.001$).

Although survival of young and older double short-shoot units was 80-82% (Figure 4b), the production of new rhizome meristems was almost exclusively by older short shoots (Figure 4c). Young units produced only 2 new apical meristems on 34 short shoots, while older ones produced 29 new meristems on 35 short shoots. Hormone and nitrate treatments did not affect survivorship or rhizome tip production (data not shown).

DISCUSSION

The restoration or mitigation (see Lewis, 1989 for definitions) of seagrass beds requires a source of transplants and presently all efforts use existing communities as donor sites (Fonseca et al., 1998). Not only are the donor beds damaged in the process, but survival of the transplants ranges from 0 to 100% with a mean planting unit survival of 35% and only 10% of the plantings achieving 100% cover within the monitoring period (Fonseca et al., 1998). The data reviewed by Fonseca et al. (1998) indicates a need for seagrass nurseries as well as techniques that ensure higher survival rates and induction of vegetative expansion.

The most successful efforts in transplanting seagrasses are with species that produce multiple rhizome apices such as *Zostera marina* and *Halodule wrightii* so that vegetative expansion is rapid. However, *Thalassia testudinum* is the third most commonly planted species (21%) when compared with *Z. marina* (48%) and *H. wrightii* (26%), with 53% of the plantings using bare short shoots (fonseca et al., 1998). This is because *T. testudinum* beds are the most developed and widespread in Florida and Gulf of Mexico waters, and support a highly diverse community (Dawes, 1998). Unfortunately, use of single short shoots with a rhizome segment has a survival rate of about 30% (Tomasko et al., 1991).

Thalassia testudinum rhizomes rarely branch during horizontal growth so that the

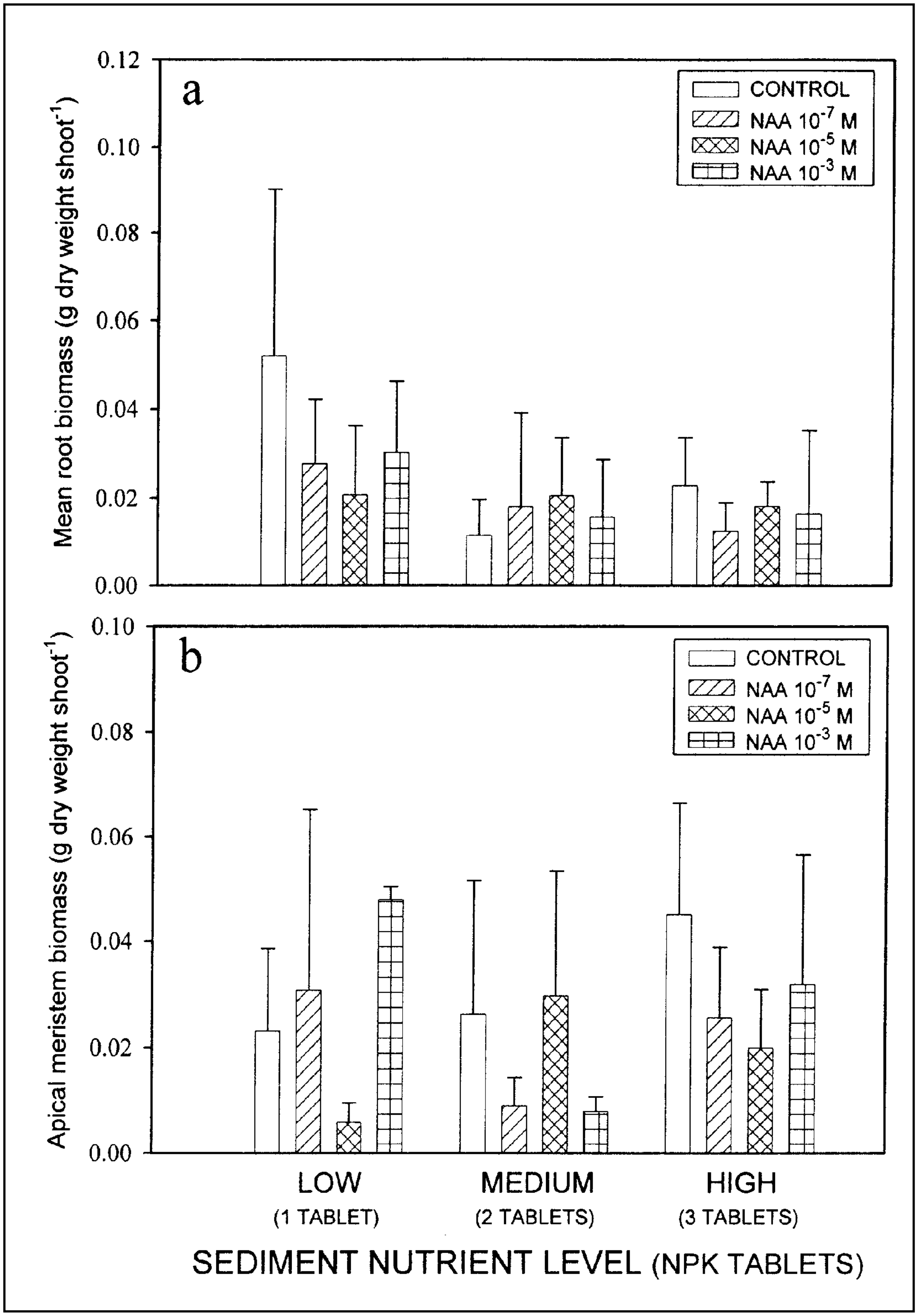

Figure 2. The response of *Thalassia testudinum* double short-shoot transplants after 2 months of growth in glass tanks. The double short-shoot units were treated with different levels of the hormone auxin and exposed to different concentrations of sediment nutrients. The mean root biomass of these transplants (Figure 2a) and the biomass of rhizome meristem (Figure 2b) are given with standard errors (±1 S.D.).

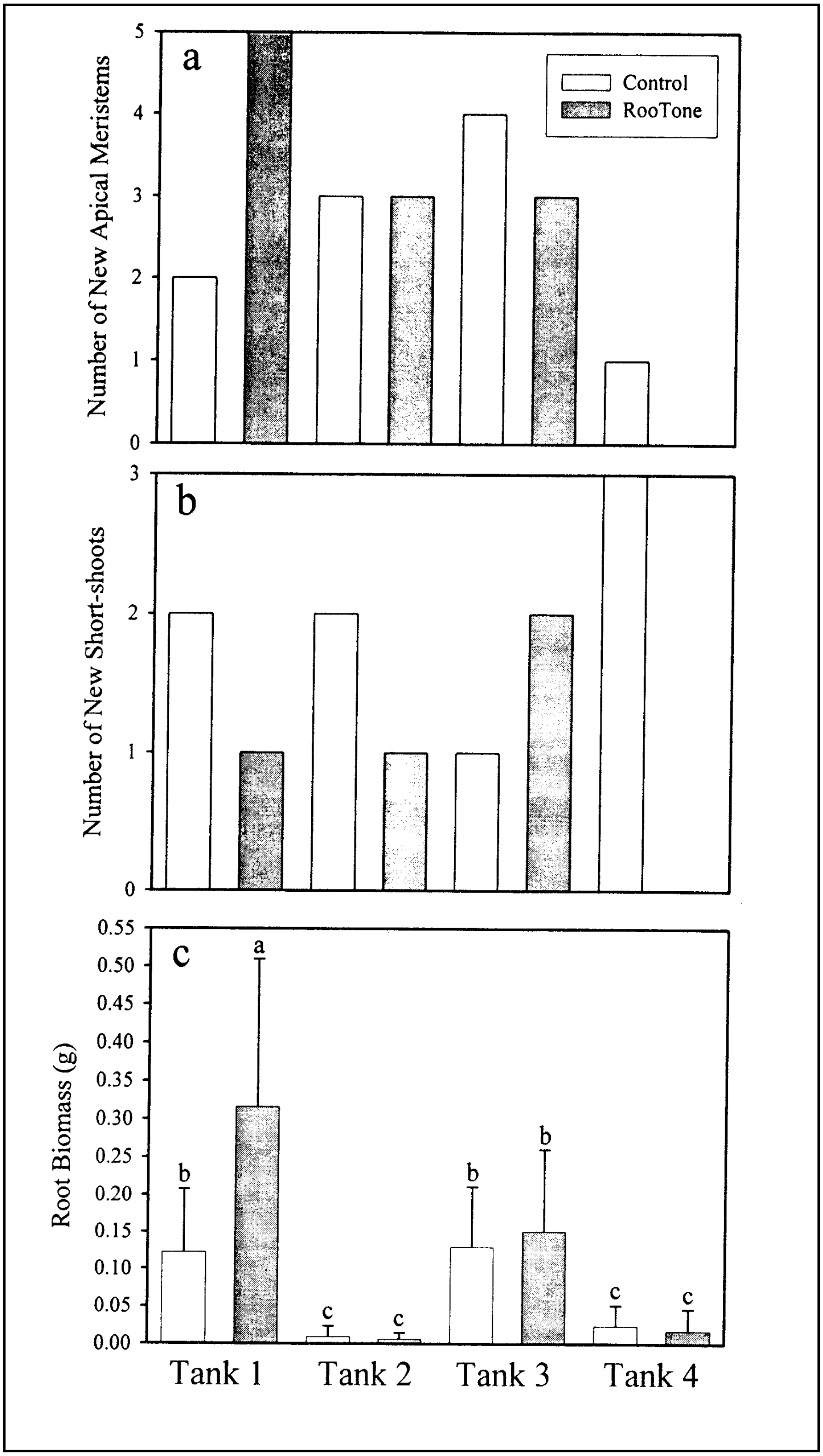

Figure 3. The response of *Thalassia testudinum* double short-shoot transplants to the application of a commercial rooting agent. Tanks 2 and 4 received 4 Forest Tablets nutrients. The number of apical meristems (Figure 3a), the number of new short shoots (Figure 3b) and the amount of root material (Figure 3c) produced are shown. Significant differences between treatments are denoted by different letters (±1 S.D.).

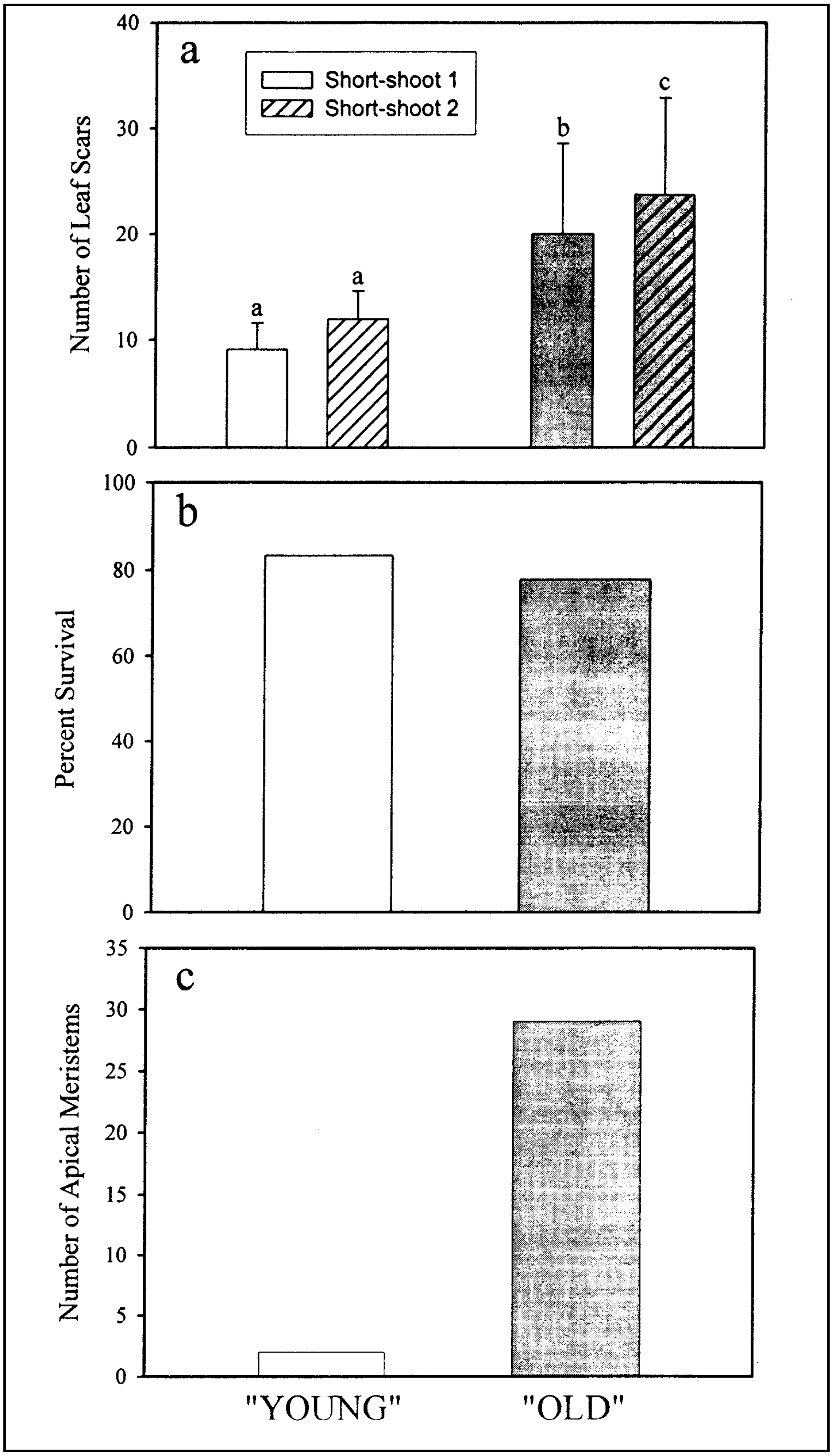

Figure 4. Responses by young and older double short-shoot units of *Thalassia testudinum*. The number of leaf scars present on plants that were assigned to the "young" and "older" treatments is shown in Figure 4a (±1 S.D.) and significant differences between treatments are denoted by different letters. Figure 4b shows the percentage of plants surviving after 4 months of growth in culture, while Figure 4c depicts the number of rhizome meristems produced.

slow recovery in propeller cuts reflects the lack of multiple apices. The rhizome apex, like apical meristems of tree branches, controls proliferation of short shoots (e.g. lateral branches), this production is ordered (Tomlinson, 1974) and always from a lateral position (Tomlinson and Bailey, 1972). The indeterminate rhizome meristem that produces the roots and short-shoot initials is the basis of all vegetative expansion in *Thalassia testudinum*. The clonal nature of all seagrasses (Dawes, 1998) results in dependence on an actively growing rhizome and its meristem. When the rhizome of *T. testudinum* is cut by a boat propeller, the portion of the genet lacking a rhizome tip will not grow until a new rhizome meristem is formed. This will require 10 (Kelly et al., 1971) or less (this study) months. The present study supports the concept of Tomlinson (1982) that production of an indeterminate rhizome meristem in turtle grass is always by a short-shoot. In addition, the short-shoot meristem produces blades, roots, and flowers and extends the ramet vertically. Further, all of the new rhizome meristems that we observed arose from below the short-shoot meristem. Perhaps there are existing sites (e.g. endogenous "buds") of initiation that could be identified by anatomical studies.

The rhizome meristem of *Thalassia testudinum* appears to strongly influence its growth form by limiting the number of lateral branches as well as their position. Thus, understanding how lateral buds are released from this dominance will help in propagation of *T. testudinum* in culture. Apical dominance, the control exhibited by the apical portions of a shoot over lateral buds (Cline, 1994), is thought to be controlled by plant growth regulators, specifically auxin, although cytokinins and abscissic acid likely play a role as well. Apically derived auxin either directly controls lateral bud development by repressing outgrowth upon entering this region or indirectly via some other mechanism (Cline, 1997), and begins with the formation of a lateral bud within the axils of leaf primordia.

Dormant lateral rhizome buds have not been observed on short shoots of *Thalassia testudinum* although the existence of dormant short shoots is known (Van Tussenbroek et al., 2000). That study demonstrated that foliar activity of dormant short shoots can be re-initiated through fertilization. If quiescent rhizome buds are also present, they most likely are inhibited by the existing rhizome tip and, or the short shoot meristem. Anatomical studies on the short-shoot meristem are needed to clarify if endogenous rhizome initials are present in *T. testudinum*. Additionally, the interaction between the short-shoot and rhizome meristems in controlling lateral bud development must be investigated. The present study showed that removal of only the rhizome apical meristem results in production of new rhizome meristems from the short shoots, but only after many weeks. The slow response may be the result of some inhibition imposed by the short-shoot meristem. For example, a combination of shoot excision and rhizome meristem decapitation results in the highest percent of bud activity in *Elytrigia repens* (McIntyre and Cessna, 1998).

The following characteristics of *Thalassia testudinum* can be summed up based on the present study. (1) After removal from sediment, the original roots die and it takes about 1–2 months before new roots are established with a delay in growth. (2) New roots and rhizome meristems are produced from short shoots only if the rhizome tip is removed. (3) Older short shoots (ca 300–375 days old or about 25 leaf scars) are the source of rhizome meristems when the original apex is removed. (4) If a rhizome meristem is present, it inhibits production of others. (5) Rhizome apices can develop rapidly (within 2 months) as shown in tank culture where salinity, temperature, light and nutrients can be controlled. (6) Thus far, use of some plant growth regulators did not significantly induce rhizome meristems.

Based on the present studies, some preliminary suggestions can be given for creation of *Thalassia testudinum* nurseries. Older double short shoots or young ones with the rhizome tip removed can be planted in 50 cm^2 by 15-cm deep pressed paper planters using silica sand and placed in open tanks with running seawater. The present tank and field studies as well as previous ones (Dawes et al., 1997) indicate that transplants should be exposed to at least 200 M photons m^{-2} s^{-1} or more of sun or artificial light using a 12-h photoperiod. Salinity should be around 30 ppt and water temperature 20-30 ^{o}C. After 4–8 weeks, new roots should have formed and some rhizome apices developed on older short shoots. The planters can then be moved to the field in flat carrying trays and placed in 20-cm deep depressions in the sediment taking care not to damage newly formed roots. Excess plantings can be subdivided every 8–12 weeks in the nursery tanks.

ACKNOWLEDGEMENTS

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THE DYNAMICS AND DISTRIBUTION OF THE SLIME MOLD *LABYRINTHULA* SP. AND ITS POTENTIAL IMPACTS ON *THALASSIA TESTUDINUM* POPULATIONS IN FLORIDA

Barbara A. Blakesley, Donna M. Berns, Manuel F. Merello,
Margaret O. Hall, Jitka Hyniova

ABSTRACT

Data from three different studies were used to evaluate the distribution of the parasitic slime mold *Labyrinthula* sp. in Florida Bay, the eastern Gulf of Mexico, and the Tampa Bay area as well as predict potential impacts on *Thalassia testudinum* populations. A preliminary hypothetical model developed during the Florida Bay study was used to explain and predict impacts in the eastern Gulf of Mexico and Tampa Bay where sample sizes were much lower. Infection levels comparable to those found in Florida Bay where substantial seagrass losses have occurred were found in the eastern Gulf of Mexico and Tampa Bay. We propose that long-term careful monitoring of *Labyrinthula* sp. in *T. testudinum* be carried out in estuaries other than Tampa Bay, especially those with environmental stresses.

INTRODUCTION

The protist, *Labyrinthula*, is a parasite of the subtropical seagrass *Thalassia testudinum* Banks ex König (turtle grass). Data from 3 different studies were used to evaluate the distribution of this marine slime mold in Florida Bay, the eastern Gulf of Mexico, and the Tampa Bay area as well as to predict potential impacts on its host populations. This parasite had been proposed as a possible cause of a massive acute seagrass die-off in Florida Bay that began in the summer and fall of 1987. No definitive cause(s) of this acute die-off was ever determined, but many possible etiologies were proposed, including high temperatures and salinities, overdeveloped seagrass beds, elevated sediment sulfide levels, hypoxia, and disease (Porter and Muehlstein, 1989; Robblee et al., 1991; Durako and Kuss, 1994; Carlson et al., 1994). "Wasting disease" in the eelgrass *Zostera marina*, which decimated the seagrass beds of Europe and North America in the 1930s and 1940s had previously been shown to be caused by a species of *Labyrinthula*. Some of these seagrass beds took 40 years to recover. Such a possible outcome in Florida made it important to attempt to determine the distribution pattern of *Labyrinthula* in *T. testudinum* populations in different Florida estuaries with varying environmental conditions and try to understand this parasite's potential impacts on the health and survival of those seagrass beds.

METHODS

Data were collected in three separate studies of *Labyrinthula* distribution in different geographical areas. The most extensive data set was collected over the last 5 years in a study initiated in Florida Bay in 1995. Two other preliminary studies were also done, one in the eastern Gulf of Mexico, and the other in the Tampa Bay area. The same field methods were used in all 3 studies so that results could be easily compared. Results from the Florida Bay studies were extrapolated to propose potential impacts of this slime mold on *T. testudinum* populations in the other areas of Florida where less extensive studies of *Labyrinthula* sp. have been done.

Florida Bay Studies (1995–present)

Four principal questions were asked during our Florida Bay studies:

1. Did *Labyrinthula* have a role in the initial acute *Thalassia* die-offs (summer, 1987)?
2. Is *Labyrinthula* involved in the chronic

die-off that we have been monitoring since the beginning of this study (1995-present)?

3. If it is involved, what role does *Labyrinthula* play in the chronic die-off?
4. Does *Labyrinthula* have a role in the current (first noticed in summer, 1999) acute die-off in Barnes Key?

Methods. Data from four years of biannual sampling in Florida Bay were examined to determine the relationship between the distribution and abundance of the seagrass *T. testudinum* and *Labyrinthula* sp. Ten basins with varied physical characteristics were studied intensively, including microscopic examination of thousands of *T. testudinum* blades from more than 2,500 sites within these basins. We used ArcView's extension Spatial Analyst and the Inverse Distance Weighted (IDW) method to visualize the pattern of and changes in distribution and abundance of infection in *T. testudinum* (Blakesley et al., 1999a).

Results. Both lab and field studies show that ongoing low salinities prevent *Labyrinthula* sp. from infecting *T. testudinum.* Field studies also suggest that a drop in salinity to below 15 ppt will reduce the existing level of infection. The data collected both during this field study and from associated laboratory studies (Blakesley et al., 1998) resulted in the formulation of a preliminary hypothetical model (Figs. 1 and 2) describing the effects of *Labyrinthula* sp. on *T. testudinum* populations in Florida Bay (Blakesley et al., 1999b). Where seagrass densities are low, *Labyrinthula* sp. does not cause major mortality. In moderate to high salinities and high seagrass densities, *Labyrinthula* sp. plays a major role in seagrass mortality. With optimal conditions for seagrass, *Labyrinthula* sp. can be a primary pathogen controlling seagrass densities. In suboptimal conditions for seagrass, such as lowered light levels, stressed seagrass may be weakened by opportunistic *Labyrinthula* sp. that further contributes to chronic seagrass die-off.

Discussion. The theoretical model suggests 3 different roles that *Labyrinthula* sp. might play in Florida Bay under different environmental conditions. These include: (1) a nonpathogenic parasite; (2) an opportunistic secondary pathogen; and (3) a primary pathogen. Five different factors are considered to be critical elements in determining the role(s) of *Labyrinthula* sp. in seagrass health at a particular site in Florida Bay (Blakesley et al., 1999c). Salinity controls infection (infection does not occur at <15 ppt). Seagrass density determines the extent to which *Labyrinthula* sp. infection spreads because the slime mold transmission is thought to depend on blade-to-blade contact (Muehlstein, 1992). Pathogenicity of a particular strain of *Labyrinthula* sp. will determine severity of infection. Environmental stressors (abiotic factors) such as low light or high temperatures may weaken *T. testudinum* and, in combination with the infection by pathogenic *Labyrinthula* sp., cause seagrass die-off. Resistance to disease due to genetic factors or production of phenolic compounds may be important in determining the health of *T. testudinum* in Florida Bay. The model predicts that in areas with high seagrass density, high salinity, suboptimal seagrass conditions (environmental stress), and presence of pathogenic *Labyrinthula* sp., the slime mold could contribute to either chronic or acute die-off acting as an opportunistic secondary pathogen. With the same conditions, but without environmental stress, it suggests that *Labyrinthula* sp. can still cause thinning or patchy die-off acting as a primary pathogen (Blakesley et al., 1999c)

Barnes Key (in Florida Bay) Die-Off

Introduction. In late summer 1999, a new

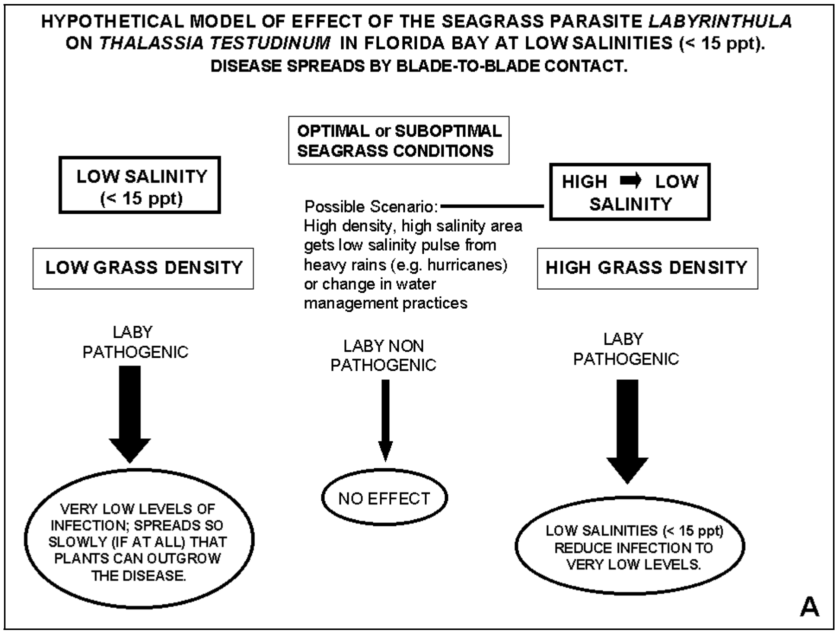

Figure 1. Hypothetical model describing the relationship between *Labyrinthula* infection of *Thalassia* and *Thalassia* mortality in Florida Bay in both high and low-density seagrass beds when either salinities are low or salinities were high and were then lowered to <15 ppt. The effects are the same in either stressed or unstressed seagrass conditions.

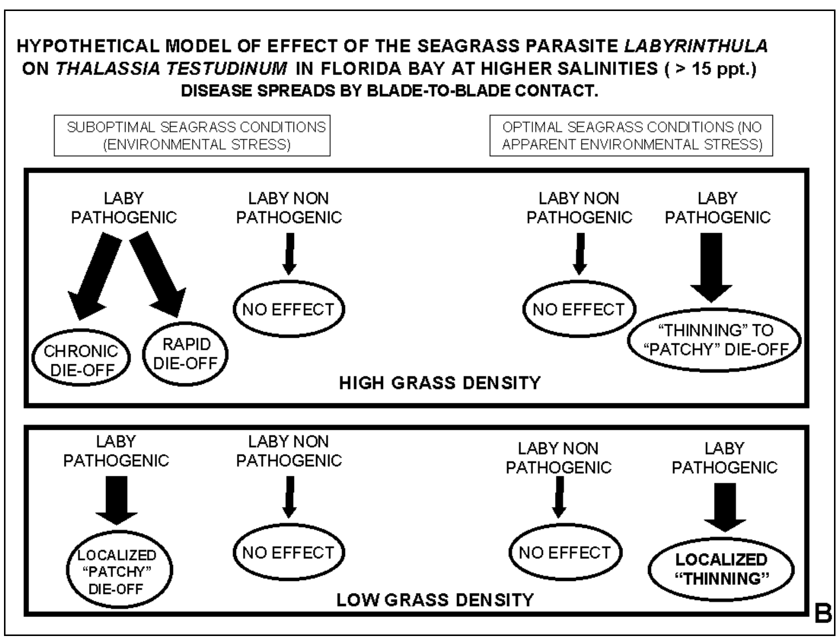

Figure 2. Hypothetical model describing the relationship between *Labyrinthula* infection in *Thalassia* and *Thalassia* mortality in Florida Bay at salinities >15 ppt. in stressed and unstressed conditions and in high and low seagrass densities.

seagrass die-off was first noticed north of Barnes Key that resembled the acute die-off of 1987, but was very different from the chronic die-off we had been studying since 1995. As in the acute event of 1987–89, *T. testudinum* appeared to be the only seagrass affected and the die-offs were occurring in dense, apparently "healthy" seagrass beds. Seagrass affected by the new acute die-off exhibited symptoms like those of the 1987 event, i.e. the lateral meristem tissue appeared to be the tissue most immediately affected (P. Carlson, personal communication). Meristem tissue seemed mushy and smelled like "mustard" while the rest of the blade looked green and healthy.

Methods. Beginning in early winter 1999, an investigation of the new acute seagrass die-off in Barnes Key was initiated to try to readdress the question first asked and unresolved 12 years before; why did the *T. testudinum* suddenly start dying in Florida Bay? Three 150 m transects were set up and monitored every 2–6 weeks. Changes in vegetative cover and infection by *Labyrinthula* sp. were recorded and sediment sulfide levels in different vegetative categories were monitored. In addition a total of 9 die-off patches were marked and monitored.

Results and Discussion. Preliminary data from our investigation of this event show that *Labyrinthula* sp. is probably not the initial cause of the acute die-off but instead appears to be an opportunistic secondary pathogen. There may be a seasonal component; our extensive data from Florida Bay, both field and laboratory, indicates that *Labyrinthula* sp. activity is related to temperature. When we first evaluated the site, in January, *Labyrinthula* sp. was rarely present anywhere in the Barnes Key area and this remained the case in repeated sampling through July. In September, we first noticed *Labyrinthula* sp. lesions, but only around old die-off patches. In November, we recorded the highest levels of *Labyrinthula* sp. infection since our studies began in Florida Bay in 1995. Infection was found in the margins of both active and inactive die-off patches as well as in the dense beds around those patches. Additional studies of this area are needed to follow the progression of the infection and die-off through time.

Apparent impacts of *Labyrinthula* sp. on *T. testudinum* seagrass beds in Florida Bay are summarized below:

- In low salinity areas (<15 ppt): **no impact**
- In dense beds with no apparent environmental stress (and salinity >15 ppt): **thinning, patchy die-off**
- In dense beds with environmental stress (and salinity >15 ppt): **chronic patchy die-off**
- In areas that have experienced an acute die-off (environmental stress present, and salinity >15 ppt): **severe loss**

Eastern Gulf of Mexico Studies (1997–1999)

Methods. In this study of 10 sites in the eastern Gulf, 10 *T. testudinum* shoots from 30 stations at each site were evaluated annually for lesion coverage and *Labyrinthula* sp. infection. The sites were chosen based on perceived environmental stress such as salinity fluctuations, thermal stress from power plants, and urban stresses, including several sites categorized as pristine.

Results. Results are shown in Figure 3. Infection levels were high at all sites except Charlotte Harbor and Perdido Bay; these sites had salinity fluctuations (Charlotte Harbor, 10–25 ppt; Perdido Bay, 12–34 ppt). Most other sites (including Tampa Bay) had infection levels higher than those found in the one Florida Bay site in this study (Rabbit Key basin) where seagrass losses have been substantial in the past. These results led to a request by the Tampa

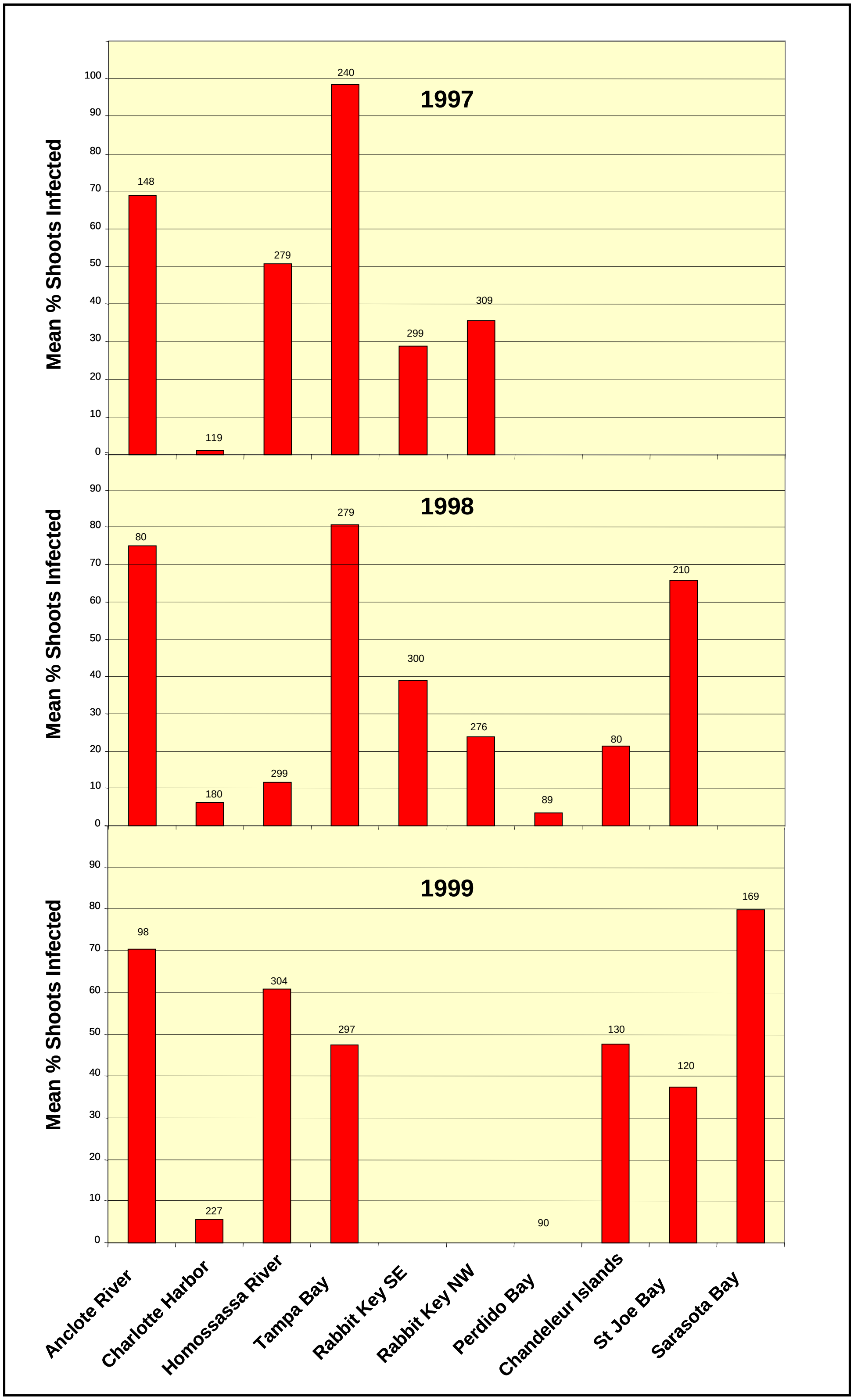

Figure 3. Mean percent *Thalassia* shoots infected with *Labyrinthula* at 10 sites along the east coast of the Gulf of Mexico.

Bay Estuary Program to begin a monitoring study of *Labyrinthula* sp. in the Tampa Bay area in the fall of 1999.

Discussion, Potential impacts of *Labyrinthula* sp. on *T. testudinum* in the Eastern Gulf of Mexico were extrapolated based on our data from these 10 sites and the information gained in the Florida Bay studies. The impacts are not now known, but the few data from this study suggest that the role(s) of *Labyrinthula* sp. in these areas may be similar to that found in Florida Bay. In summary:

- Low salinity sites (Charlotte Harbor, Perdido Bay): **no impact?**
- Environmentally stressed sites (Anclote River, Homossassa River, Tampa Bay, Rabbit Key, Sarasota Bay): **acute or chronic die-off?**
- Pristine sites (Chandeleur Islands, St. Joe Bay?): **thinning to patchy die-off?**

The St. Joe Bay site, which is categorized as pristine, may actually be experiencing some stress from elevated sediment sulfide levels caused by sea urchin grazing in the area (Paul Carlson, public communication, this meeting). Alternately, the *Labyrinthula* sp. may have been able to invade the blade tissue more easily because of mechanical damage caused by the urchins, with a subsequent indirect increase in sediment sulfide levels (microbial activity produces sulfide during the decay of below ground tissue from sick and dying plants). The Chandeleur Islands site may be a site similar to the Sunset Cove site in Florida Bay, where we hypothesize that *Labyrinthula* sp. plays the role of primary pathogen, thinning the dense seagrass beds and causing patchy die-off.

Tampa Bay Area (1999)

Methods and Results. Samples were collected during the fall 1999 Tampa Bay Estuary monitoring program to begin looking at the distribution of *Labyrinthula* sp. and severity of lesions in *T. testudinum* in this area. The data are very limited, but our analysis of 5-10 shoots randomly collected from 32 transects showed that most of the sites were positive for *Labyrinthula* sp. (Fig. 4). Results of our monitoring studies show that the largest number of transects infected was in Old Tampa Bay (Fig. 5), but the highest percentage of shoots infected was in Clearwater (Fig. 6). Infection levels at all of the sites except the 3 transects in Sarasota Bay were either comparable or higher than those found in the western parts of Florida Bay where most seagrass losses have occurred. Potential impacts of *Labyrinthula* sp. on *T. testudinum* populations in the Tampa Bay area could range from no impact, through seagrass losses resulting in beneficial thinning of overdeveloped beds, to chronic or acute patchy die-off.

DISCUSSION

In summary, the distribution and potential impacts of *Labyrinthula* sp. infection on *T. testudinum* populations depend on a suite of interacting factors (salinity, seagrass density, pathogenicity, environmental stressors, seagrass resistance to disease). All of these factors, as well as others that may as yet not have been identified, need to be taken into consideration before the potential impacts of *Labyrinthula* sp. infections on *T. testudinum* populations in any particular geographic area can be predicted. The roles of *Labyrinthula* sp. in seagrass health in Florida Bay have been studied for 5 years. We are presently testing our model in Florida Bay in two areas (Barnes Key and Sunset Cove) with different environmental conditions, one with acute die-off and the other with chronic die-off. Only very preliminary *Labyrinthula* sp. and lesion distribution data have been collected elsewhere in the state.

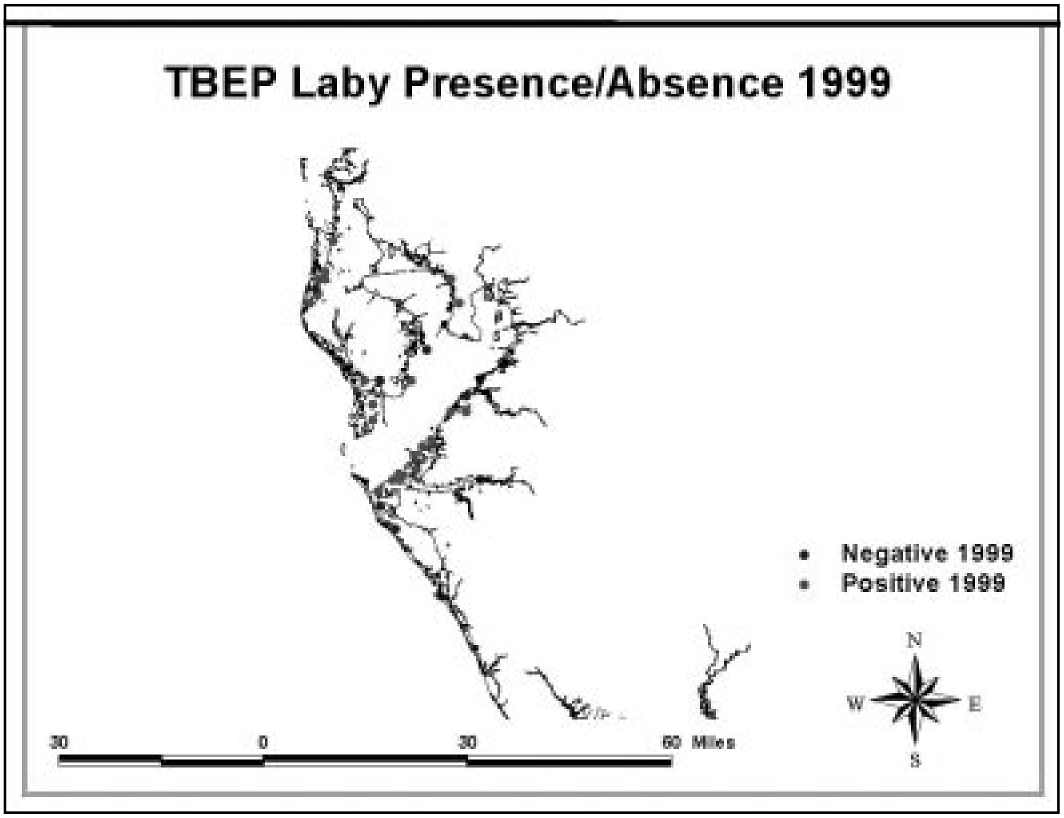

Figure 4. Map of sites sampled in the Tampa Bay area in the fall of 1999. Positive sites contained lesioned grass infected by *Labyrinthula* sp. cells.

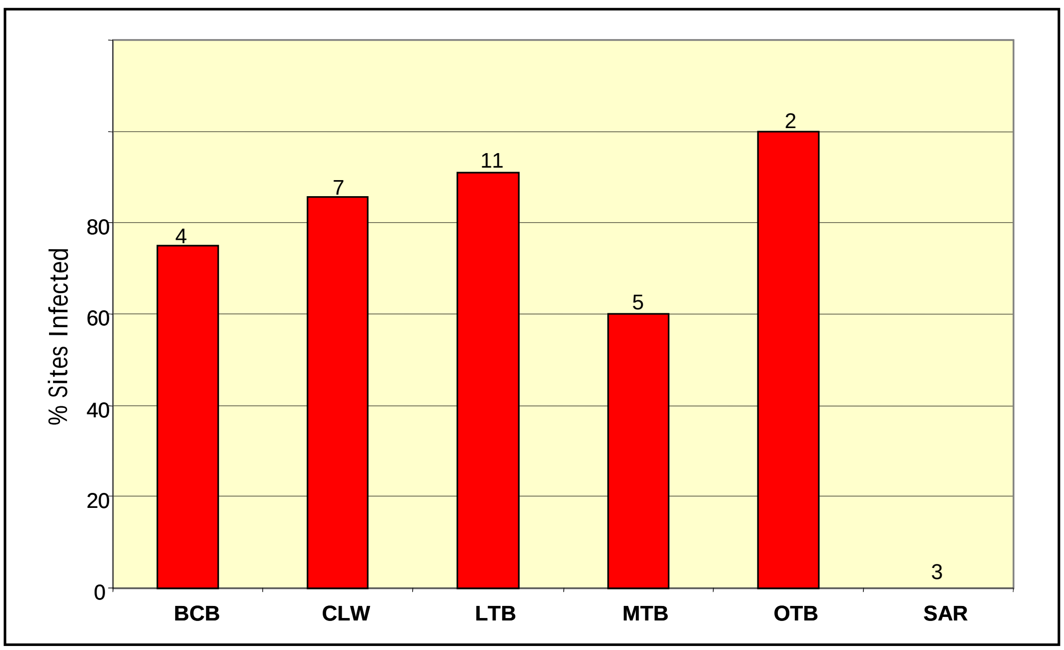

Figure 5. Percent of transects with *Thalassia* shoots infected with *Labyrinthula* in the Tampa Bay area in fall 1999. BCB = Boca Ciega Bay; CLW = Clearwater; LTB = Lower Tampa Bay; MTB = Middle Tampa Bay; OTB = Old Tampa Bay; SAR = Sarasota Bay.

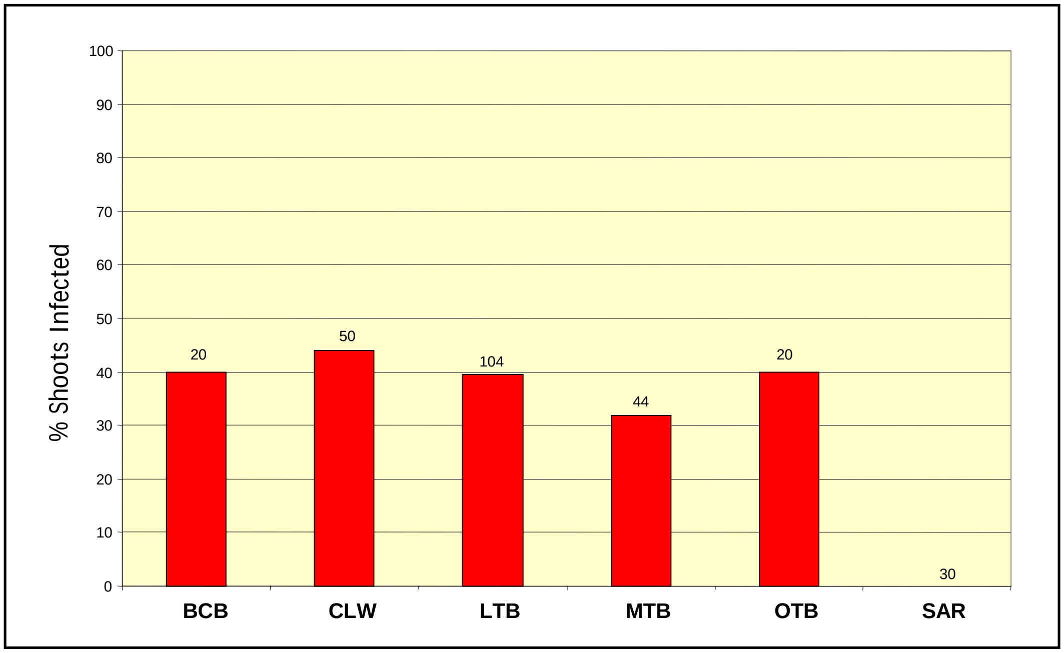

Figure 6. Percent of *Thalassia* shoots infected with *Labyrinthula* in the Tampa Bay area in fall 1999. BCB = Boca Ciega Bay; CLW = Clearwater; LTB = Lower Tampa Bay; MTB = Middle Tampa Bay; OTB = Old Tampa Bay; SAR = Sarasota Bay.

Long-term careful monitoring of *Labyrinthula* should be carried out in estuaries other than Florida Bay, especially in those with environmental stresses. The dynamics of *Labyrinthula* sp. distribution must be more clearly understood before the impacts of this slime mold on seagrass populations can be predicted. Seagrass recovery in urban estuaries must include health evaluations of seagrass beds to insure that gains in seagrass coverage can be maintained over time.

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SEAGRASS MAPPING: ACCURACY ISSUES

Raymond C. Kurz

ABSTRACT

Seagrasses have been used as living resource indicators of water quality and ecosystem health in a number of estuaries throughout Florida, including Tampa Bay. The assessment of mapping accuracy has recently become an important issue since the magnitude of seagrass change between 1992 and 1996 in Tampa Bay has declined and may be approaching or within the range of error of the actual mapping process. This paper discusses this issue and provides recommendations to assess mapping accuracy for future seagrass mapping projects conducted in southwest Florida.

INTRODUCTION

The seagrass mapping program currently funded and administered by the Surface Water Improvement and Management (SWIM) Section of the Southwest Florida Water Management District (SWFWMD) was developed to provide accurate measures of submerged aquatic vegetation (SAV) coverage for several important estuaries in southwest Florida. This data has been used to develop water quality and resource management goals and strategies for Tampa Bay, Sarasota Bay, and Charlotte Harbor. Mapping began in 1988 based on methodology developed by staff from the SWFWMD and Geonex Corporation (Kurz et al., 2000).

The goal of this paper is to identify and discuss issues related to the precision and accuracy of the current mapping methodology and to develop techniques for assessing tolerances for future mapping efforts. As the rate of increase of seagrass coverage in Tampa Bay has slowed during the past four years, the ability to distinguish actual SAV increases or declines from mapping error will be critical to the agencies and their staff who depend on this data for making important resource management decisions.

CURRENT METHODOLOGY

The basic mapping process used between 1988 and 1996 involved acquiring 1:24,000 scale aerial photography, identifying signatures (in the field) of seagrass beds, macroalgae, bare bottom, and hard bottom communitites, photointerpreting those signatures from the photography, and digitizing polygons representing seagrass coverage to create seamless maps for the study area (nearshore waters from Tampa Bay to Charlotte Harbor) (see Kurz et al., 2000, for a more detailed description of this methodology). The seagrass polygons were originally classified as one of three types: sparse, patchy, or dense. In 1990, only two classes were used due to difficulties in differentiating between sparse and patchy coverage. Those classes were patchy (>25% of polygon unvegetated) or continuous (<25% of polygon unvegetated). A quality control program to verify the accuracy of identifying the presence/absence and correct classification (patchy vs. continuous) of seagrass signatures was performed during each mapping period. A 90% accuracy rate was required for polygons greater than 1.0 acre in size. Finally, maps and data output are produced for analysis of trends.

This methodology was refined for the most recent mapping effort in 1999 and is generally similar to the previous methods. The process now includes the following changes to increase spatial accuracy (David Tomasko, personal communication):

- Collection of georeferenced aerial photography (at a scale of 1:12,000)
- Delineation of SAV polygons from

the aerial photography directly into Arc/Info using a stereoplotter

ISSUES RELATED TO ACCURACY OF CURRENT SEAGRASS MAPPING TECHNIQUES

A number of issues have emerged regarding the accuracy of current mapping techniques used to map seagrasses. These issues are not restricted to this particular project and have been explored by a number of other authors (Dicks and Lo, 1990; Foote and Huebner, 1995).

For this paper, the following questions were developed with respect to the assessment of seagrass acreage in southwest Florida estuaries:

How much error exists for actual seagrass area (coverage) based on the existing SWFWMD mapping techniques (digitization of SAV signatures)?

Current seagrass mapping results in data that are often presented as total number of acres per year, however, error bars for each year of mapping have never been calculated due to questions concerning overall mapping error rates (Figure 1). Due to the high cost to produce both extremely accurate and precise maps, multiple trials (i.e., to produce a n>1 for the purpose of performing statistical comparisons among years) to remap the entire study area are not feasible. However, it may be possible to perform accuracy assessments based on a subpopulation of seagrass polygons. A trial mapping exercise to estimate potential polygon delineation error was performed by the author and is presented in Figure 2. The differences in seagrass acreage between multiple delineations of the same seagrass bed indicate an inherent and measurable error rate associated with this mapping technique. Although this exercise is not an independent test of the actual error rate, it does illustrate that the delineation of a particular seagrass bed is not free from error.

How accurate are seagrass maps based on the current classification system (patchy vs. continuous) since not every identifiable SAV patch is delineated?

The following figure depicts seagrass polygons for the 1994 mapping period overlaid onto the 1994–1995 digital orthophoto quarter quad (DOQQ) in a

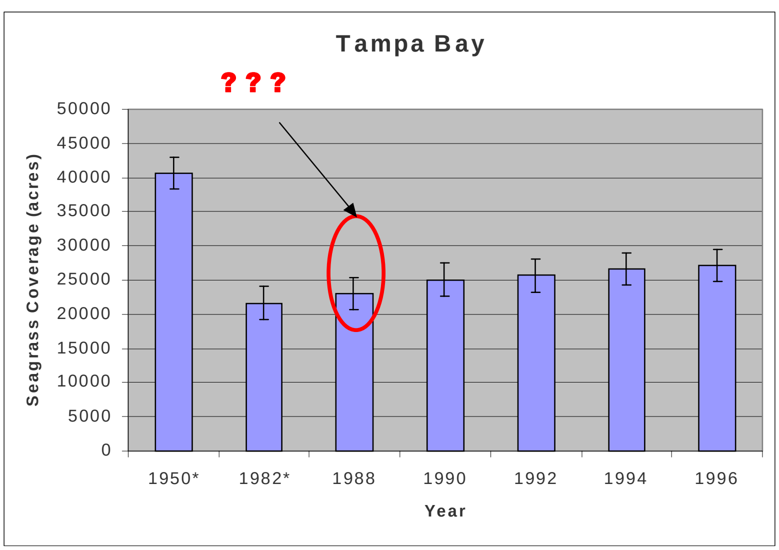

Figure 1. Seagrass coverage estimates between 1950 and 1996. Current techniques do not result in an estimation of error or standard deviation.

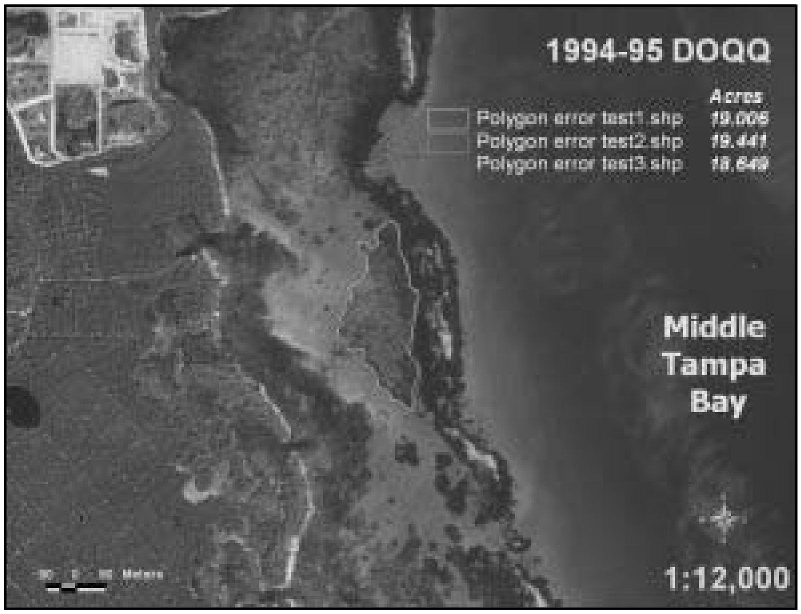

Figure 2. Delineation of a seagrass bed in Middle Tampa Bay multiple times. Note multiple delineation trials result in different acreages.

portion of Middle Tampa Bay (Figure 3). The graph following this figure (Figure 4) depicts a hypothetical scenario for a single seagrass polygon from Figure 3 over time. In 1988 when the seagrass bed is initially delineated, an acreage value was derived for that polygon. In 1990, the seagrass bed decreases in size at its interior due to some biotic (e.g., stingray foraging disturbance, *Labyrinthula* parasitism) or abiotic factor (e.g., salinity change) resulting in open bottom patches. Since the methodology for delineating seagrasses in 1990 is basically a "change analysis," the outer boundary of the polygon does not change even though the actual acreage of seagrass making up the polygon may have declined.

POTENTIAL METHODS FOR ASSESSING MAPPING ACCURACY

Since seagrass beds are delineated manually into shapes or polygons by photo interpretation of aerial photography, the shape drawn around a bed can vary both between individual photointerpreters and also by a single individual since they are estimating a perimeter based on a visual representation. As the current measure of accuracy involves only classification error, it is still unclear as to how much inherent error exists in the physical mapping of seagrass polygons. To measure the error of polygon size (i.e., "this bed is 5.0 acres ± 0.02 acres"), various photointerpreters could be given a representative seagrass bed and each asked to digitize the boundary (e.g., as performed in Figure 2). The variation in the area delineated by each photointerpreter could be averaged and a standard deviation or standard error computed based on some number of photointerpreters (e.g., n value). Another measure of error could include a number of repetitive delineations by the same photointerpreter (since the same person typically maps a large number of seagrass polygons within the project boundary). To get an even better feel for the range or accuracy of this error rate, that same person could be asked to repetitively delineate seagrass beds of various sizes such as small (1.0 acre or less), medium (5.0 acres or less), and large (20.0 acres or less).

Using existing seagrass data, GIS staff could generate a list of all seagrass polygons by size which could then be analyzed to determine the range of seagrass

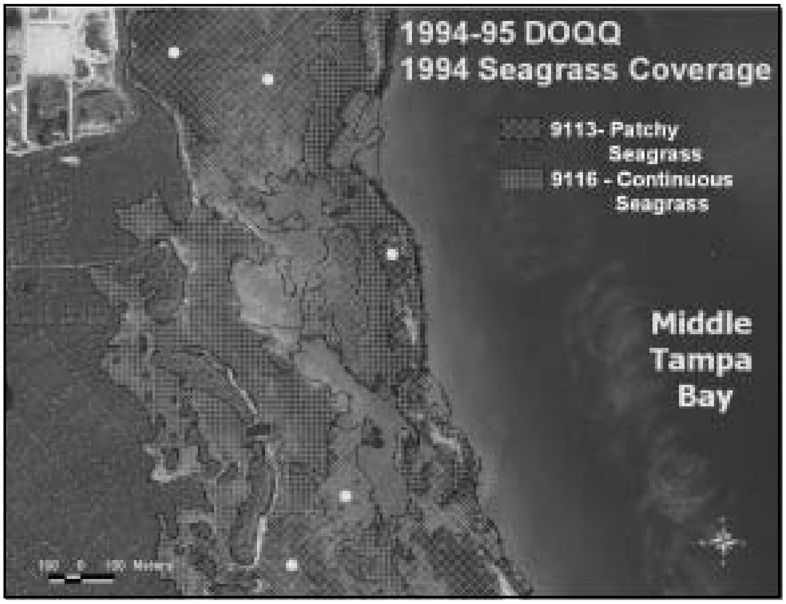

Figure 3. Seagrass polygons in Middle Tampa Bay.

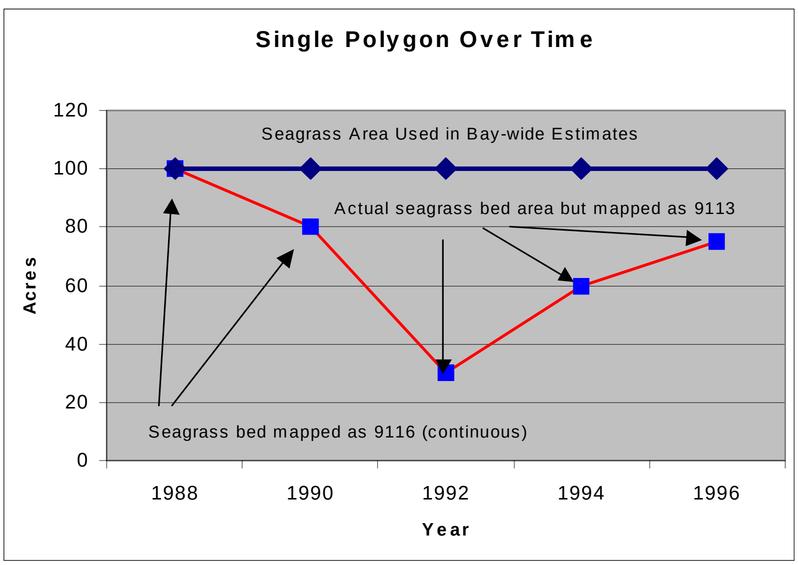

Figure 4. Trends in a single seagrass polygon over time. Note that even though the area of the polygon used to calculate baywide seagrass coverage for a given year may not change, the actual seagrass area within a given polygon may have changed significantly.

bed sizes (and also the range by density—patchy vs continuous) as well as the proportion of beds within various size ranges. This data could then be used to determine what size intervals should be tested and serve to refine whether mapping smaller beds is easier and more accurate than larger beds or whether mapping continuous beds is more accurate than patchy beds (which might be the case since the edge of the bed is typically more easily defined in a dense bed than a patchy one).

This measurement or "calibration" should be performed at the start of each mapping project and, possibly midway and at the end of the project to determine if error rates decrease over time. However, if funding to perform these additional assessments is limited, these tests may only need to be done once (during the initial mapping of all new seagrass polygons) since subsequent trend mapping only involves changes to existing polygons.

CONCLUSIONS

A number of techniques could be used to estimate seagrass acreages and ranges of acreages based on an estimate of accuracy/ error in the current mapping techniques. A few examples that were proposed above have been incorporated into SWFWMD's future mapping efforts (D. Tomasko, personal communication).

In addition to these very specific mapping technique assessments, the following recommendations may improve our interpretation and use of this important living resource data:

1. Researchers should evaluate different methods for reporting data that are useful to resource managers (e.g., changes in patchy to continuous and continuous to patchy rather than just overall acreage changes estuary-wide)
2. Increased focus on interpretation of classification (patchy vs. continuous) trends, spatial changes, and their causes (e.g., El Niño, hydrologic alterations, parasitism)
3. Acquisition of accurate bathymetry will be necessary to measure seagrass expansion/retreat from deeper waters (e.g., bay segment target depths) and related to changes in water quality (light penetration)
4. Stronger linkage between mapping and fixed transect monitoring data (e.g. finer scale definition of deep edge, explanation of spatial changes through field observations including species shifts which result in signature changes)
5. Continued interaction among researchers and resource managers to exchange information about seagrass mapping techniques and uses of data (e.g., through seagrass workshops which resulted in this collection of papers)

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THE INFLUENCE OF SEDIMENT SULFIDE ON THE STRUCTURE OF SOUTH FLORIDA SEAGRASS COMMUNITIES

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ABSTRACT

Sediment porewater sulfide affects the structure of South Florida seagrass communities because it is abundant, it is toxic, and because sulfide tolerance varies among seagrass species. In this paper, we present four datasets which illustrate the influence of sulfide on the structure and species composition of subtropical seagrass communities: 1. Porewater sulfide measurements made during seagrass die-off episodes in Florida Bay during 1990 and 2000 which show that porewater sulfide concentrations in active die-off areas (4-10 mM) far exceed normal concentrations throughout Florida Bay (<2 mM); 2. Field experiments which raised porewater sulfide concentrations up to 12 mM resulting in complete mortality of *Thalassia testudinum*, some loss of *Syringodium filiforme*, and almost no mortality of *Halodule wrightii*; 3. Guano-addition experiments performed in the early 1990's by Fourqurean et al. (1995), which we re-interpret to show that porewater sulfide concentration changes might contribute to species shifts from *Thalassia* to *Halodule* around bird roosting sites, and 4. Sediment sulfide concentrations measured in nine estuaries of the eastern Gulf of Mexico that show potentially toxic sediment sulfide concentrations in several estuaries.

We suggest that *Thalassia* is much less tolerant of high porewater sulfide concentrations than *Halodule* because *Thalassia* has a much higher ratio of below-ground:above-ground biomass. Any physical or chemical process which increases sediment sulfide concentrations can therefore cause sub-lethal, chronic stress which reduces *Thalassia* productivity. Higher concentrations can result in acute, lethal sulfide stress. Sulfide stress might also occur without changes in porewater sulfide concentrations if the oxygen balance of seagrasses is affected by decreased photosynthetic or oxygen transport capacity due to shading by phytoplankton, epiphytes, or resuspended sediments. Pathogens, physical injury, and grazing organisms can also reduce oxygen transport from seagrass leaves to roots and rhizomes, setting the stage for hypoxic stress and sulfide toxicity.

Recent measurements of porewater sulfide concentrations in several west Florida estuaries suggest that the problem of sulfide toxicity is not limited to seagrasses growing in carbonate sediments. In fact, sulfide toxicity might play a synergistic role in recent seagrass losses in Tampa Bay. Seagrass restoration efforts should consider sediment sulfide levels and sulfide tolerance of seagrass species in transplant projects.

INTRODUCTION

Although seagrasses are vital components of the nearshore ecosystem (Zieman 1982; Nelson 1992), drastic declines in the distribution and abundance of seagrass communities have occurred in many estuaries throughout the Gulf of Mexico (Lewis et al. 1982; Pulich and White 1991), the United States (Orth and Moore 1983; Dennison et al. 1993), and the world (Cambridge et al. 1986). In most cases, concurrent declines in water quality have been blamed for seagrass loss. Declines in water quality and clarity, in turn, often result from anthropogenic nutrients which stimulate estuarine phytoplankton blooms. In Tampa Bay, an 80% decline in seagrass cover over the past 100 years has been largely attributed to deterioration of water clarity (Lewis et al. 1982). Aggressive efforts to reduce nitrogen loading to Tampa Bay have resulted in improved water clarity and increases in seagrass cover between 1990 and 1996 (Johansson and Greening 2000).

Recent studies have shown that sediment sulfide concentrations can also act alone or synergistically to cause chronic, sublethal or acutely lethal stress on seagrasses (Carlson et al. 1994; Goodman et al. 1995; Erskine and Koch 2000). Sulfide is produced naturally in anaerobic marine sediments by heterotrophic bacteria which use sulfate as a terminal electron acceptor in breakdown of organic matter (Goldhaber and Kaplan, 1975). Because seagrass sediments typically have high organic matter content, sulfate reduction rates in seagrass sediments are higher than in unvegetated marine sediments (Carlson et al. 1994, Holmer and Nielsen 1997). Sulfide is also a potent cytotoxin, irreversibly binding enzymes involved in electron transport for both photosynthesis and respiration (see review by Bagarinao 1992). Sulfide also causes hypoxia in seagrass roots and rhizomes by reacting with photosynthetically-produced oxygen diffusing from leaves to below-ground tissue. Marine plants and animals vary in their ability to tolerate sulfide, using a variety of avoidance strategies to exclude sulfide and accommodation strategies to detoxify sulfide (see review by Bagarinao 1992). However, the tolerance limits of seagrasses can be exceeded if sulfide accumulates to toxic levels in sediment porewater.

The amount of sulfide which accumulates in seagrass bed sediments depends on a number of physical and chemical characteristics. Tidal currents, wave action, and sandy sediments facilitate exchange of sediment porewater with the overlying water column, resulting in oxidation or export of sulfide produced by bacteria. In contrast, sulfide concentrations are generally higher in quiescent areas with fine-grained sediments. Siliceous, terrigenous sediments typically contain high concentrations of iron (up to 5%) which bind sulfide as pyrite or iron monosulfide. Biogenic carbonate sediments, such as those in Florida Bay, the Florida Keys, and Biscayne Bay, however, have very low iron concentrations, resulting in porewater sulfide concentrations which are considerably higher than those typically found in Central and North Florida seagrass beds.

In keeping with the theme of this conference "Seagrass Management: It's not just nutrients," we suggest that sediment porewater sulfide concentrations also influence the species composition, survival, and growth of seagrass beds in South Florida. If this is the case, two important ramifications for seagrass management should be considered: 1. human activities which affect organic matter and sediment accumulation in seagrass beds might increase seagrass sulfide stress; 2. Sediment sulfide levels might affect the survival and growth of transplanted seagrass beds.

We draw on data from four projects to examine the role that sulfide plays in determining the species composition, survival, and growth of South Florida seagrass communities: Sediment sulfide data collected in studies of Florida Bay seagrass die-off over the last 12 years provide powerful, albeit circumstantial, evidence that high sulfide concentrations kill turtle grass. We also performed experiments in Florida Bay seagrass beds to examine the response of *Thalassia testudinum*, *Halodule wrightii*, and *Syringodium filiforme* to elevated sulfide concentrations, and we re-sampled experiments which examined the effects of bird guano on seagrass beds (Fourqurean et al. 1995). Finally, we compared sulfide concentrations in seagrass beds of nine estuaries located along the coast of the Gulf of Mexico from the Chandeleur Islands to Florida Bay.

STUDY AREA AND METHODS

Florida Bay Seagrass Die-Off Studies. We have measured porewater sulfide concentrations associated with die-off episodes of *Thalassia* since 1988. Two major episodes have occurred in that time period. The first episode occurred between 1987 and 1991, affecting over 10,000 ha. of seagrass beds, primarily in the central and western portions of Florida Bay (Robblee et al. 1991). The second episode has occurred since 1998 in a much smaller area near Barnes Key near the southern edge of Florida Bay, and it is still ongoing.

In the initial die-off studies, we collected small (60 cm^3) cores from three die-off patches in each of three basins affected by die-off (Rankin Lake, Johnson Key Basin, and Rabbit Key Basin). Three replicate cores were collected in February, June, and October each year from dead areas, fringes of die-off patches, and surviving seagrass surrounding the die-off patches. Cores were submersed in chilled seawater for transport, and porewater sulfide was determined in the laboratory using a sulfide ion-specific electrode (Carlson et al. 1994). Samples were collected from die-off patches in 1989 and 1990. Additional porewater sulfide samples were also collected from equilibrium-dialysis samplers deployed in a die-off patch and in an adjacent, surviving grass bed in Johnson Key Basin in fall 1990.

Porewater sulfide sampling was also carried out quarterly between 1994 and 1996 at 24 sites throughout Florida Bay using the same coring and analysis techniques. Quarterly sampling of porewater sulfide at the Barnes Key die-off site began in October 1999 and is continuing.

Sediment Perfusion Experiments. As part of our investigation into the role of sulfide in seagrass die-off in Florida Bay, we have carried out a series of experiments to determine the response of seagrasses to elevated porewater sulfide concentrations. We removed the bottoms from 15 5-gallon plastic buckets and inserted them into the sediments of dense, nearly-monospecific beds of *Thalassia testudinum*, *Halodule wrightii*, and *Syringodium filiforme* near Man O'War Key in the western bay in June 1990. Each bucket isolated a column of sediment and seagrass from the surrounding bed; rhizomes around the perimeter of each bucket were severed to prevent translocation between seagrass shoots inside and outside the buckets. After allowing seagrasses to recover from bucket insertion for one month, we randomly selected three buckets for each seagrass species to serve as bucket controls and three buckets to receive glucose amendments. Three "perfusers" (porous polyethylene tubes containing 140 g glucose) were inserted into the sediments of each glucose-amended bucket. Three pieces of PVC pipe (1" nominal diameter) were inserted into each of the control buckets. A porous polyethylene porewater sampler was also inserted in the center of each bucket to allow monthly collection of sediment porewater for sulfide and pH determination. Buckets were harvested eight weeks after the perfusers were installed in the buckets. All plant material retained on a 1 mm sieve was retained for determination of live and dead above- and below-ground biomass. Seagrass shoots were classified as live, new-dead, or old-dead, counted, and measured.

Bird Stake Experiments. Several stakes were installed as bird roosts on Cross Bank by J. Fourqurean and G. Powell in 1981 (Powell et al. 1989, Fourqurean et al. 1995). Approximately two years after stakes were installed, five stakes were pushed down into the sediment so they could no longer serve as bird roosts. In July 1991, ten years after the experiment began, we collected duplicate sediment cores adjacent to each of twelve stakes: four active bird roosts, four stakes which had been pushed into the sediments, and four control sites. Cores were transported in chilled seawater and analyzed using a sulfide ion-specific electrode as described above.

Comparison of Gulf Coast Estuaries. Porewater sulfide concentrations were measured at 25–30 sites in nine estuaries distributed between the Chandeleur Islands and Florida Bay. At each site, we collected 60-cm^3 cores and measured sulfide using an ion-specific electrode.

RESULTS AND DISCUSSION

Florida Bay Seagrass Die-Off Studies. Vertical profiles of sediment porewater sulfide concentrations of surviving *Thalassia* beds and die-off patches at Johnson Key Basin differed in several ways (Figure 1). Sulfide concentrations were low near the sediment surface and increased with depth in the sediments for both profiles. In the surviving grass bed, however, maximum sulfide concentrations ~3 mM occurred at a depth of 8–10 cm, corresponding roughly to the depth where most rhizomes are located. Sulfide concentrations in the die-off patch increased to values near 1.5 mM at a depth of 20 cm in the sediments. Low porewater sulfide concentrations of the die-off patch sediments probably resulted from depletion of seagrass organic matter in the year that elapsed between the time that seagrass died and the time we sampled. Previous studies have shown that sediment sulfide concentrations increase dramatically during die-off episodes and then decline slowly over 12 to 18 months (Carlson et al. 1994).

During initial die-off investigations in 1989 and 1990, mean porewater sulfide concentrations in sediments of Johnson Key Basin, Rabbit Key Basin and Rankin Lake were 1.6 to 1.8 mM (Table 1A). Concentrations varied seasonally with concentrations generally lower than 1 mM in spring and summer. Concentrations in

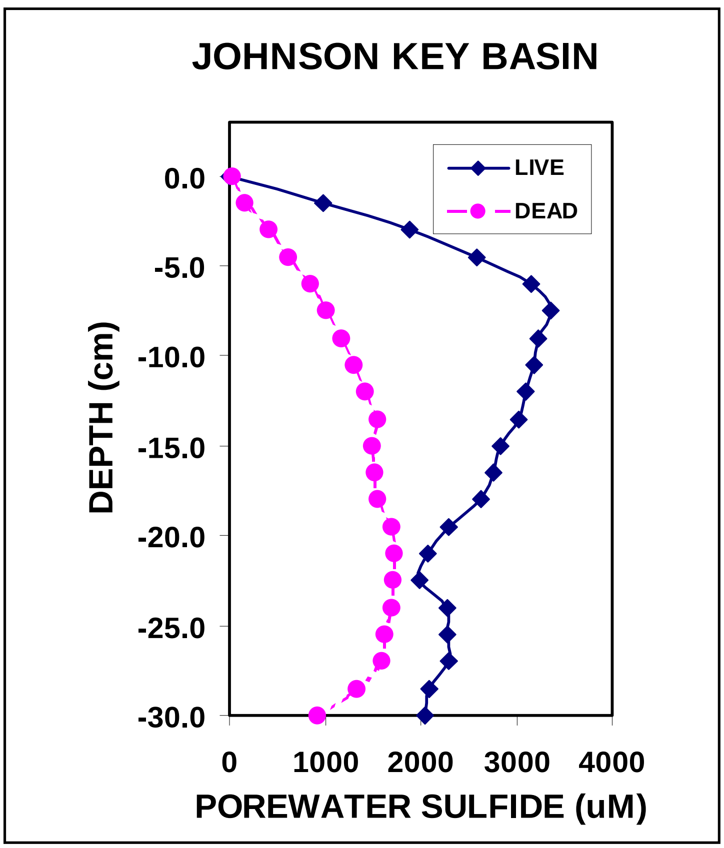

Figure 1. Vertical profile of porewater sulfide in die-off patch sediments and adjacent surviving *Thalassia* testudinum bed, Johnson Key Basin, fall 1990.

Table 1. Florida Bay sediment porewater sulfide concentrations,1989 to 1999. Data are mM (mmol S= per liter porewater). Within each grouping (season, year, or region), values with the same letter subscript are not significantly different. Barnes Key 1999 data are means followed by standard deviation values in parentheses.

A. Initial Die-Off Period 1989–1991		
Basin averages	Johnson Key Basin	1.7 (1.4)
	Rabbit Key Basin	1.6 (1.2)
	Rankin Lake	1.8 (1.7)
Seasonal averages	June 1989	0.52 (0.09)
	October 1989	2.30 (0.69)
	February 1990	0.86 (0.34)
	June 1990	0.88 (0.24)
	October 1990	3.80 (0.89)
B. Baywide averages 1994–1996		
Annual averages	1994	0.62 a
	1995	0.59 a
	1996	0.71 a
Seasonal averages	Spring	0.54 b
	Fall	0.73 a
Regional averages	North-central	0.10 a
	Western	0.75 b
	Southeast	0.48 c
	Northeast	0.47 c
	South-central	0.47 c
Barnes Key site	May 1994	0.53
	October 1994	0.90
	May 1995	0.66
	October 1995	0.56
	April 1996	0.43
	September 1996	0.76
	May 1997	0.68
C. Barnes Key die-off site, October 1999		
	Old die-off	2.70 (0.92)
	Dense *Thalassia*	3.30 (0.55)
	New die-off	4.70 (1.10)

October of 1989 and 1990 were 2.3 and 3.8 mM, respectively, coinciding with *Thalassia* die-off events. In active die-off patches, Carlson et al. (1994) measured porewater sulfide concentrations over 13 mM. Although much of the sulfide production in active die-off patches is fueled by microbial decomposition of dying roots and rhizomes, elevated porewater sulfide concentrations (1.5–1.8 mM) preceded die-off by three months at one monitored site in Johnson Key Basin in 1990, indicating that events which stimulated sediment sulfide production in summer might have caused die-off directly or made *Thalassia* vulnerable to other stressors.

Between 1994 and 1996, no large seagrass die-off episodes occurred in Florida Bay and porewater sulfide concentrations sampled at 24 sites throughout the Bay ranged between 0.47 mM and 1.1 mM (Table 1B). Highest concentrations occurred in the north-central region of the Bay (1.1 mM), followed by the western region (0.75 mM). The southeast, south-central, and northeast regions all had porewater sulfide concentrations less than 0.5 mM.

These values were significantly lower than mean concentrations measured in the western Bay during the first major die-off episode (Table 1A). During this same period, surface water salinity in Florida Bay dropped from hypersaline values to 25–35 ppt (Everglades National Park Marine Monitoring Program, unpublished data). Although the Barnes Key die-off episode has cast doubt about the importance of hypersalinity as a contributing cause for *Thalassia* die-off episodes, the drop in porewater sulfide values from 1.8 to 1.1 mM in the north-central region and a decline from 1.7 to 0.75 mM in the western Bay over the time period between 1989–90 and 1994-96 is statistically significant. The drop might indicate a regional-scale influence—climatic, hydrographic, or biological—which had contributed to *Thalassia* die-off in 1987–91 declined during this period.

During the period 1994–96, annual variation in porewater sulfide concentrations was not significant, but fall values (0.73 mM) were significantly higher than spring (0.54 mM). Both sampling periods (1989–90 and 1994–96) exhibited the same seasonal trends suggesting that processes operating in the fall cause elevated porewater sulfide concentrations. As noted above, *Thalassia* die-off episodes have also occurred during the fall suggesting a link between the seasonal dynamics of sediment sulfide and *Thalassia* die-off. The Bay-wide survey data support this link, but they also indicate that a moderate rise in porewater sulfide concentrations during fall is not sufficient by itself to cause die-off. The processes that cause the seasonal cycle in porewater sulfide concentrations are not known but Yarbro and Carlson (1989) suggested that hypoxic stress of *Thalassia* might occur in fall as the combined result of declining day length and warm water temperatures.

During the 1994–96 period, porewater sulfide concentrations at one sampling point near Barnes Key fluctuated between 0.43 and 0.9 mM. When a new episode of *Thalassia* die-off occurred in this area in fall 1999, porewater sulfide concentrations over 6 mM were measured (Table 1C). Even surviving *Thalassia* beds had porewater sulfide concentrations over 3 mM, suggesting the potential for sulfide stress, although no visible stress symptoms were observed.

Sediment Perfusion Experiments. Addition of glucose to bucket sediments caused significant increases in porewater sulfide, indicating that sulfate reduction rates in Florida Bay sediments are limited by the availability of labile organic matter rather than by sulfate. Initial porewater sulfide concentrations measured May 16, 1991 ranged from 0.8 to 1.2 mM with no significant difference among species or treatments (Table 2). Final concentrations varied markedly among treatments. Glucose-amended buckets of all three species had porewater sulfide concentrations between 10 mM and 13 mM. Concentrations within control buckets ranged from 2 mM in *Halodule* beds to 2.5 mM in *Syringodium* beds, to 3.5 mM in *Thalassia* beds. Sediments outside the buckets had the lowest sulfide concentrations, generally less than 1 mM.

Seagrass survival was inversely related to sediment sulfide concentrations (Table 2). Lowest mortality (5%) occurred in outside control plots in *Thalassia* beds and in control buckets in *Halodule* beds. Mortality in control buckets in *Syringodium* and *Thalassia* beds (8% and 15%, respectively) was slightly higher, indicating that the process of severing rhizomes and installing buckets does kill some seagrass shoots.

Seagrass mortality in glucose-amended sediments of *Halodule* and *Syringodium* beds (5% and 12%, respectively) was slightly, but not significantly, higher than controls. *Thalassia* mortality in glucose-amended buckets was approximately 70%, much higher than in control buckets. We interpret these results to indicate that *Thalassia* is very sensitive to elevated sulfide concentrations, *Syringodium* is less sensitive, and *Halodule* is relatively insensitive to elevated sulfide concentrations in the range of 10–13 mM, values which were measured in sediments of active die-off patches.

One difference among these three seagrass species which might account for their differing sulfide tolerance is their below-: above-ground biomass ratios. Species which have a proportionally large investment in belowground tissue might be more vulnerable to sulfide toxicity than a species with less belowground biomass. Up to 80% of *Thalassia* biomass is roots and rhizomes, while *Syringodium* and *Halodule* roots and rhizomes typically comprise less than 30% and 20% respectively of total plant biomass (Kenworthy and Thayer 1984). In our experiments, *Thalassia*, the species with the greatest fraction of

Table 2. Porewater sulfide concentrations and seagrass shoot mortality in sediment perfusion experiments, summer 1991. Shoot mortality is calculated as the percent of shoots present at the beginning of the experiment within each bucket.

PARAMETER		TREATMENT: OUTSIDE CONTROLS	CONTROL BUCKETS	GLUCOSE-AMENDED BUCKETS
Initial Porewater Sulfide Concentration (mM)	*Thalassia* beds	—	1.2	1.2
	Syringodium beds	—	1.0	1.0
	Halodule beds	—	0.8	0.8
Final Porewater Sulfide Concentration (mM)	*Thalassia* beds	1.0	3.5	13.0
	Syringodium beds	—	2.5	11.0
	Halodule beds	—	2.0	10.0
Shoot Mortality	*Thalassia* beds	5%	15%	70%
	Syringodium beds	—	8%	12%
	Halodule beds	NA	5%	5%

belowground biomass, was the most vulnerable to sulfide toxicity. *Halodule*, the species with the lowest ratio of below-:above-ground tissue, was least affected by elevated sediment sulfide concentrations.

Our results are consistent with those of Goodman et al. (1995) who found that photosynthesis and growth of *Zostera marina* was inhibited by experimentally-elevated sulfide concentrations. Our data are also remarkably consistent with Terrados et al. (1999), who found that sucrose addition to sediments in a Phillipine seagrass bed resulted in significant mortality and slower growth in *Thalassia hemprichii* shoots, while *Halodule uninervis* and *Syringodium isoetifolium* survival and growth did not differ significantly from controls.

In contrast, laboratory experiments by Erskine and Koch (2000) found that short-term (48-hour) exposure of *Thalassia testudinum* to sulfide concentrations up to 10 mM caused lower leaf elongation rates and lower adenylate charge ratios, but the effects were found to be reversible. They concluded that sulfide could not “initiate rapid die-off episodes of *T. testudinum* in Florida Bay.” However, their results were influenced by the short-term exposures used in their experiments and by removal of root and rhizome tissue from *Thalassia* shoots prior to sulfide exposure. In the natural environment, *Thalassia* roots and rhizomes are continuously exposed to sulfide, and natural *Thalassia* beds have high root:shoot ratios.

One additional result of the perfusion experiments is noteworthy: glucose amendments not only increased sediment sulfide concentrations but they also caused sediment pH values to decline from approximately 7.5 to values below 6.5 (Figure 2). Becaused the pK1 of H_2S is 7.0, the 1.0 unit decline in pH causes a three-fold increase in the fraction of total porewater sulfide present as H_2S (Goldhaber and Kaplan 1975). Hydrogen sulfide (H_2S) has been shown to be more toxic than bisulfide (HS-) because H_2S penetrates plant tissues more easily. Not only did sediment sulfide concentrations increase 10-fold in glucose-amended sediments, but the fraction of gaseous H_2S also increased by a factor of three, resulting in a 30-fold increase of the most toxic fraction of the sediment sulfide pool.

Because all experimental studies to date have shown deleterious effects of sulfide on seagrass survival and growth, we conclude that chronic sulfide toxicity is at least a synergistic influence on the die-off of *Thalassia* in Florida Bay. It is also likely that acute toxicity caused by elevated sulfide concentrations is sufficient by itself to cause die-off.

Bird Stake Experiments. Previous studies (Powell et al. 1989, Fourqurean et al. 1995) found that seagrass communities adjacent to bird roosts changed over a period of several years from *Thalassia* dominated beds to monospecific beds of *Halodule*. They concluded that, under nutrient-rich conditions caused by bird guano deposition, *Halodule* outcompeted *Thalassia*. When we resampled bird roosts on Cross Bank in 1991, we found that, in addition to higher sediment nutrient concentrations, sites where birds actively roosted had elevated sediment sulfide concentrations (Figure 3). In fact, mean porewater sulfide concentrations adjacent to active roosting stakes were 2 mM. Adjacent to stakes where birds could no longer roost, mean porewater sulfide concentrations were 1.2 mM. Control sites had porewater concentrations of approximately 0.8 mM. We also found that porewater ammonia and porewater sulfide concentrations in seagrass sediments were positively correlated (Figure 4).

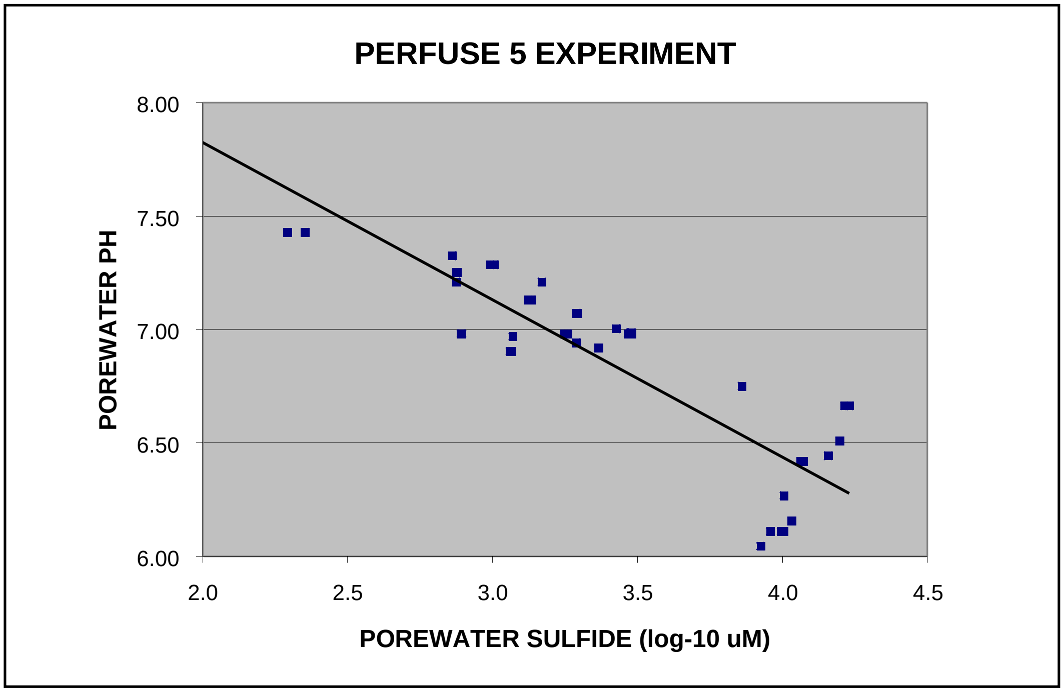

Figure 2. Relationship of sediment porewater pH and sulfide in sediment perfusion experiments.

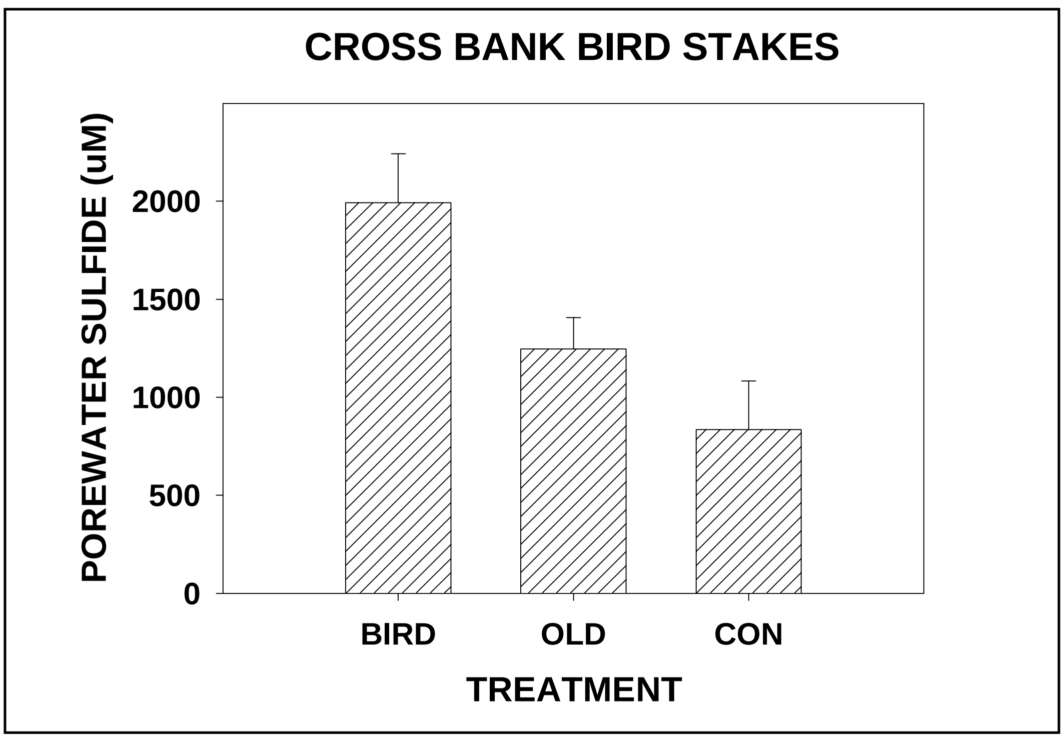

Figure 3. Effect of bird guano on sediment porewater sulfide concentrations. See Fourqurean et al. 1995 for experimental details.

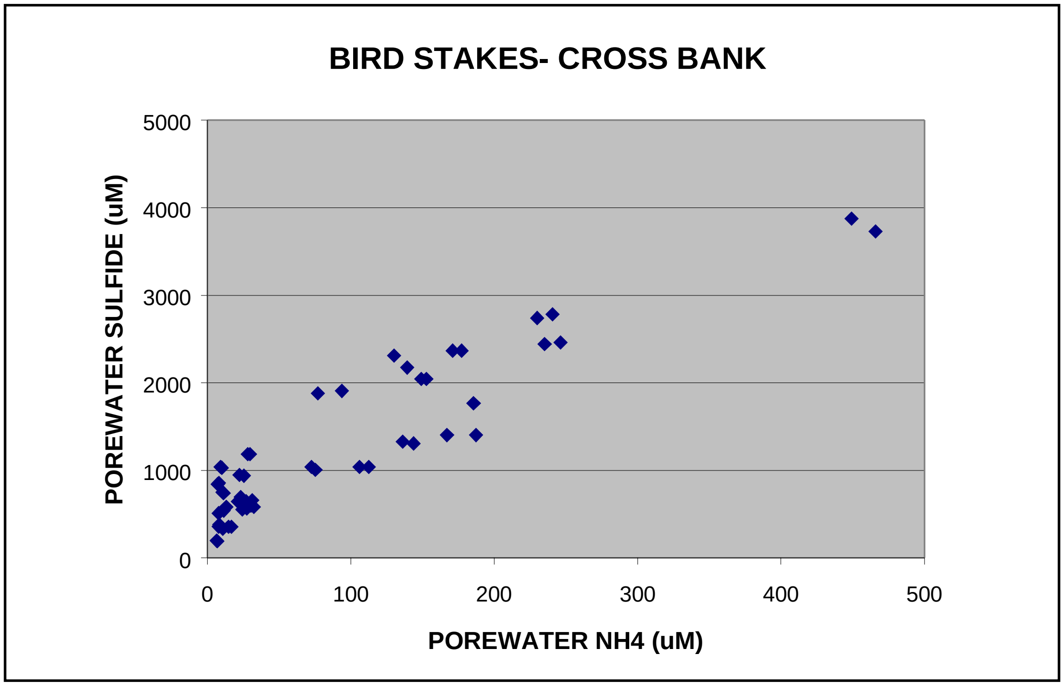

Figure 4. Relationship of porewater ammonia and porewater sulfide concentrations in sediments receiving bird guano.

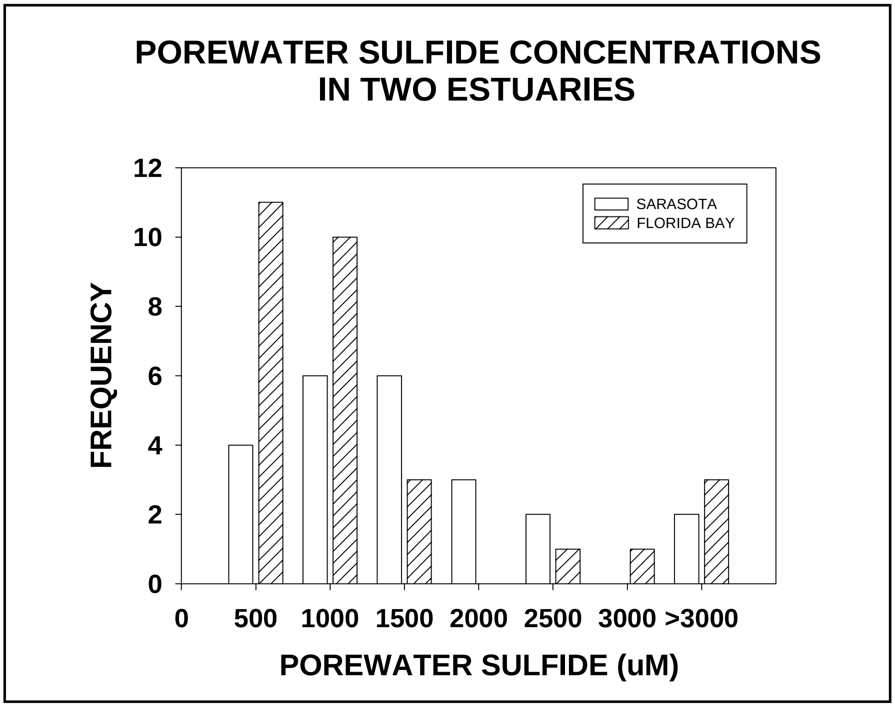

Figure 5. Frequency distribution of porewater sulfide concentrations in biogenic carbonate sediments of Florida Bay (Rabbit Key Basin) and siliceous, terrigenous sediments of Sarasota Bay.

Our porewater sulfide and ammonia data suggest another interpretation of these results. If sediment microbial communities in general, and sulfate-reducing bacteria specifically, are nutrient-limited, then guano addition stimulates heterotrophic microbial activity in the sediments. Porewater sulfide concentrations double, and, because *Halodule* is more tolerant to sulfide than *Thalassia*, *Thalassia* is slowly replaced by *Halodule*.

Surveys of Gulf Coast Estuaries. We originally anticipated that sulfide toxicity was a phenomenon limited to the iron-poor biogenic carbonate sediments of Florida Bay and the Florida Keys. However, in our survey of several estuaries, we found that, while mean sulfide concentrations were less than 2 mM in all estuaries, some estuaries had individual sampling stations with values greater than 2 mM (Figure 5). For example, seven sampling points at Perdido Key near Pensacola and four in Sarasota Bay had sulfide concentrations greater than 2 mM, a value that our work and other studies support as a toxic threshold, at least for *Thalassia testudinum*.

CONCLUSIONS

Our research suggests four conclusions: 1. Acute sulfide toxicity can kill *Thalassia testudinum*; 2. The acute toxicity threshold for *Thalassia* is between 2 mM and 3 mM; 3. *Halodule wrightii* tolerates much higher sulfide concentrations; and 4. Chronic sulfide toxicity causes shifts in seagrass species composition. Like other plant species, *Thalassia* and other seagrasses are probably affected by both direct and indirect sulfide toxicity effects. The direct, cytotoxic effects result from the reaction of sulfide with enzymes required for photosynthesis and respiration. Indirect toxicity effects are caused by hypoxia when photosynthetically-produced oxygen oxidizes sulfide which enters roots and rhizomes.

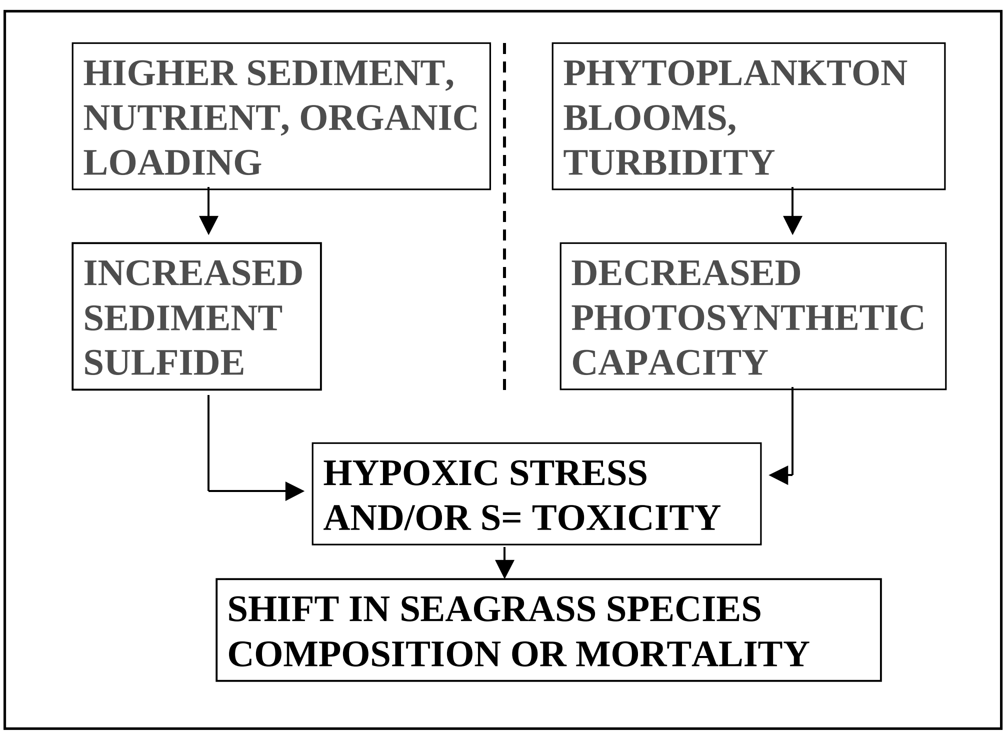

Figure 6. Factors affecting the vulnerability of seagrasses to hypoxia and sulfide toxicity.

Oxygen production and transport within plants is the key to resistance to hypoxia and sulfide toxicity, and seagrass survival depends on a balance between the plant's oxygen supply and sediment porewater sulfide as shown in Figure 6. Any process which causes elevation of sediment sulfide increases hypoxia or sulfide toxicity in seagrasses. Sulfide toxicity can also be increased by factors which decrease seagrass photosynthesis. If the balance between the internal oxygen supply of seagrasses is shifted slightly, seagrass species with higher sulfide tolerance might replace less tolerant species over a period of years.

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COMPACT AIRBORNE SPECTROGRAPHIC IMAGER (CASI) IMAGING OF THE COASTAL ZONE NEAR TAMPA, FLORIDA

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ABSTRACT

An aerial survey crew was in Florida as part of a multi-spectral imaging project, permitting them to offer acquisition of additional imagery with a Compact Airborne Spectrographic Imager (CASI) at reduced rate because of shared mobilization costs. The Florida Department of Environmental Protection (FDEP) used the opportunity to acquire CASI imagery of the Tampa area on April 30, 2000. Target areas selected were as follows: two areas in Tampa Bay (Terra Ceia Bay and Apollo Beach), one area in St. Joseph Sound (where rhodamine dye was injected into adjacent industrial and domestic wastewater discharges) and Anclote Anchorage. An overview of data acquisition, processing, and classification of submerged vegetation in the Apollo Beach area is provided here.

The CASI was flown over the designated areas at several altitudes (10,800 feet for 4 m resolution, 6,000 feet for 2 m resolution and 3,000 feet for 1m resolution). Raw image data were calibrated to radiance units (nW/cm^2/sr/nm) measured at the sensor. The imagery was first corrected for aircraft roll, pitch, yaw and partially for atmospheric interference, then projected into Universal Transverse Mercator (UTM) World Geodetic System (WGS) 84 coordinates and re-sampled to square pixels.

A multispectral classification of submerged vegetation and bottom types for the Apollo Beach area was performed. Data for the targeted areas were delivered as Band InterLeaved (BIL) files and as 3-band tiff files for input into a GIS database.

BACKGROUND

Cost effective remote sensing of resources is an important management tool. Currently, the Tampa Bay Estuary Program (TBEP) utilizes seagrass coverage distribution change as one surrogate for overall health of the Tampa Bay ecosystem. The Southwest Florida Water Management District (SWFWMD) similarly monitors additional geographical areas along the southwest coast of Florida for the same purpose. Recent development in remote sensing technology and availability has led to evaluations of the cost effectiveness of various tools for resource management purposes.

Mumby et al. (2000), include a comparison of cost effectiveness of CASI and aerial photograph interpretation (API). The authors conclude that as the area that needs to be assessed increases, the cost of API is likely to rise faster than the cost of digital airborne scanner surveys (e.g., CASI), making API progressively less cost effective. Virnstein et al. (1997) describe a comparison of digital multispectral imagery versus API for mapping seagrasses in Indian River Lagoon, Florida. Because of contractual problems a cost comparison was not completed, but the authors commented favorably on advantages of multispectral imagery over API, including classification capabilities and the utility provided by a digitally acquired product. The present project was implemented by the Southwest District of FDEP as a further evaluation of the utility of digital multispectral imagery for resource.

Objectives

The main objective of the data acquisition was to acquire aerial multispectral imagery in the Tampa area that could be used to assess and demonstrate the capability of these data. A detailed project plan was developed that included standard operating protocols, instrument bandset and planned flight lines. Borstad Associates was responsible for mission planning, instrument bandset configuration, flight line planning, data collection, image geocorrection, registration and preliminary processing.

METHODS
The Compact Airborne Spectrographic Imager (CASI)

The CASI is a push-broom imager built by Itres Instruments Ltd. of Calgary, Alberta. The Borstad CASI (s/n 101 manufactured in 1990 by Itres Instruments Inc. and modified in 1995 to improve blue sensitivity) was flown in spatial mode, configured to acquire 11 spectral channels for high altitude flights (Table 1, band set MRT1.CCF). The eleven-channel band set was developed specifically for coral reef mapping and used previously for reefs in Mauritius, Reunion (Borstad et al. 1997), Puerto Rico and the Bahamas.

Table 1. Bandsets used for the Tampa survey.

MRT1 Band No.	Wavelength range (nm)	Mean wavelength (nm)
1	421.9–453.6	437.75
2	461.9–490.4	476.15
3	493.4–516.7	505.05
4	526.8–544.9	535.85
5	569.2–587.4	578.3
6	610.0–626.5	618.25
7	640.3–653.2	646.75
8	665.3– 685.3	675.3
9	704.6–715.7	710.15
10	745.8–762.3	754.05
11	774.5–801.8	788.15

With the airplane at 10,800 feet altitude, the cross-track pixel size was 4 m and the image swath was 2 km wide. At an integration time of 75 msec and a ground speed of 104 knots, the along-track pixel size was also 4 m. The imagery was captured with the instrument fore-optics at f8 (or occasionally 5.6) and a narrow angle (35°) lens. Slight variations in speed or altitude will somewhat expand or contract pixels; however, during processing all pixels are re-mapped using a nearest neighbor approach to the desired pixel spacing.

The Borstad CASI also records aircraft roll and pitch from a separate mechanical gyro and latitude and longitude from a GPS receiver to provide data for subsequent geo-correction of the imagery. These flights were staged on April 30, 2000 just before Selective Availability was turned off. However a GPS base station was not used on this mission due to its short duration. The base station would require a minimum of 72 hours operation at an unsurveyed location (within 50 km of the target location) to derive an accurate reference position. As a result, the navigation data were not differentially corrected. However, during second stage rectification of the data, ground control points collected from well navigated reference imagery were used to improve the rectification of the airborne imagery.

Twenty-two flight transects were acquired in just over 2.5 hours of flight time. Two target areas in Tampa Bay (Terra Ceia Bay and Apollo Beach), one target area in St. Joseph Sound (where rhodamine dye was injected into adjacent industrial and domestic wastewater discharges) and Anclote Anchorage were all imaged several times. Some targets were imaged at several altitudes to obtain different spatial resolutions. Only a small portion of one flight line over Apollo Beach is discussed here. The Apollo Beach transect (Figs. 1, 2) was flown 4 times on April 30, 2000.

DATA PROCESSING
Radiometric Calibration

During radiometric calibration, raw image data are read from tape the raw radiance values are converted into upwelling radiance units (nW/cm^2/sr/nm), using a responsivity function obtained during laboratory calibration of the instrument. After removing dark and electrical offset signals from each scan line of the data, the

Figure 1. Flight path for the Apollo transect.

responsivity of each individual array element from the calibration is averaged according to the bands used for the flight data collection, and applied to the image data to produce a radiance for each band. The values calculated represent radiance at the sensor, that includes the effect of the atmosphere below the aircraft as well as the extra path radiance at the edge of the swath. The imagery can be viewed at this stage, but because no corrections have been applied for aircraft motion it can be difficult to interpret.

Geometric and Geographic Correction

During acquisition, the roll and pitch of the aircraft are recorded by the gyro for each scan line while the CASI image is being acquired. As part of the first order geo-correction process the roll, pitch, and GPS data for each image scan line are used to re-map the imagery to remove aircraft motion, using a nearest neighbor approach. Aircraft yaw is not recorded by the gyro but instead was compensated for by re-orienting the camera on each flight line using the photo-camera mount. Data processed to this stage is in 16-bit signed format, in units of radiance (nW/cm^2/sr/nm) at the sensor altitude, each file representing an individual flight line, mapped north up to WGS 84 UTM coordinates and corrected for aircraft motion. Such first order geo-corrections should be accurate to within 10–25 meters over flat terrain.

Second stage rectification, which forces a fit at the ground control point (GCP) and interpolates the regions between the GCPs allows improvement of absolute positioning of image data. This rectification generally reduces position error to within 2

Figure 2. DOP of the Apollo Bay target area.

to 3 pixels relative to the reference imagery.

The geo-referenced data used for the collection of ground control points for the Apollo Beach area was a USGS digital orthophoto retrieved from the USGS website. The original map projection for these maps was NAD27, UTM zone 17. The 1-meter resolution orthophotos for the Apollo Beach area were converted to NAD83 and resampled at 4-meter resolution to match the acquired CASI imagery.

During all transformations the data were resampled using a nearest neighbor technique, ensuring that no spectral distortion was introduced. This is an important consideration for subsequent classification.

Classification

The classification of a resampled and georectified subscene of the Apollo Beach area (Fig. 3) was performed using ENVI's Spectral Angle Mapper (SAM) and PCI's Imageworks spectral tools. The process involved 5 SAM analysis using various angles and modeling algorithms in PCI. The classification thematic channel was filtered in several steps to merge small classification polygons (Fig. 4).

Each class was assigned a color code that was saved into a pseudo-color table. A report on the filtered channel was produced (Table 2) containing areas and percentages for each thematic class. The "null" class referred to the land pixels not included in this classification and zero data. Eleven spectral classes were separated according to spectral similarities and RGB appearance.

Class Descriptions

The classes described in this section were grouped according to spectral similarities, RGB appearance and identified classes from previous classifications of similar targets from other projects done by Borstad Associates' analysts. Some ground truth observations were performed (C. Kovach, J. Culter, Mote Marine Laboratory; personal communications and photos); however, further groundtruthing would be necessary to confirm and identify the species present in each class.

Throughout this section the "blue peak" refers to a radiance maximum near 476 nm, the "green peak" to a radiance maximum near 536 nm, the "orange peak" to a radiance maximum near 578 nm. The "710 peak" refers to a local radiance maximum near 710 nm caused by an interaction between strong infrared reflectance of plants and strong absorption of water. "NTC color" refers to the color of a class in a "near true color" scene composite of the MRT1 bands 8 (675.3 nm), 5 (578.3 nm) and 2 (476.15 nm).

RESULTS

Aquatic Vegetation

The vegetation group accounted for 48.83% of the total classification. The "Submerged" class was the most abundant with the "Sandy bottom with algae growth" group presenting the highest occurrence (23.99%), followed by the "Dark green algal mats" (19%), the "*Gracilaria* & *Lyngbya*" patches (3.03%), the "*Halodule wrightii*" patches (2.29%), the "Brown algae" (0.22%) and finally the "Emergent" class (0.3%).

Submerged Vegetation

All the submerged vegetation exhibited low near infrared (NIR) radiance values spectra but strong/apparent 710 peak. Several classes were obtained as follows:

Sandy bottom with algal growth (Classes 8 & 9). The spectra were characterized by a high "orange peak" (578 nm) and variable but clear 710 peaks (Appendix, Graph 1). This class was identified by groundtruthing as "very shallow bare sandy bottom"; however, the 710 and "orange" peaks

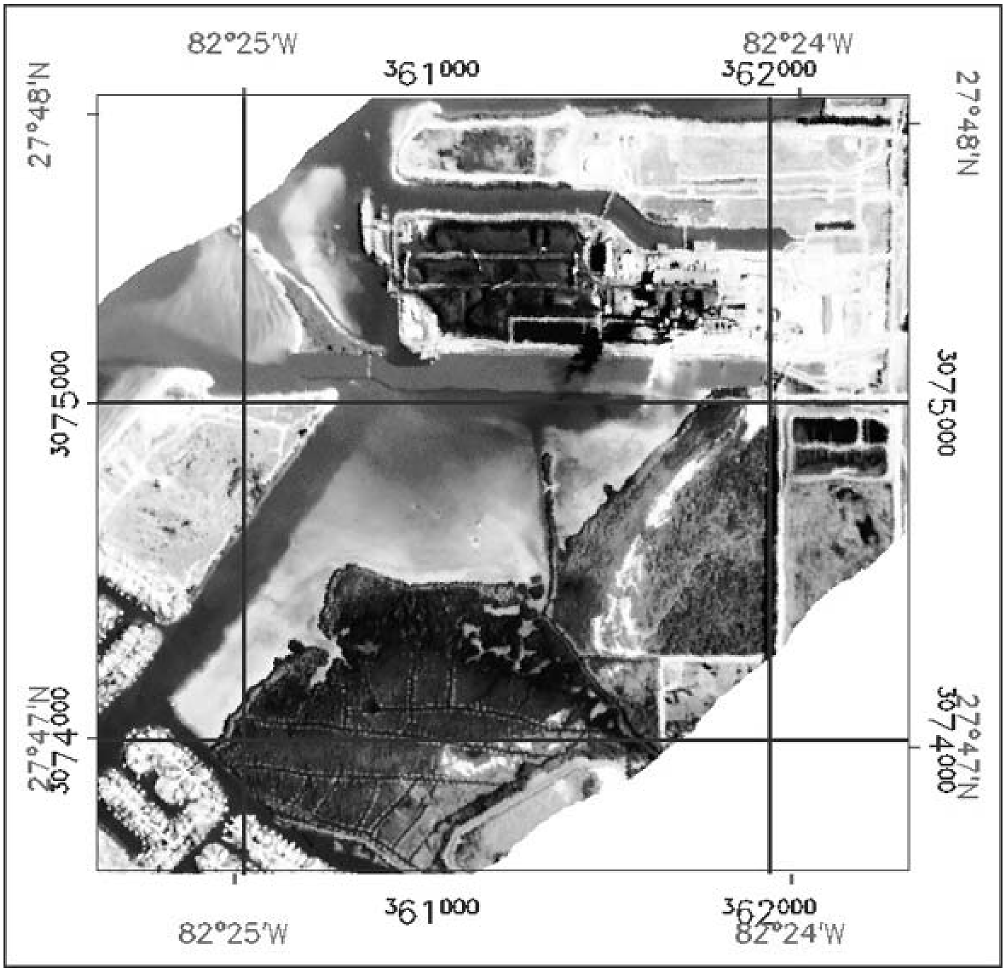

Figure 3. Near true color scene of the Apollo Bay target area.

indicate algal growth, perhaps diatomaceous mats (Paterson et al., 1998) or algal turfs. Two subclasses were separated based on the spectral values:

Very shallow sandy bottom with algae growth (class 8). The spectra had peak radiance values at 578 nm ("orange peak") higher than 3500 DN, with radiance values decreasing towards both ends of the spectral range (Graph 1, green spectrum). The NTC was almost white. This class accounted for 5.09% of the image.

Shallow sandy bottom with algae growth (class 9). The most abundant of this group (18.9 % of the total classification), were spectrally similar to the previous class except that the values of the "orange peak" were lower than 3500 DN, a subtle "blue peak" was present but never higher than "green peak" (Graph 1**,** red spectrum). The NTC was light pink.

Dark green algal mats (classes 6 and 7). This class was recorded from field groundtruthing observations as algal mats (dark green coloration, species unidentified). It was the second most abundant submerged group (19%) and was spectrally similar to the seagrass (Graph 2, blue spectrum) with the presence of a variable "double hump" (strong blue and green peaks) but with lower 710 radiance peaks. Two subclasses were separated based on the spectral values:

Dark green algal mat (class 6). Spectra showed the presence of subtle a 710 peak and a 'double hump' with the radiance peak

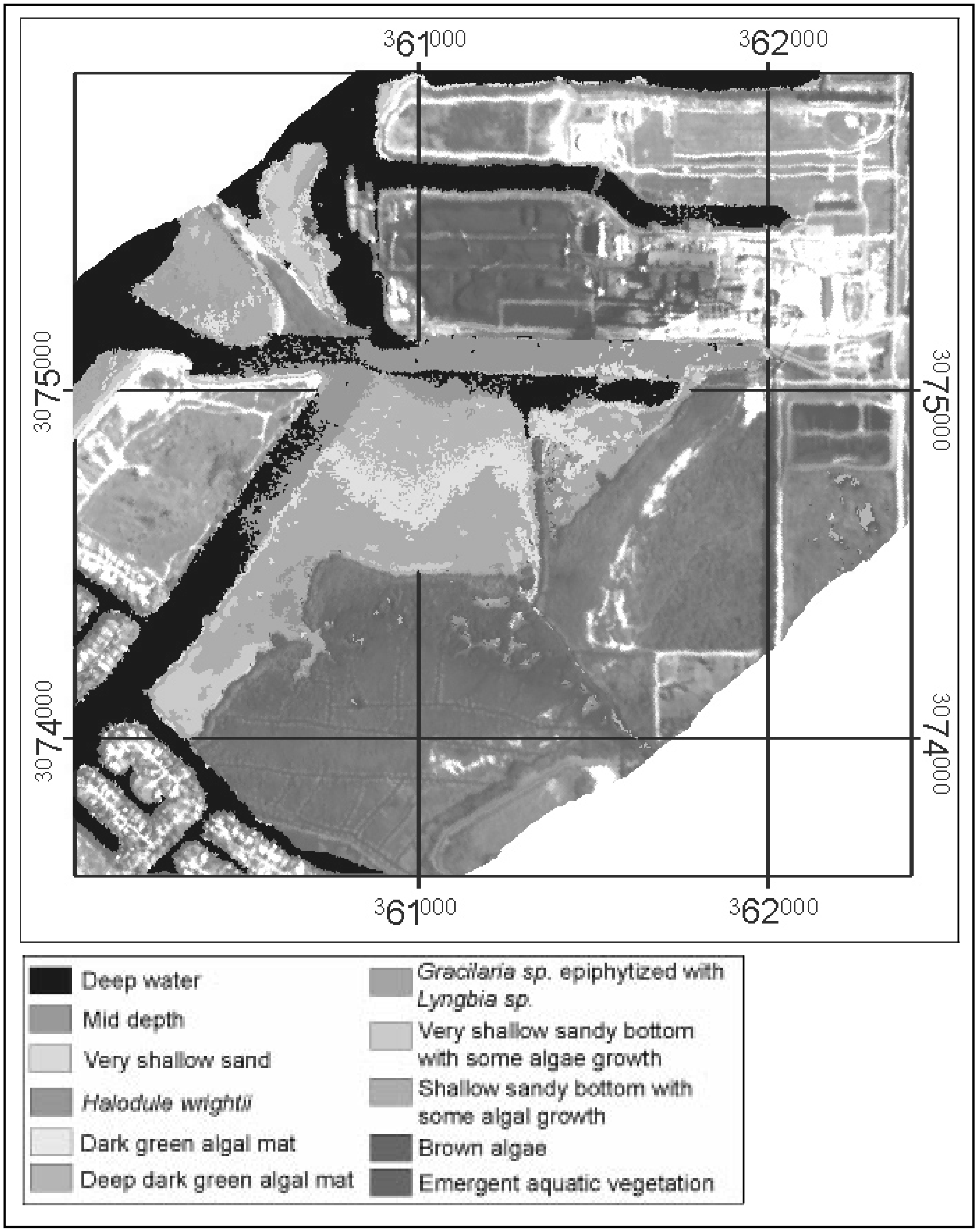

Figure 4. CASI multispectral classification for Apollo Bay.

at 536 nm lower than that at 47 nm (Graph 2, light green spectra). The NTC ranged from light green, yellow-green to dark green.

Deep dark green algal mat (class 11). The most abundant of the two subclasses and the second most abundant class within the submerged group (10.24%). Spectrally similar to the previous class, but with a more defined 'double hump' with the green peak values higher than those of the blue peak. The NTC was light red-green.

***Gracilaria* sp. & *Lyngbya* sp.** (class 7). This class was identified by field observation as the red algae *Gracilaria* sp. epiphytized with the blue green algae *Lyngbya* sp. Spectrally similar to the shallow sandy bottom with algal growth classes (Graph 3)

Table 2. Summary of data acquired for transect 4 on April 30, 2000.

Line No.	Tape 5, AM flight 4 BA	Tape 5, AM flight 4AB	Tape 6, PM flight 4BA	Tape 7, PM flight 4AB
Start Time (GMT)	13:46:52	15:22:49	17:31:46	19:01:01
CASI file	3	14	2	3
Integr. Time (msc)	75	45	45	75
Aircraft Speed (knts)	111	94	116	109
altitude (ft)	10,800	6,200	6,200	10,800
F stop	8	11	11	11
Band set	MRT1	Tampa	Tampa	MRT1
Sun Elevation	38	58	77	65
Intensity	248	306	329	317
Aircraft Hdg (T)	229	47	227	47
Comments	clear	clear, haze on horizon	turbulence, sunglint, clear, overhead	clear

in presenting a clear "orange peak" but differing from those classes in the strong "blue peak" (which values never exceeded those of the "green peak"). The NTC was green-brown-reddish. It was found mainly in the northwest of the mosaic and accounted for 3.032% of the classification.

Seagrass beds: *Halodule wrightii* (class 5). This class was easily separated from the other submerged vegetation classes by the 'double hump' spectra composite of strong blue and green peaks and for the high 710 peak values (Graph 2, blue spectrum). The NTC was dark green. The identification of the *Halodule* patches in the classification was confirmed by field observation and compared favorably with historical records and aerial photos (J.Culter, personal communication) of the seagrass beds in the area.

Brown algae (class 10). Least abundant of the submerged vegetation classes (0.22%) The spectra (Graph 4) exhibited a strong 710 peak and blue peak values higher than any other class. The NTC was dark green-brown.

Emergent Vegetation
This class was not very abundant (0.3 %) and included all spectra with strong 710 peak and high NIR values (Graph 5). It was found along the shoreline and it might represent exposed aquatic vegetation, partially submerged land vegetation or mixed pixels (vegetation and land spectra).

Others
This group includes three no-vegetation aquatic classes, which together accounted for 51.3% of the total classification.

Deep water (class 1). Most abundant in the classification (39.12%) and it was separated by the low spectra with blue peak and radiance values decreasing steadily towards the NIR bands (Graph 6). No 710 peak was present. The NTC was dark blue.

Mid depth (class 2). Spectrally similar to the deep water class but with higher radiance values at 536 nm and 710 nm (Graph 6). The NTC was blue-green and light blue. This class accounted for 11.46 % of the classification.

Shallow sand (class 15). Spectra for this class presented very high radiance values (>4000 DN) at the low wavelength bands (476–578 nm) values decreasing rapidly towards the NIR and a subtle 710 peak was present, indicating possible/sparse growth of vegetation over very shallow/shallow sand/rubble (Graph 7). This class was found along the shoreline and was not very abundant (0.55%).

CONCLUSIONS

The results presented here are the beginning of work to provide cost comparisons and classification precision estimates for multispectral imaging versus traditional aerial photography interpretation for emergent and submerged vegetation classification. The limited areas covered do not allow for useful cost comparisons at this time. Further post-processing of data associated with this project, including emergent vegetation classification and time series of dye movement and relative concentration gradients are in preparation.

ACKNOWLEDGEMENTS

Funds for this project were provided by the Watershed Monitoring and Data Management Section of FDEP. The CASI data was acquired by Wally Horniak and Jose Lim of Borstad Associates Ltd. in an aircraft piloted by Kenny Creed of GPS Aerial Inc., Ormond Beach, Florida. Their dedication and professionalism is greatly appreciated.

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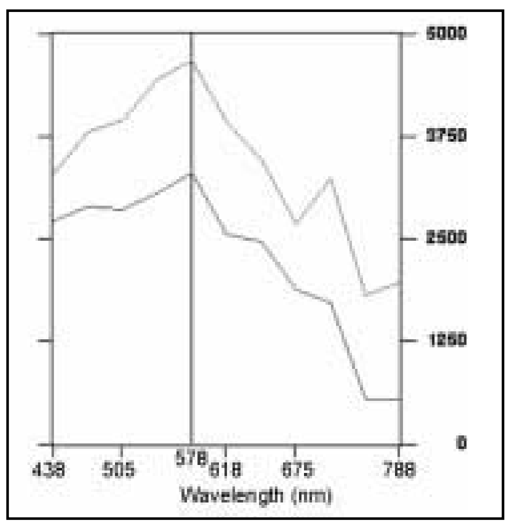
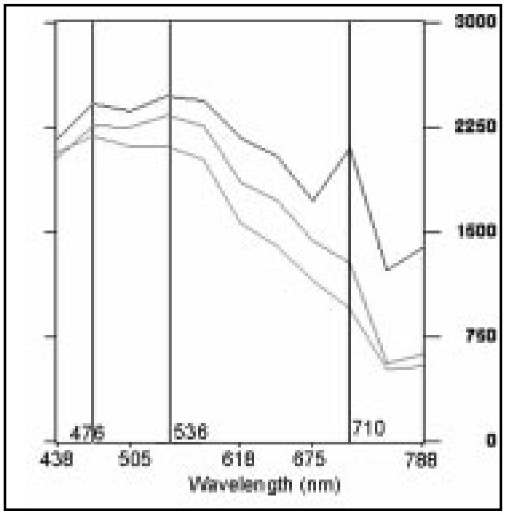
(CWK) Florida Department of Environmental Protection, Southwest District Office, Watershed Management, Technical Services, 3804 Coconut Palm Drive, Tampa, Florida 33619-1352 ; (GAB, MMA) G. A. Borstad Associates Ltd., 114 - 9865 West Saanich Road, Sidney, British Columbia, Canada V8L 5Y8.

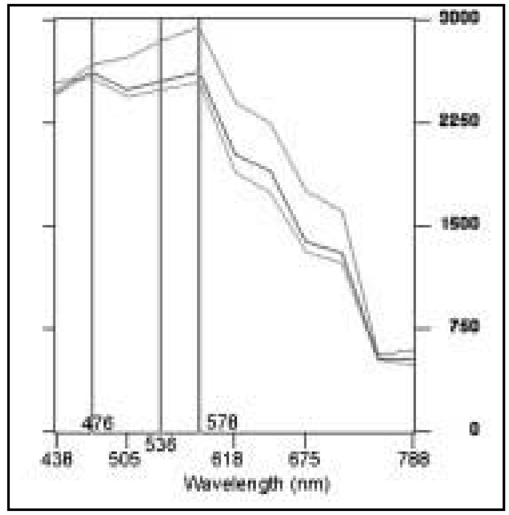
APPENDIX: GRAPHS



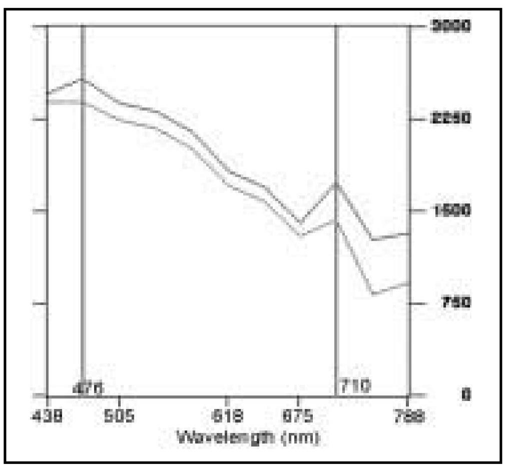
Graph 1. Spectra of sandy bottom with algal growth.



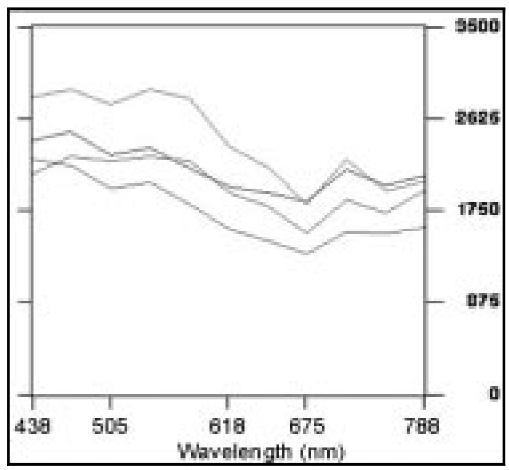
Graph 2. Example of green algae and seagrass spectra. Pink = dark green algal mat; green = deep dark green algal mat; blue = seagrass, Halodule wrightii.



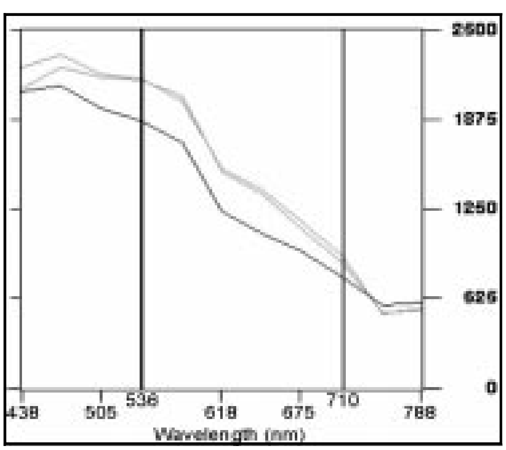
Graph 3. Spectra of *Gracilaria*/*Lyngbia* vs. sandy bottom/algae. Green and blue = *Gracilaria* sp. with epiphyte *Lyngbia* sp.; pink = sandy bottom with algal growth.



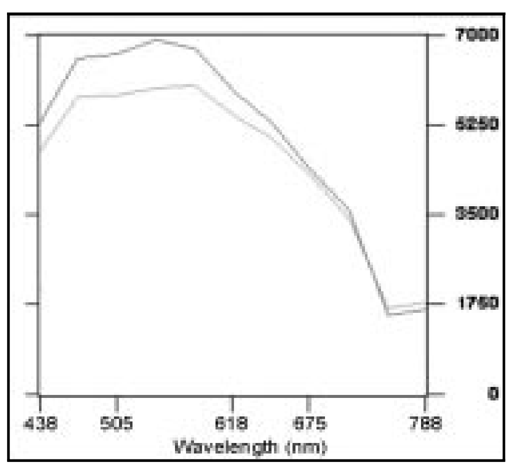
Graph 4. Brown algae spectra.



Graph 5. Example of emergent vegetation spectra.



Graph 6. Example of water spectra. Dark blue = deep water; light blue = mid-depth.



Graph 7. Example of shallow sand spectra.

NITROGEN ISOTOPIC COMPOSITIONS OF SEAGRASS AND ALGAE: IMPLICATIONS FOR TRACING NUTRIENT SOURCES IN TWO FLORIDA ESTUARIES

Kevin S. Dillon, J.P. Chanton, D.R. Corbett, W.C. Burnett

ABSTRACT

Seagrasses and macroalgae samples were collected from Florida Bay and Sarasota Bay and their del ^{15}N compositions were determined in an effort to evaluate nutrient sources within each estuary. In Florida Bay, seagrass and algae samples tended to have similar nitrogen isotopic signatures. Samples from Florida Bay showed ^{15}N enrichment (5-11+ per mil) in the northeast portion of the Bay and along the northern coast. Samples from the western portion of the Bay typically ranged from 1 to 5 per mil. We believe that the enrichment along the northern shore is due to denitrification in the anoxic sediments of the Everglades and subsequent groundwater flow to Florida Bay. The highest del ^{15}N values in Florida Bay were found adjacent to the Florida Keys and are most likely due to wastewater disposal practices, which introduce extraordinary amounts of nutrients to the porous saline aquifer underlying the Keys.

In Sarasota Bay, the del ^{15}N composition of the algae tended to be enriched by up to 4 per mil compared to the seagrasses. The most enrichment was observed in samples collected near the creeks and bayous that flow into Sarasota Bay. Seagrasses collected near Whitaker Bayou and Philippe Creek where enriched relative to the seagrasses from the rest of the Bay. This enrichment is most likely due to wastewater disposal as Whitaker Bayou receives treated sewage effluent while Philippe Creek has a high density of septic tanks in its drainage basin. Both of waterways consistently have elevated DIN concentrations (8-20 M). Macroalgae samples in Sarasota Bay were most enriched near Bowlegs Creek and Hudson Bayou, both of which had DIN concentrations of less than 1 M at the time of sampling. The source of this isotopically heavy nitrogen is currently being investigated.

INTRODUCTION

As coastal regions of Florida become more developed and more populated, there is a subsequent increase in the anthropogenic nutrient loading to many aquatic systems. This increased loading can be attributed to many processes including atmospheric deposition, storm water runoff, sewage disposal practices, farming practices, fertilizer applications, and groundwater seepage (Greening et al. 1997). Pinpointing and quantifying these sources have historically proven difficult as there are often several of these processes working in concert. Stable isotopes have recently been used extensively to determine trophic interactions in aquatic ecosystems. Primary producers typically reflect the stable nitrogen isotopic composition of the nutrients assimilated as the plant or algae matures. If the various isotopic signatures of each nutrient source are distinct then the primary producers growing in the vicinity of each source should reflect the nitrogen isotopic composition of that source. This technique may also allow the zone of influence of a source to be evaluated.

STUDY SITES

In this study, we compare the isotopic signature of seagrasses and benthic macroalgae in two different estuarine systems in Florida: Florida Bay and Sarasota Bay. The seagrass samples included the species *Thalassia testudinum* (turtle grass), *Halodule wrightii* (shoal grass), and *Syringodium filiforme* (manatee grass). At this time, no taxonomic identification has been made on the macroalgae samples. At sites where more than one species of seagrass were collected, the isotopic data were pooled, as were multiple macroalgae samples from individual sampling sites. This was done to allow comparisons to be made between seagrasses and macroalgae samples across the entire Sarasota Bay estuary regardless of spatial variability in species composition.

Florida Bay is a shallow lagoon bordered by the Keys and the Florida mainland. It covers an area of approximately 1800 km^2 and has an average depth of about one meter. Its western margin is open to the Gulf of Mexico. Shallow carbonate mud banks divide the bay into basins, restrict circulation, and attenuate tidal influences from the Gulf (Robblee et al. 1991). Most freshwater enters the bay from the north through Taylor Slough or as sheet flow from the Everglades generated by local precipitation. Salinity in the bay oscillates between brackish and hypersaline. Extensive seagrass beds can be found in the bay. In 1989, Zieman et al. estimated that seagrasses covered more than 80% of the bay.

During the late 1980s seagrass die-offs began occurring throughout Florida Bay (Robblee et al. 1991). The clear and quiescent waters that once characterized the Bay began appearing green and turbid. Algae blooms and seagrass die-offs became commonplace. With seagrasses' death, the muddy bottom sediments of Florida Bay are more easily disturbed. Newly suspended sediments release nutrients to the water column, which in turn fuel microalgae blooms. As turbidity and algal densities increase, light penetration to the bottom decreases and prevents seagrasses from recovering which in turn leads to a less stable bottom. The scenario could lead to a shift from a system dominated by benthic primary production to one dominated by water column photosynthesis. This drastic change has been attributed to elevated salinity and/or increased nutrient loading resulting from the rapid urbanization of south Florida and the Florida Keys (USEPA 1991).

Sarasota Bay is a shallow barrier island type estuary located just south of Tampa Bay in southwest Florida. Lagoonal deposits seaward of the barrier islands that isolate Sarasota Bay from the Gulf suggest that these are transgressive islands that migrated shoreward following the rise in sea level of the Pleistocene (Evans et al. 1985). Freshwater enters Sarasota Bay via several small tidal bayous and creeks which receive most of their input though rainfall and subsequent stormwater runoff and submarine groundwater discharge from the surrounding drainage basin. The major inputs to the Bay from north to south are Bowlegs Creek, Whitaker Bayou, Hudson Bayou and Philippe Creek. Whitaker Bayou also receives effluent from a nearby sewage treatment facility and many of these waterways are lined with homes that utilize septic tanks as a means of sewage disposal (Sarasota County 1998).

Sarasota Bay is hydrologically connected to the Gulf of Mexico by several small passes located between the barrier islands that isolate the Bay from the Gulf. Salinity in the Bay is brackish to saline and is highly dependent on local rainfall. There are extensive seagrass beds along the shallow fringes of northern portions of the Bay and in most of the smaller basins located in the southern portion of the Bay system (Robert's Bay and Little Sarasota Bay). Seagrasses and algae are not found in the deeper portions of the Bay due to high turbidity. Most seagrasses in Sarasota Bay are found at depths of less than 2 meters (Dixon and Kirkpatrick 1995). In the early 1980s Sarasota Bay lost approximately 30% of its seagrass cover due to extensive dredging, pollution and poor water clarity in the Bay. In the late 1980s, municipal wastewater treatment facilities reduced nitrogen loads to Sarasota Bay by as much as 25% through advanced treatment processes and reduced discharges (Sarasota Bay National Estuary Program, 1995). As a result, water clarity began to increase and seagrass beds began to expand to greater depths around Big Sarasota Bay. Between 1988 and 1994, there was a 10% increase in grass cover while between 1994 and 1996 there has been a 22% increase. Similar rates of seagrass increase have also been observed in Little Sarasota Bay although no improvements have been seen in water clarity. In Robert's Bay no change has been documented in water clarity and the increase of seagrass has been about 4% since 1988 (Kurz et al. 1999).

Seagrass and algae samples were collected from across both Florida Bay and Sarasota Bay and analyzed for their nitrogen isotopic composition. The isotope composition of nitrogen is expressed by the del ^{15}N parameter defined as:

$$\text{del } 15N = [\ ((^{15}N/^{14}N)_{sample} - (^{15}N/^{14}N)_{standard}) \ / \ (^{15}N/^{14}N)_{standard}\ ] \times 10^3$$

where the standard is N_2 of the atmosphere. If the sources of dissolved nitrogen in a system have distinct isotopic signatures, this approach may allow the major sources of dissolved inorganic nitrogen (DIN) to be determined as well as the extent of the spatial area that the source is impacting.

METHODS

Seagrass and macroalgae samples were collected by hand and stored on ice until return to the laboratory where they were rinsed with deionized water and dried in a 65°C oven. Seagrass samples with epiphytes were scraped with a razor to remove them. Samples presented here were collected during July and November 1999 and April 2000. After drying, the samples were then ground to a fine powder and rinsed three times with DI water. After each rinse the sample tubes were shaken vigorously, centrifuged and then decanted. After cleaning, the samples were dried, reground and sent to Isotope Services Inc. (Los Alamos, NM) for stable nitrogen isotopic analysis. Samples were packed into tin foil, in duplicate, and placed into a Carlo-Erba NA 1500 elemental analyzer (EA). The EA combusts the sample and a gas chromatograph column yields a pure nitrogen pulse, which is sampled by a VG-Isomass mass spectrometer for ^{15}N analysis. The analysis system relies on a reference gas, which is injected into the helium carrier stream and measured along with every sample (Corbett et al. 1999).

Salinity was measured with a hand-held refractometer. Ammonium concentrations were measured colormetrically as described by Strickland and Parson (1972). Nitrate concentrations were measured with a chemiluminescence NO_x analyzer. The limits of detection for ammonium (NH_4) and nitrate (NO_3) were 1.0 uM.

RESULTS AND DISCUSSION

Florida Bay

The ^{15}N results for Florida Bay were presented by Corbett et al. (1999) and are summarized in Figure 1. For this portion of the data, the seagrass and macroalgal results are combined. A strong spatial pattern occurred in the del ^{15}N signatures of seagrass and macroalgae in Florida Bay, with relatively light values (-1 to 4 per mil) in the western Bay and heavier values (+6 to +13 per mil) in the northeastern portion of the Bay. This gradient in del ^{15}N values is likely a combination of two processes: (1) delivery of nitrate-enriched water from the Everglades (Rudnick et al. 1999); and (2) groundwater seepage from the Florida Keys. The impact of seepage seems to be restricted to the nearshore water adjacent to the Keys (Corbett et al. 1999). The highest concentrations of nitrate in Florida Bay can be found around Taylor Slough and the C-111 canal. This NO_3 is most likely enriched in ^{15}N due to denitrification in the anoxic sediments found in the Everglades (Corbett et al. 1999).

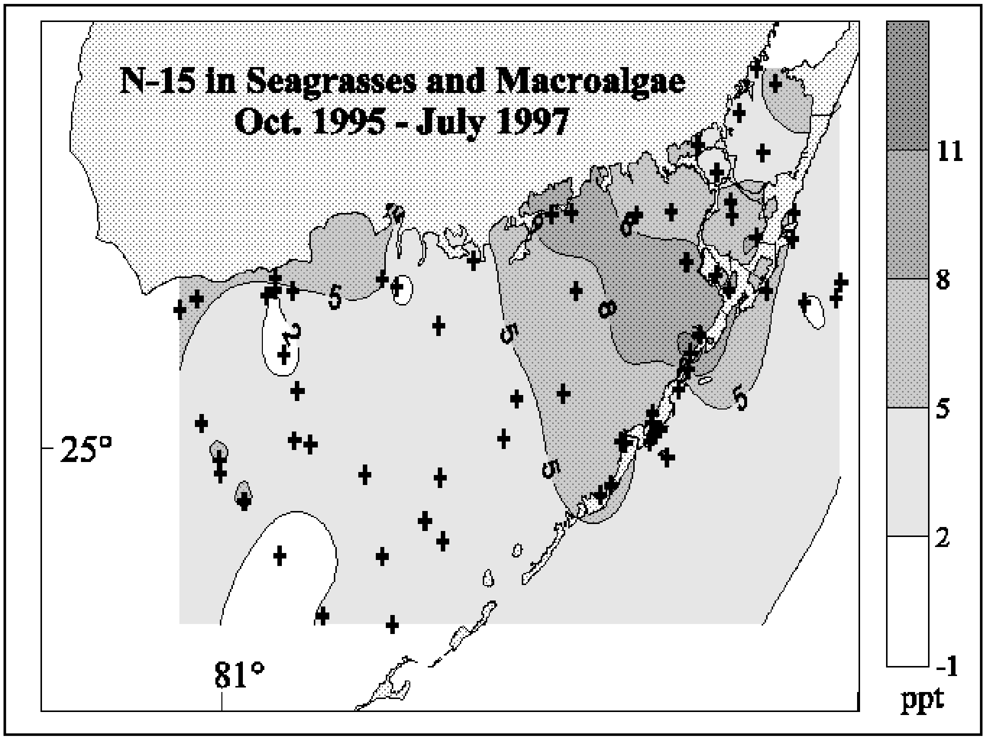


Figure 1. Contours of ^{15}N in seagrasses and microalgae collected throughout Florida Bay. Solid crosses indicate sampling locations. Note the darker contours, indicating higher enrichment, near the upper Keys.

Isotopically heavy seagrasses were also found along both the Atlantic side and the Bay side of the Florida Keys, where nitrate concentrations are very low. This region of Florida Bay is relatively sediment-poor and in many areas the bottom is exposed limestone making denitrification in anoxic sediment highly unlikely in this region. However, sewage disposal practices in the Keys deliver extraordinary quantities of nitrogen to the subsurface via septic tanks and sewage disposal wells. For example, nitrate concentrations as high as 400 µM can be found near septic tanks and sewage injection wells in the Keys (Lapointe et al. 1990; Dillon, unpublished data). Concordant with elevated nitrate, groundwaters in this region are usually anoxic ([O2] = 0.2–0.3 mg/L; Dillon, unpublished data) and very rich in dissolved organic carbon. These conditions are appropriate for denitrification to occur and it is believed that sewage disposal practices may result in a large flux of ^{15}N enriched nitrate to nearby surface waters.

Studies have shown that there is a tidal exchange of water from beneath the Keys to the surrounding surface waters and vise versa (Dillon et al. 1999). When the Atlantic tide is high, the water level in the Atlantic is higher than in the Bay and water is transported into the bedrock from the Atlantic toward Florida Bay. When the Atlantic tide is low, the opposite occurs as water is driven from Florida Bay into the bedrock and toward the Atlantic. It has been shown that inert substances injected into the subsurface near Florida Bay can be pulsed into Florida Bay with each high tide for as long as a year (Dillon et al. 1999). We hypothesize that this mechanism delivers ^{15}N enriched groundwater to the surface waters adjacent to the Keys, resulting in the heavy grass along the Keys themselves. This idea is supported by other natural tracers, which are elevated in groundwaters. Corbett et al. (1999) showed elevated concentrations of both radon and methane collected along the Bay and Atlantic side of the Florida Keys during the same study period, suggesting that groundwater discharge to these areas are high.

Sarasota Bay

For the Sarasota portion of this data, the different types of macrophytes (seagrasses and benthic macroalgae) have been presented separately in order to determine if any uptake or fractionation differences occur between seagrasses and macroalgae as these macrophytes assimilate nitrogen into their biomass. The seagrass portion of the ^{15}N data is shown in Figure 2. Seagrass beds located in the vicinity of Whitaker Bayou and Philippe Creek are isotopically heavier (4.27 ± 0.44 per mil to 5.48 ± 0.6 per mil) than those found throughout the rest of the estuary. In July and November 1999 nitrate concentrations in Whitaker Bayou were 7.87 µM and 16.79 µM, respectively (Table 1). Ammonium concentrations were near the limit of detection for both time periods in Whitaker Bayou. Average nitrate concentrations in Philippe Creek during these periods were 5.2 (July) and 22.6 µM (November). Corresponding ammonium concentrations were 6.9 and 2.3 µM. As can be seen in Table 1, DIN concentrations were relatively low in the rest of the estuary and were often below detection.

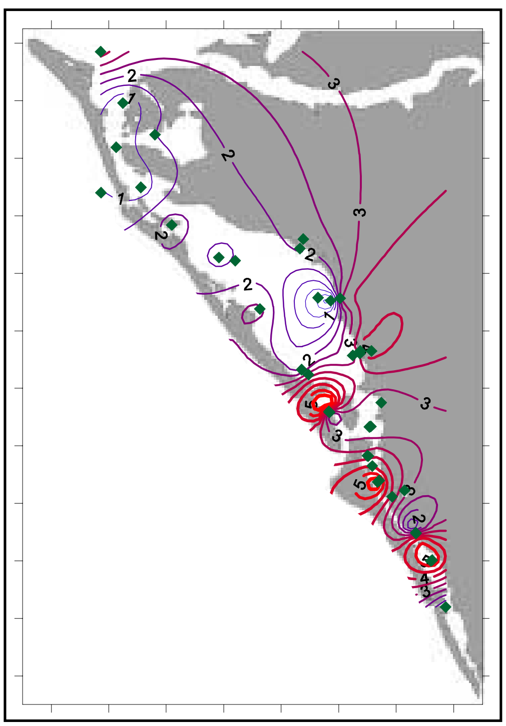


Figure 2. Seagrass del ^{15}N values for Sarasota Bay. Data represents a compilation of data from July and November 1999 and April 2000.

The del ^{15}N signatures of the macroalgae samples in Sarasota Bay showed a slightly different pattern than that of the seagrasses (Figure 3). Generally, macroalgae samples are slightly enriched in ^{15}N compared to the seagrass samples. This pattern is most pronounced from samples collected near Bowleg's Creek and Hudson Bayou. Samples collected from near Whitaker Bayou and Philippe Creek had ^{15}N signatures similar to that of the seagrasses. This suggests that although the dissolved inorganic nitrogen concentrations in Bowleg's Creek and Hudson Bayou are low, the flux may be high enough to deliver significant quantities of DIN to Sarasota Bay. In addition, these inlets may be impacted by a pollutant or a trace nutrient that inhibit seagrass growth and/or encourage the proliferation of certain algal species. These algae may outcompete seagrasses for the small amount of nitrogen present. Alternatively, certain algal species may fractionate nitrogen during uptake.

Figure 4 shows seagrass ^{15}N data plotted against that of the macroalgae for both Florida Bay and Sarasota Bay. Although there is a lot of scatter in the data, it is clear that the macroalgae in Sarasota Bay is more enriched relative to seagrass than the macroalgae found in Florida Bay. Future work in Sarasota Bay will include more taxonomic identification of the macroalgae in order to determine if fractionation is greater in certain algae. Additional work is currently being conducted to examine the del ^{15}N composition of different DIN sources to the Bay (wet atmospheric deposition, storm water runoff, sewage, etc.) in order to determine the origins and

Table 1. Nutrient data for Sarasota Bay surface water stations.

July 8-9, 1999		**NH_4 conc**	**SD**	**NO_2 conc**	**SD**	**NO_3 conc**	**SD**
sample		**uM**		**uM**		**uM**	
S1	Whitaker Bayou	1.16		0.29	0.07	7.87	0.55
S5	midbay	BD		BD	0.00	BD	
S5 dup		BD		BD	0.01	BD	
S6	Bayshore Garden	BD		BD	0.01	BD	
S9	Sister Key mrkr 32	1.77	1.78	BD	0.01	BD	
S15	Gulf	BD		BD	0.03	0.44	0.40
S16	north Sarasota Bay	1.45	1.13	BD	0.01	0.36	
S19	Roberts Bay	BD		BD	0.01	BD	
S21	Phillippi Creek	6.91	0.01	0.73	0.03	4.89	0.64

November 5-6, 1999		**NH4 conc**		**NO2 conc**		**NO3 conc**		**PO4 conc**	
sample		**(uM)**	**SD**	**(uM)**	**SD**	**(uM)**	**SD**	**(uM)**	**SD**
Gulf	Gulf	BD		BD	0.02			BD	
S1	Whitaker Bayou	BD	0.70	BD		16.79	0.26	1.23	0.03
S5	Midbay	BD		BD				BD	
S19	Roberts Bay	BD		BD				BD	
S21	Phillipi Cr	2.26	0.04	0.59	0.00	22.57	0.21	4.34	0.03
S29	Marker 8A	BD		BD				BD	
	Hudson Bayou	BD		BD		0.48		0.56	0.02

		NH4 conc		**NO2 conc**		**NO3 conc**		**PO4 conc**	
sample		**(uM)**	**SD**	**(uM)**	**SD**	**(uM)**	**SD**	**(uM)**	**SD**
S1	Whitaker Bayou	BD		0.07	0.07	BD		1.46	0.06
S1 dup		BD		BD		BD		1.42	0.01
S5	Midbay	BD		BD		BD		BD	
S7		BD		BD				BD	
S9	Sister Key mrkr 32	BD		BD				0.07	0.01
S15	Gulf	BD		BD				BD	
S16	north Sarasota Bay	BD		BD		BD		BD	
S16 dup		BD		BD		BD		BD	
S19	Roberts Bay	BD		BD				0.55	0.04
S21	Phillipi Cr	7.31	0.29	0.15	0.00	0.61	0.01	9.31	0.23

detection limits
NH_4 1.0 uM, NO_2 0.2uM, NO_3 1.0 uM, PO_4 0.5 uM

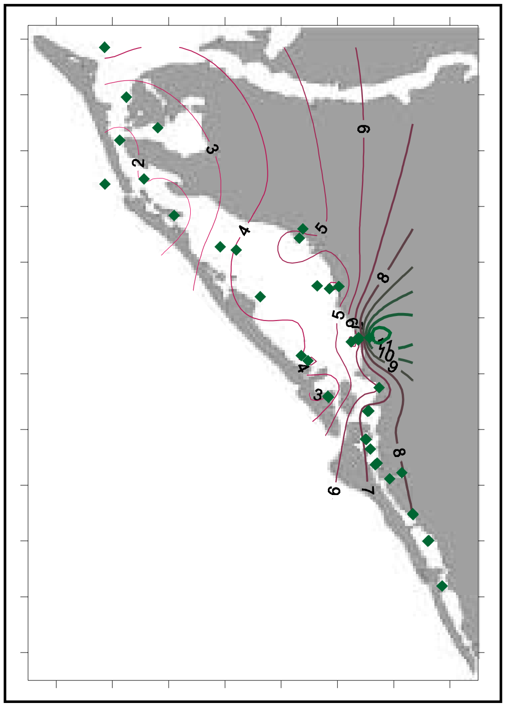


Figure 3. Macroalgae del ^{15}N values for Sarasota Bay. Data represents a compilation of data from July and November 1999 and April 2000.

fates of the isotopically enriched nitrogen that is delivered to Sarasota Bay.

CONCLUSIONS

It appears the stable nitrogen isotopic signature of seagrass can be used to determine source areas of nitrogen to different aquatic ecosystems. When used in conjunction with other chemical parameters such as nutrients, radon and methane even more can be learned about the origins of DIN available to primary producers. In Florida Bay, these parameters suggested that there are two major sources nitrogen to the northeastern portion of Florida Bay. Clearly, more work needs to be conducted to identify all of the sources of nitrogen to Sarasota Bay. The results of this study do suggest that the major sources of nitrogen to Sarasota Bay may be attributed to atmospheric deposition and consequential storm water runoff as the heaviest nitrogen signals were found near the creeks and bayous that drain into Sarasota Bay. Wastewater disposal is likely an important source near Whitaker Bayou. Philippe Creek may also be impacted by human waste disposal methods, as there is a high density of septic tanks along this waterway, which may be responsible for the elevated nitrate concentrations found during this study. This idea is supported by high fecal coliform counts that have been measured in this creek (Sarasota County 1998). Ongoing work in Sarasota Bay involves measuring the del ^{15}N signature of the many possible sources of DIN to the Bay and will shed more light on the nitrogen dynamics of the Sarasota Bay estuary.

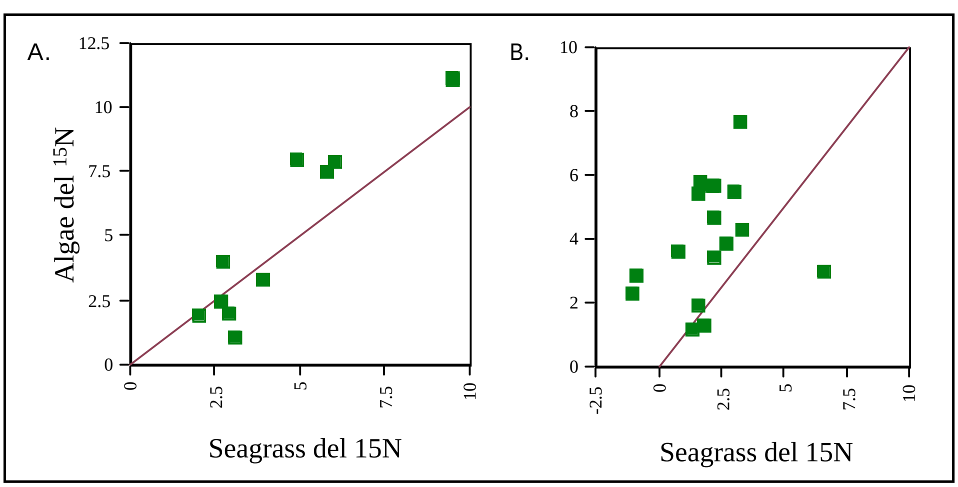


Figure 4. Seagrass del ^{15}N values plotted against macroalgae del ^{15}N values for A) Florida Bay and B.) Sarasota Bay. The solid line indicates a 1:1 ratio in both cases.

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