

Chapter 4. Seagrass

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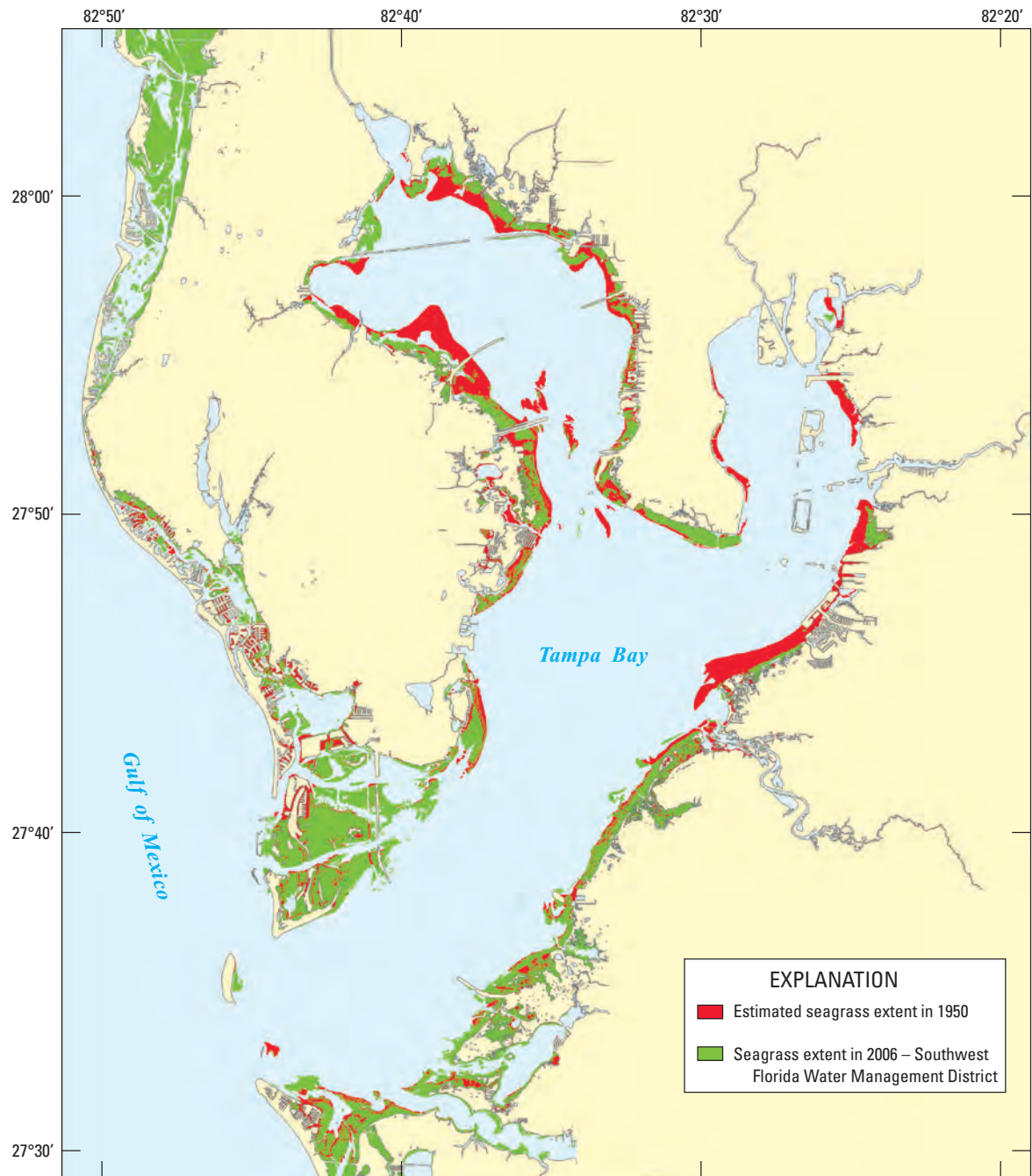
SEAGRASSES ARE A UNIQUE group of flowering plants that have become adapted to life in estuarine and marine waters (Larkum and others, 2006; Orth and others, 2006). They have been identified as a key environmental resource in Tampa Bay because of their critical importance in providing habitat for large numbers of environmentally and economically important fish and shellfish species. In addition they are sensitive to water-quality degradation, play a valuable role in nutrient cycling and improving the stability of bottom sediments, and serve as a food source for manatees, sea turtles, and other wildlife (TBEP, 2006) (fig. 4–1). Primary productivity in seagrass habitats, by a combination of seagrasses, macroalgae, and epiphytes, can reach levels that are similar to or greater than many cultivated terrestrial systems (Duarte and Chiscano, 1999). In terms of monetary value, seagrass meadows and submerged algal beds have been estimated to provide ecosystem services worth about \$7,700 per acre/yr (in 1994 \$U.S.) (Costanza and others, 1997).

Figure 4–1. Seagrass habitat at Pinellas Point in Middle Tampa Bay. Photo by Southwest Florida Water Management District.



Figure 4-2. Estimated changes in seagrass cover in Tampa Bay between 1950 and 2006. Green areas show seagrass extent as mapped in 2006; red areas represent locations where seagrass occurred in 1950 but was not mapped in 2006. Data from Haddad, 1989; Southwest Florida Water Management District.

Seagrass cover has declined in many estuaries on a worldwide basis in recent decades, due largely to anthropogenically induced declines in water quality and other human impacts (Orth and others, 2006). Substantial reductions in seagrass cover have also occurred in Tampa Bay, particularly during the period of rapid coastal development and bay eutrophication (discussed in more detail in Chapter 5) that took place between the 1950s and early 1980s (Johansson, 1991, 2002a; Crane and Xian, 2006). Two central objectives of the Tampa Bay resource-management community are to protect the 28,000 acres of seagrass currently estimated to be present in the bay, and to restore 10,000 additional acres in order to return bay-wide seagrass cover to a level similar to that of the early 1950s (fig. 4-2) (TBEP, 2006).



Seagrass Species Found in Tampa Bay

Of the seven seagrass species that occur in Florida, five are found in Tampa Bay (fig. 4–3). *Thalassia testudinum* (turtle grass) (fig. 4–3A) is the most physically robust species, with long strap-shaped leaves and large subsurface rhizomes. In higher-salinity parts of the bay, large seagrass meadows are usually dominated by this species, in combination with *Syringodium filiforme*. *Syringodium* (manatee grass) (fig. 4–3B) can be distinguished by its cylindrical leaves, which are frequently broken off from the parent plant and dispersed widely by winds and currents. *Halodule wrightii* (shoal grass) (fig. 4–3C), which is also a common species in large parts of Tampa Bay, has flat, narrow leaves and a shallow root system. It is thought to be an early successional species in the development of seagrass beds in the Gulf of Mexico and Caribbean Sea, and is a dominant species in many estuarine environments. *Halodule* is able to survive more frequent and prolonged exposure during periods of low tide (fig. 4–3D), and is often the predominant species at the shallow water fringe of large, mixed-species seagrass meadows. In some areas, *Halodule* also dominates the deep-water edge of many meadows. It tends to have smaller carbohydrate reserves than *Thalassia*, however, and less ability to tolerate periods of reduced light availability (Czerny and Dunton, 1995; Neely, 2000). A fourth species, *Halophila engelmannii* (star grass) (fig. 4–3E), is not common in Tampa Bay, but in other areas it is commonly present in meadows dominated by *Thalassia* and *Syringodium*. *Halophila* also occurs in deeper areas where those genera are absent. The fifth species, *Ruppia maritima* (widgeon grass) (fig. 4–4), tolerates a wide range of salinities and often forms dense beds in the lower-salinity parts of upper Tampa Bay. In recognition of its broad salinity tolerance, some researchers have suggested that *Ruppia* can be thought of as a freshwater plant that is also capable of living in saline environments, rather than as a seagrass *sensu stricto* (Dawes and others, 2004). Generic depth zonations of seagrasses in nearshore waters of the Florida Gulf Coast are shown in figure 4–5.

Seagrass Habitat Requirements

Extensive field research at the global level indicates that water clarity and the availability of adequate sunlight are key habitat requirements for all seagrass species (Duarte, 1991; Fourqurean and Zieman, 1991; Johansson, 1991; Batiuk and others, 1992; Dennison and others, 1993; Onuf, 1994; Dixon and Leverone, 1997; Short and Coles, 2001) (fig. 4–6). Photosynthetically active radiation (PAR), which generally corresponds to the visible part of the electromagnetic spectrum (wavelengths ranging between 400 and 700 nanometers), is the critical part of the spectrum for seagrasses and other plants. Within the water column PAR is usually measured using quantum sensors (fig. 4–7) or estimated using a Secchi disk (fig. 4–8). Unlike phytoplankton and macroalgae, which typically require only 1 to 5 percent of incident PAR to maintain a positive energy balance, seagrasses require much higher levels — on the order of 11 to 20 percent or more — for long-term survival and growth (Duarte, 1991; Dennison and others, 1993; Gallegos and Kenworthy, 1996; Kenworthy and Fonseca,

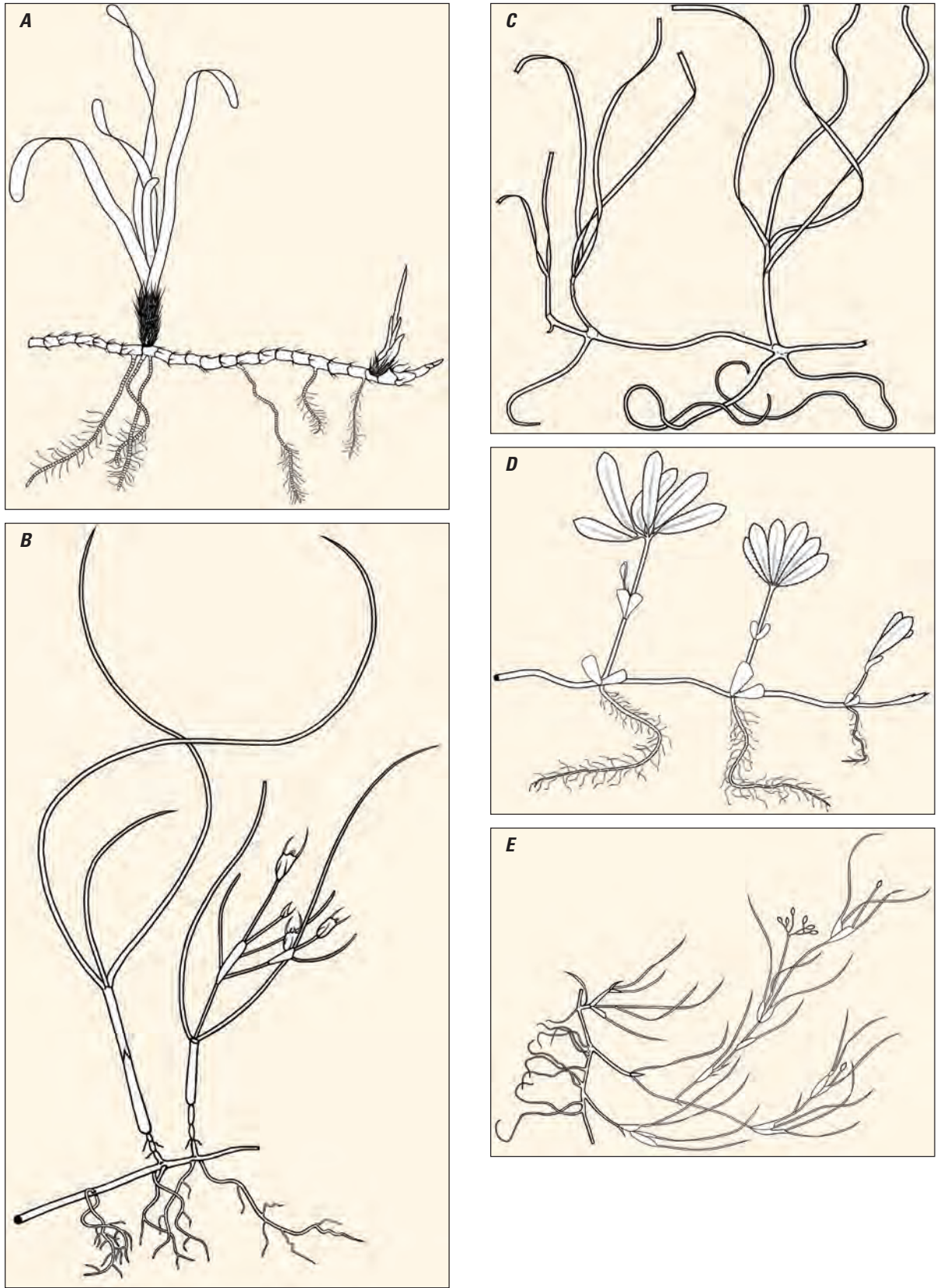


Figure 4-3. Common seagrass species found in Tampa Bay. A, *Thalassia testudinum* (turtle grass); B, *Syringodium filiforme* (manatee grass); C, *Halodule wrightii* (shoal grass); D, *Halophila engelmannii* (star grass); and E, *Ruppia maritima* (widgeon grass). From Dawes and others (2004).

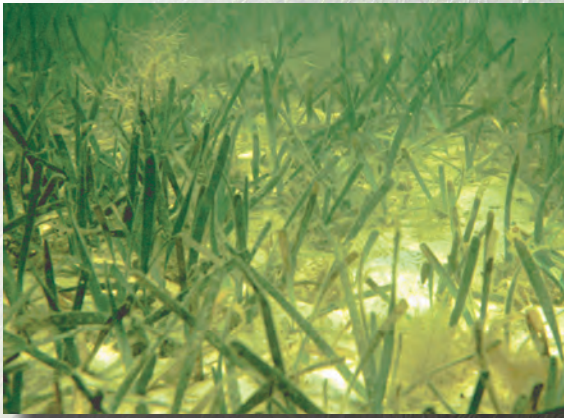


Figure 4-3A. *Thalassia testudinum*. Photo by Lindsay Cross, Tampa Bay Estuary Program.



Figure 4-3B. *Syringodium filiforme*. Photo by Fish and Wildlife Research Institute.



Figure 4-3C. *Halodule wrightii* and tubular sponge. Photo by Roger Johansson, City of Tampa Bay Study Group.



Figure 4-3D. *Halodule wrightii* exposed at low tidal stage. St. Petersburg, Florida, is in background. Photo by Roger Johansson, City of Tampa Bay Study Group.



Figure 4-3E. *Halophila engelmannii* in petri dish. Photo by Roger Johansson, City of Tampa Bay Study Group.



Figure 4-4. *Ruppia maritima*. Photo by Roger Johansson, City of Tampa Bay Study Group.

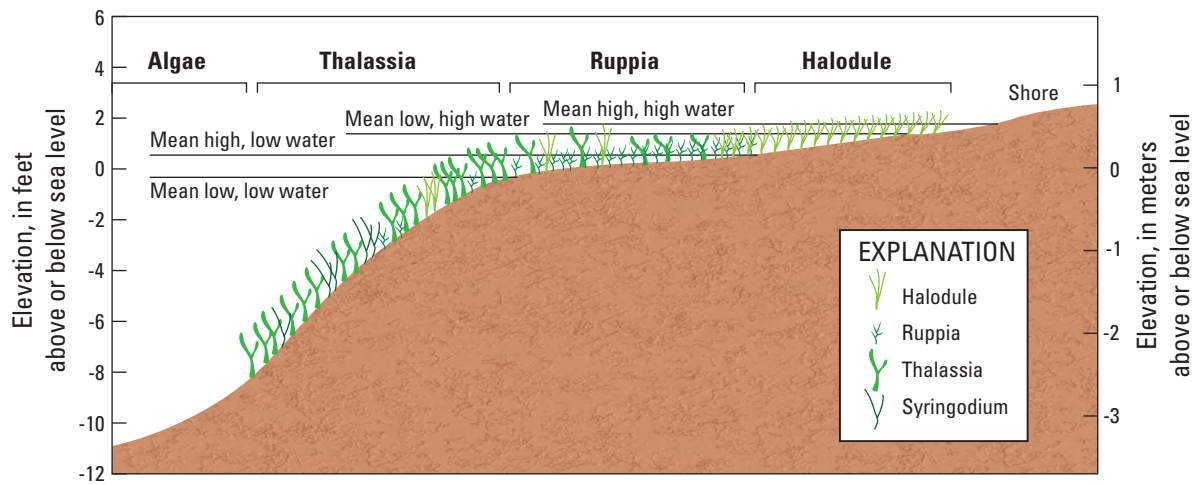


Figure 4-5. Typical depth zonation patterns of Gulf Coast seagrass species. Modified from McNulty and others (1972) and Dawes and others (2004).

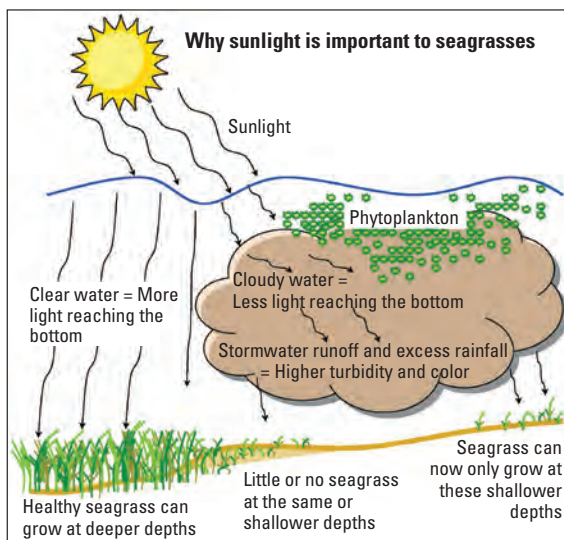


Figure 4-6. Why sunlight is important to seagrass. From Cross (2007).



Figure 4-7. Measuring photosynthetically active radiation using quantum sensors. Photo by U.S. Geological Survey.



Figure 4-8. Measuring water clarity using a Secchi disk. Photo by U.S. Geological Survey.

1996; Dawes and others, 2004). Recent results suggest that even higher levels (perhaps up to 45 percent of incident PAR) may be required in highly turbid areas (Duarte and others, 2007). As a result, seagrasses are usually found in relatively clear and shallow waters. On a global scale, reductions in light availability caused by reduced water clarity have been implicated in many cases of large-scale seagrass loss (Orth and others, 2006).

In Tampa Bay anthropogenically induced reductions in water clarity that occurred between the 1950s and early 1980s appear to have played a major role in the declines in seagrass acreage that occurred during that period (Johansson 1991, 2002a; Johansson and Lewis, 1992; TBEP, 2006). Since the early 1980s, as water quality improved, reduced anthropogenic nutrient loads, reductions in the abundance of phytoplankton and macroalgae, improvements in water clarity, increased penetration of PAR through the water column, and increases in seagrass cover on a bay-wide scale have been widely documented (Johansson and Greening, 2000; TBEP, 2006). At a more detailed level, Neeley (2000) conducted field experiments to examine the effects of shading (using neutral density screening to reduce light penetration) in individual *Halodule* beds near the mouth of the bay. The irradiance reductions, which ranged from 43 to 86 percent and occurred from September 1994 to early March 1995, resulted in progressive reductions in *Halodule* shoot density and photosynthetic biomass (Neely, 2000). Respiration rates, leaf chlorophyll content, and photosynthesis-irradiance (P-I) relations showed no trends related to the experimental light reductions, although P-I curves (based on laboratory measurements of O₂ production by excised leaf segments) showed evidence that photosynthesis rates became light-saturated at irradiance levels >400 microEinsteins ($\mu\text{E m}^{-2} \text{s}^{-1}$) (Neely, 2000).

Dixon (2002) collected a continuous record of PAR levels in stable *Thalassia* beds in the southern part of the bay during a 12-month period, extending from November 1993 through early December 1994, to provide additional information on the light requirements of *Thalassia*. Previous work in the area had provided evidence that the deepest beds in that part of the bay were light-limited (Hall and others, 1991). The deep edges of the beds were found to correspond to the depth at which about 20.5 percent of subsurface irradiance (sunlight that penetrates the water surface and enters the top of the water column) reached the bay bottom on an annual average basis (Dixon, 2002). The timing of adequate light availability also seemed to be a factor in maintaining bed condition, with light-stressed beds receiving inadequate irradiance levels during the early growing-season months of March, April, and May. The annual average light level (20.5 percent of immediately subsurface PAR), which was estimated by Dixon (2002) to occur at the deep edges of these stable *Thalassia* beds, has been used to establish initial water-clarity targets for seagrass restoration in Tampa Bay (Greening, 2002; TBEP, 2006), as part of a broader, resource-based water-quality management process that is discussed in more detail in Chapter 5.

In addition to water clarity and the overall quantity of sunlight that is available in the water column to support plant growth, other habitat quality factors that affect the size and condition of seagrass beds include:

- Light “quality” (that is, spectral distribution) (Tomasko, 1992; Rose and Durako, 1994);
- Current velocity and wave energy (Fonseca and Bell, 1998; Koch, 2001; Koch and others, 2006a,b);

- Sediment grain size and stability (Orth and others, 2006);
- Sediment chemistry (Batiuk and others, 2000; Carlson and others, 2002);
- Variations in freshwater influx and salinity (Philips, 1960a,b; Dunton, 1996; Tomasko and Hall, 1999; Doering and Chamberlain, 2000);
- Boating impacts, such as propeller scarring and vessel groundings (Kruer, 1994; Sargent and others, 1995);
- Destruction of beds by dredging or filling during coastal construction activities (Lewis, 1977; Janicki and others, 1995);
- Bioturbation (Fonseca and others, 1998; Short and others, 2006);
- Herbivory (Camp and others, 1973);
- Epiphyte loads (Tomasko and Lapointe, 1991; Dixon, 2000);
- Disease (Rasmussen, 1977; Blakesly and others, 2002); and
- Sea-level rise (Glick and Clough. 2006).

Types of Seagrass Beds in Tampa Bay

Field studies summarized by Lewis and others (1985) and Lewis and Estevez (1988) indicate that seagrass beds in Tampa Bay tend to be relatively monogeneric. During the period when those summaries were prepared, about 40 percent of the beds were dominated by *Thalassia*, 35 percent by *Halodule*, 15 percent by *Syringodium*, and 10 percent by *Ruppia*. A more recent study, based on monitoring of fixed transects, indicates that *Halodule* is currently the most abundant genus in the bay, and is found at nearly twice the frequency of *Thalassia* (Avery and Johansson, 2008).

Lewis and others (1985) identified five general types of seagrass meadows in the bay based on their location, form, and species composition (fig. 4–9):

- Mid-bay shoal perennial (MBS[P]). Year-round (perennial) beds on natural mid-bay shoals, usually containing *Halodule*, *Thalassia*, and *Syringodium*.
- Healthy fringe perennial (HF[P]). Year-round beds occurring along the bay shoreline, from mean low water to depths of about 2 m, containing some or all of the five Tampa Bay seagrass species.
- Stressed fringe perennial (SF[P]). Similar to HF[P] meadows, but with reduced seagrass density and cover.
- Ephemeral (E). Nonperennial beds whose locations fluctuate from year to year, composed almost entirely of *Ruppia* with occasional *Halodule*.
- Colonizing perennial (C[P]). Meadows formed in shallow-water areas on or adjacent to manmade fill sites, such as bridge causeways, dominated by *Halodule* and *Syringodium*, perhaps because of their dispersal and colonization abilities.

Since 1997 a consortium of organizations has monitored seagrass distribution, density, and condition along a number of fixed transects that have been established in each of the major segments of the bay and within each of 30 seagrass management areas that were designated by bay resource managers in 2007 (Avery and Johansson, 2001, 2008; Environmental Protection

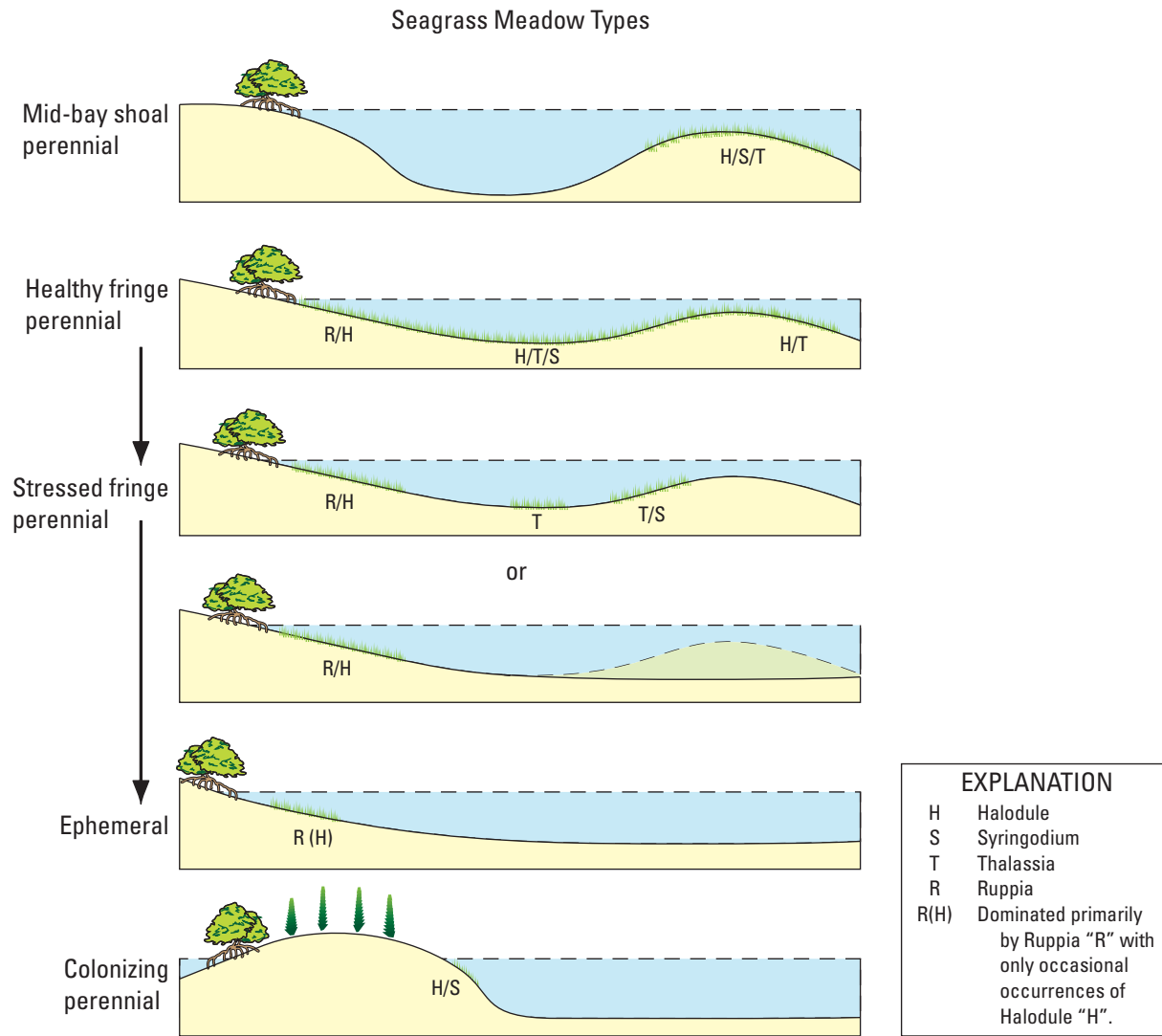


Figure 4–9. Classification system for the seagrass beds of Tampa Bay. Acronyms: MBS(P) = mid-bay shoal perennial, HF(P) = healthy fringe perennial, SF(P) = stressed fringe perennial, (E) = ephemeral, C(P) – colonizing perennial. Modified from Lewis and others (1985).

Commission of Hillsborough County (EPCHC), 2007) (fig. 4–10). Each transect starts at the shoreline and extends to a water depth that exceeds the seagrass restoration target depth for the bay segment where the transect is placed (fig. 4–11). Along each transect the occurrence and abundance of seagrass and attached algal species are estimated using the Braun Blanquet coverage-class rating system, and seagrass short-shoot density and canopy height are determined. The objective of the project, which currently includes 62 transects that are monitored on an annual basis, is to provide information on the areal extent of seagrasses (and attached algae) in each bay segment, the zonation of seagrass and attached algal species with depth, changes in the spatial distribution of seagrass and attached algal species over time, and the overall condition of seagrass beds. During the 1997–2000 monitoring period, the following patterns were noted (Avery and Johansson, 2001):

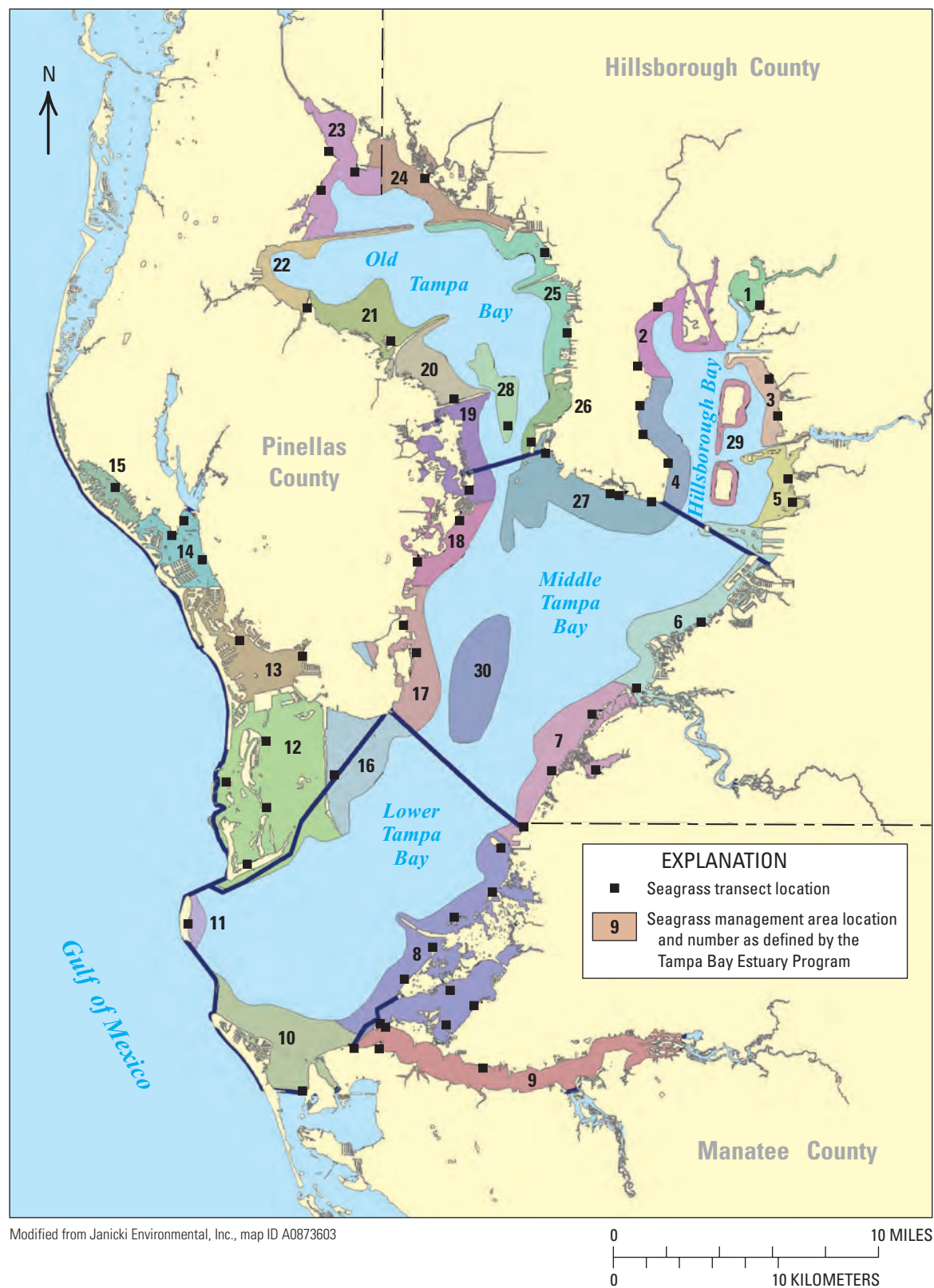


Figure 4-10. Locations of 62 fixed seagrass transects in Tampa Bay. Data from Environmental Protection Commission of Hillsborough County (2007) and Avery and Johansson (2008); map by Janicki Environmental, Inc.

- *Thalassia* and *Syringodium* were more common on transects located on the eastern side of the bay, and *Halodule* generally dominated the genus composition on the western side.
- *Halodule* was the dominant seagrass found on transects located in Hillsborough Bay.
- Generally, *Halodule* was the dominant seagrass in the northern section of Middle Tampa Bay. *Thalassia* and *Syringodium* were more common on transects located in the middle and southern sections of this bay segment.

During the 10-year period from 1997 to 2006, the following patterns were noted in the frequency of seagrass occurrence along monitored transects (Avery and Johansson, 2008):

- Old Tampa Bay showed the largest increase, primarily due to increased occurrence of *Halodule* and *Syringodium* cover.
- Seagrass cover remained generally sparse in Hillsborough Bay (>80 percent of the area along monitored transects was unvegetated), whereas the macroalga *Caulerpa prolifera* showed episodic biomass spikes in some areas.
- Seagrass occurrence along transects in Middle Tampa Bay remained relatively stable, with seagrass present in about 40 percent of the monitored transect area, although local variations in the relative abundance of *Halodule* and *Syringodium* produced fluctuations in composition.
- In Lower Tampa Bay, seagrass cover along the transects was reduced by about 20 percent between 1998 and 2000, but remained relatively stable between 2000 and 2006.
- In Boca Ciega Bay, seagrass composition transitions from a *Halodule*-dominated community in the northern part to a *Thalassia*-dominated community in the south. Seagrass occurrence remained generally consistent during the study period, although changes in composition (for example, transitions from *Thalassia* to *Syringodium* dominance) occurred in some localized areas.

The transect monitoring program has also produced information regarding the depth ranges of seagrass beds and variations in seagrass-species zonation along longshore bars in the different bay segments (Avery and Johansson, 2008).

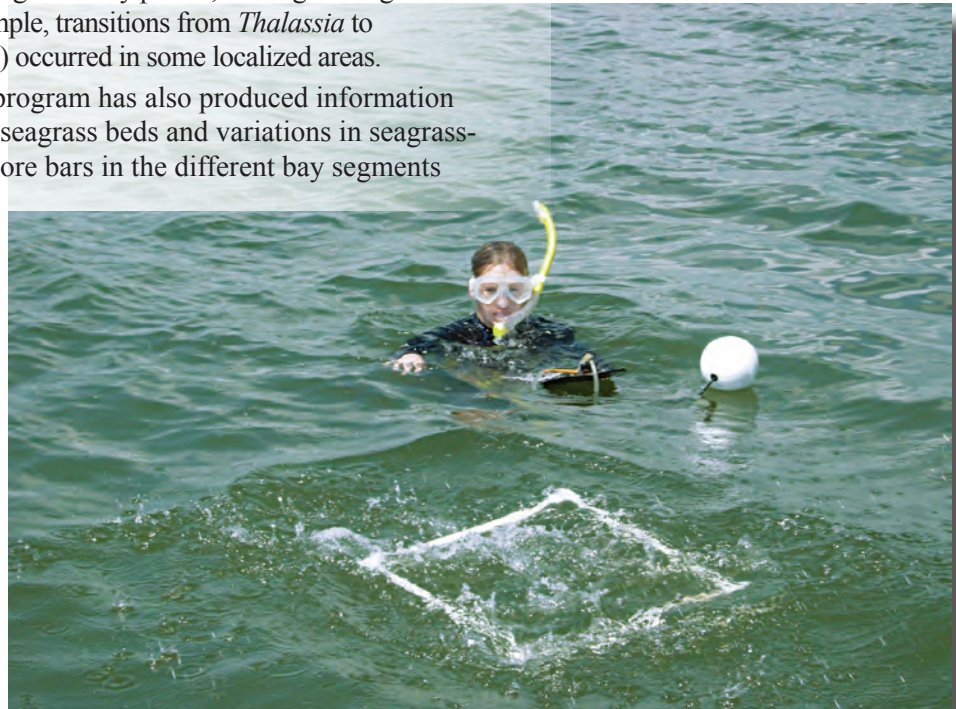


Figure 4–11. How randomly placed quadrants are used to monitor sites within seagrass transects. Photo by Jennifer Kunzelman, Fish and Wildlife Research Institute.

Box 4–1. Community Metabolism, Primary Production, and Irradiance Relations in Tampa Bay Seagrass Beds

By Kimberly K. Yates (U.S. Geological Survey—St. Petersburg, Florida)

Yates and others (2007) examined daily primary production and respiration rates of the biota within and immediately above seagrass beds and bare-sand habitats in the bay, using a submersible chamber known as the Submersible Habitat for Analyzing Reef Quality (SHARQ) and described by Yates and Halley (2003) (box 4–1, fig. 1). The SHARQ chamber was deployed for 24-hour periods at 17 locations in six different areas of the bay (box 4–1, fig. 2) during spring and fall, 2001–2003. Rates of gross daily primary production (P) and 24-hour respiration (R) by the plant and animal communities present within the chamber were recorded at each location.

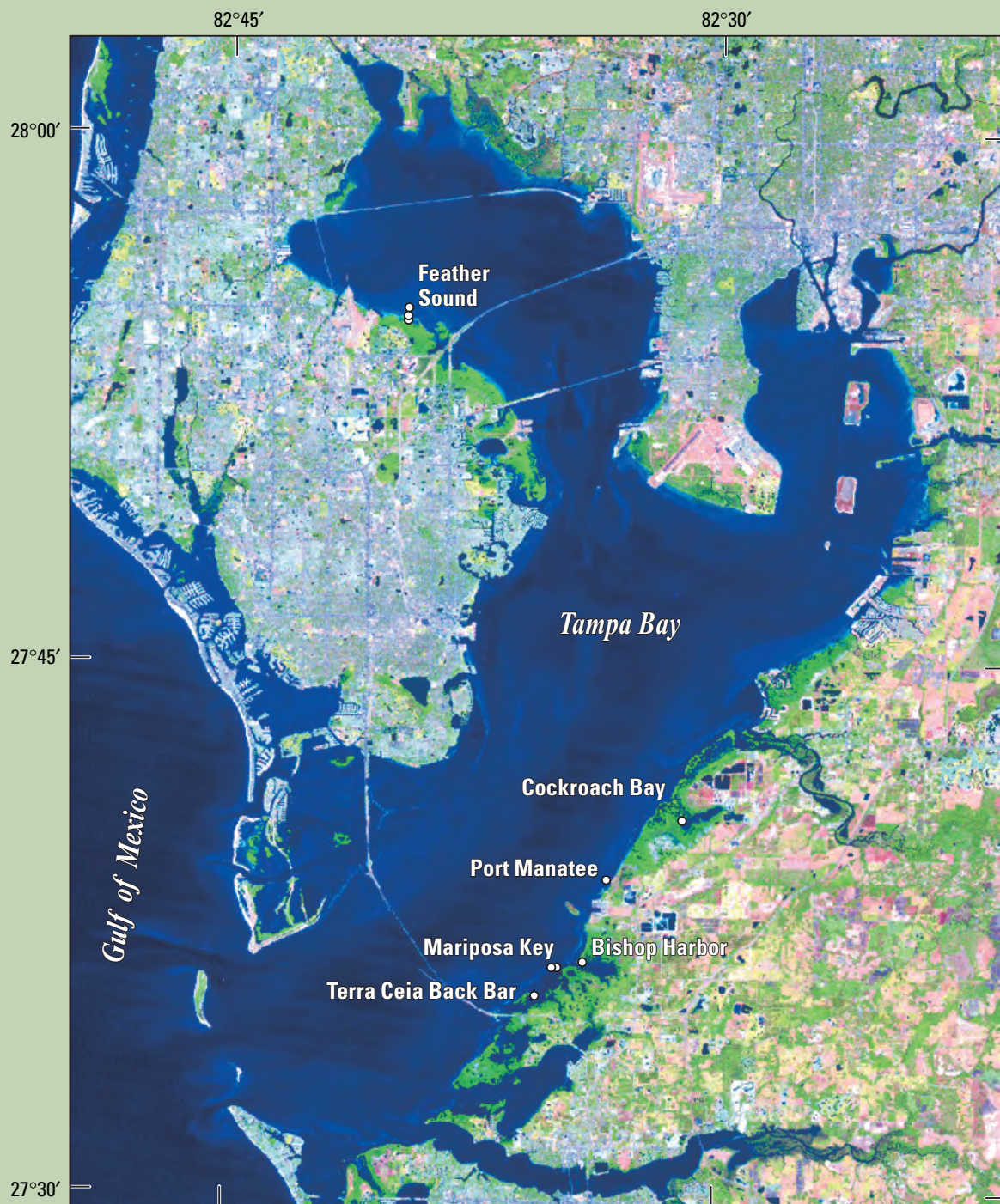
Relations between gross daily primary production (P) and irradiance (I) were measured during 2001–2003 SHARQ deployments. Results indicate that, in study areas along the southeastern shoreline of Tampa Bay, daily P–I relations can be described using the exponential or hyperbolic tangent curves that are typically used to delineate seagrass P–I responses (Zimmerman and others, 1994; Neely, 2000) (box 4–1, table 1). However,

in an intermediate-density *Halodule* habitat in Old Tampa Bay, the best-fit P–I relations were linear (box 4–1, fig. 3), showing no evidence of light saturation occurring even at PAR levels as high as 600 to 800 $\mu\text{E m}^{-2} \text{s}^{-1}$. The linear P–I relations observed in Old Tampa Bay were at PAR levels much higher than those that Neely (2000) found to cause light saturation in *Halodule* leaf segments collected from beds in Lower Tampa Bay. This indicates that seagrass responses to varying irradiance levels may differ substantially in different parts of the bay.

Similar levels of variability in P–I relations have been reported for other seagrass species and geographic regions, and in several cases have been related to spatial variations in water depth or seasonal variations in water clarity (Tomasko, 1993). In the case of Tampa Bay, the available data indicate that



Box 4–1, Figure 1. The Submersible Habitat for Analyzing Reef Quality (SHARQ) benthic incubation chamber. This chamber consists of an aluminum frame covered by clear, flexible plastic-sheeting to trap water over the seafloor enabling scientists to analyze water chemistry using flow-through analytical systems. Photo by U.S. Geological Survey.



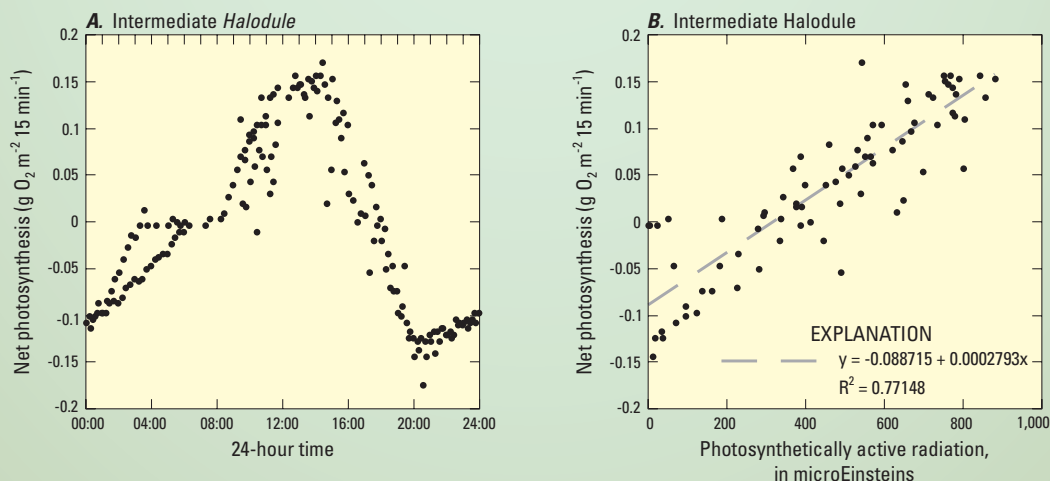
P-I relations may exhibit considerable spatio-temporal variability and that caution should be used when modeling seagrass productivity in the bay based solely on theoretical P-I curves (Yates and others, 2007). The variability observed in the bay may be related, in part, to the different sources of light attenuation, such as colored dissolved organic material, phytoplankton, and nonphytoplankton turbidity, which are present at different times in different bay segments.

Box 4-1, Figure 2. Submersible Habitat for Analyzing Reef Quality deployment locations. From Yates and others (2007).

Box 4–1, Table 1. Summary of community production and respiration observations collected during Submersible Habitat for Analyzing Reef Quality deployments during 2001–2003.

[Metabolic parameters for representative substrate types in Tampa Bay. P, daily gross production ($\text{g O}_2 \text{ m}^{-2}$), integration of metabolism curve with respect to $y + r$ from sunrise to sunset; R, 24-hour respiration ($\text{g O}_2 \text{ m}^{-2}$) calculated from the average nighttime respiration rate; P/R, ratio of daily gross production to 24-hour respiration; n, number of net photosynthesis and respiration data points collected and used to generate each metabolism curve; %, percent; Int., intermediate seagrass coverage. From <http://gulfsci.usgs.gov>]

Substrate description	Date	Hours sunlight	Linear regression equation $y = mx + b$	Correlation coefficient (R)	P	R	P/R	n
Bishop Harbor								
Int. <i>Thalassia</i> /100% drift algae	04/05/02	12.7	$y = 1.7164\text{e-}4x - 0.030808$	0.73	5.33	5.71	0.93	533
Int. <i>Thalassia</i> /50% drift algae	04/05/02	12.7	$y = 1.5943\text{e-}4x - 0.025627$	0.68	4.92	5.03	0.98	496
Cockroach Bay								
Int. <i>Thalassia</i> /decaying algae	04/30/02	13.2	$y = 1.494\text{e-}4x - 0.043975$	0.68	4.60	5.97	0.77	233
Dense <i>Halodule</i> /decaying algae	04/30/02	13.2	$y = 1.5374\text{e-}4x - 0.045568$	0.68	4.99	6.90	0.72	246
Feather Sound								
Int. <i>Halodule</i>	05/04/02	13.3	$y = 2.793\text{e-}4x - 0.088715$	0.87	6.56	7.44	0.88	172
Int. <i>Halodule</i>	05/04/02	13.3	$y = 2.8802\text{e-}4x - 0.10265$	0.88	6.53	8.17	0.80	154
Sand bottom	05/06/02	13.4	$y = 1.1341\text{e-}4 - 0.02149$	0.62	3.08	3.67	0.84	232
Sand bottom	05/06/02	13.4	$y = 1.0109\text{e-}4x - 0.022375$	0.41	3.77	4.01	0.94	246
100% drift algae	05/09/02	13.4	No light data	No data	6.91	5.94	1.16	170
Int. <i>Halodule</i>	05/09/02	13.4	No light data	No data	7.25	5.75	1.26	161
Mariposa Key								
Sparse <i>Thalassia</i>	04/19/01	13.0	$y = 1.406\text{e-}4x - 0.032323$	0.61	6.70	7.12	0.94	221
Int. <i>Thalassia</i>	04/19/01	13.0	$y = 9.8041\text{e-}5x - 0.009488$	0.54	5.69	6.12	0.93	300
Port Manatee								
Sparse <i>Syringodium</i>	05/14/01	13.5	No light data		6.16	6.12	1.01	224
Dense <i>Syringodium</i>	05/14/01	13.5	No light data		7.92	8.27	0.96	174
Terra Ceia Back Bar								
Sparse <i>Thalassia</i> /deep edge	10/07/03	11.7	$y = 1.0683\text{e-}4x - 0.016921$	0.71	2.35	3.23	0.73	289
Int. <i>Thalassia</i>	10/11/03	11.6	$y = 2.98\text{e-}4x - 0.030689$	0.66	2.82	4.70	0.60	222
Int. <i>Thalassia</i>	10/11/03	11.6	$y = 3.9056\text{e-}4x - 0.046291$	0.73	3.24	5.39	0.60	200



Box 4–1, Figure 3. A, Diurnal net primary productivity; and B, primary production and irradiance relation, observed within a Submersible Habitat for Analyzing Reef Quality incubation chamber during a 24-hour deployment within a *Halodule* bed in the Feather Sound part of Old Tampa Bay, May 4–6, 2002. From Yates and others (2007).

Factors Affecting Seagrass Cover in Tampa Bay

Analysis of aerial photographs, habitat maps, and urbanization extent shows that seagrass cover in Tampa Bay has fluctuated markedly during the roughly six-decade period (1950–2006) for which bay-wide photography is available. Cover declined by an estimated 22,000 acres (or about 50 percent) (Haddad, 1989) between the early 1950s and early 1980s, during a period characterized by widespread physical impacts (for example, dredging and filling) and increasingly poor water quality (Johansson, 1991, 2002a). Cover then increased by about 6,700 acres (or about 31 percent) from the early 1980s through 2006, as the size and frequency of physical impacts were reduced and water quality improved (Greening and Janicki, 2006). Crane and Xian (2006) noted an inverse relationship between urbanization and seagrass extent from 1940 through 1985, after which both increasing urbanization and seagrass expansion have been observed (figs. 4–12 and 4–13).

Evaluations of these changes, along with information from the literature regarding causes of fluctuations in seagrass cover observed in other geographic areas (for example, Batiuk and others, 1992, 2000; Dawes and others, 2004; Orth and others, 2006) provide insights into the following environmental factors that appear to affect the distribution and densities of seagrass meadows within the bay.

Water and Sediment Quality

As noted, water clarity plays a major role in determining the amount of sunlight that is available to support seagrass growth. Water clarity and light penetration in Tampa Bay are affected by a number of factors, including phytoplankton densities, concentrations of colored dissolved organic material, and levels of nonphytoplankton turbidity (McPherson and Miller, 1994; Janicki and Wade, 1996). Much of the water-quality-related seagrass decline that occurred in Tampa Bay between the 1950s and early 1980s took place in the upper (northern) parts of the bay, which were most strongly affected by nutrient loadings from the surrounding watershed (Haddad, 1989). A large percentage of those declines was apparently caused by reductions in water clarity that were brought about by increased phytoplankton densities associated with elevated nutrient influxes that occurred during the period (Johansson, 1991, 2002a), a topic that is addressed in greater detail in Chapter 5. The increased nutrient influxes also stimulated the growth of large amounts of drift macroalgae, particularly in Hillsborough Bay, which may have impacted seagrasses through shading, abrasion, and hypoxia (Johansson and Greening, 2000).

Sediment quality can also affect the growth and distribution of seagrasses, although this topic has not yet received the level of study devoted to water clarity and light-related issues. Sediment grain size distribution and organic content seem to be particularly important factors due to their effects on dissolved oxygen (DO) levels and concentrations of sulfide and other phyto-toxins in pore water (Koch, 2001). Sulfide, which can be toxic to seagrasses at concentrations >1 millimolar, occurs commonly in anoxic estuarine sediments as a byproduct of bacterial nutrient remineralization (Carlson and others, 2002). Sulfide toxicity may become more prevalent with increasing eutrophication and reduced light availability, if seagrass photosynthesis rates are not

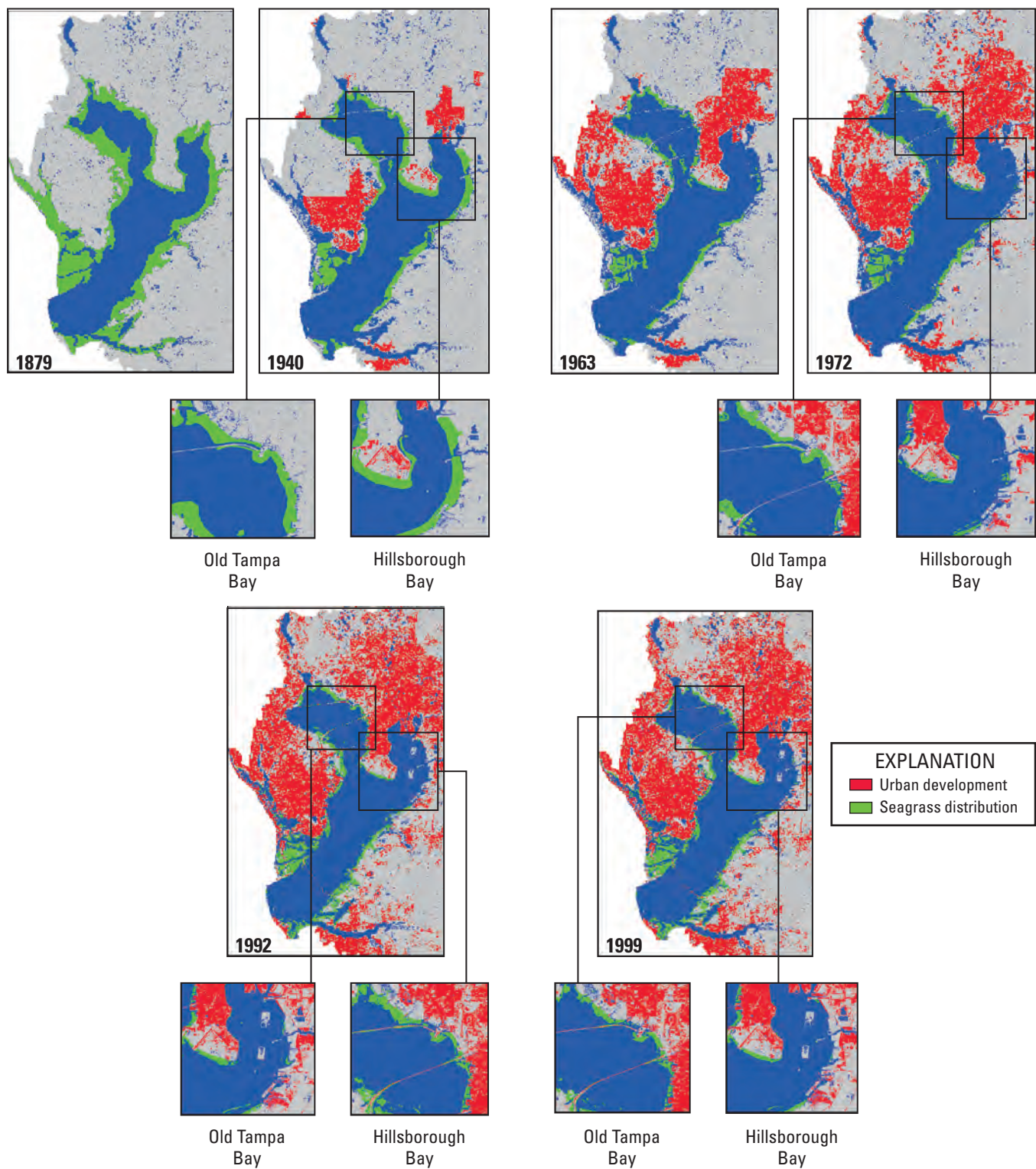


Figure 4-12. Urban development (red) and seagrass distribution (green) from 1879 to 1999. Map by Crane and Xian (2006).

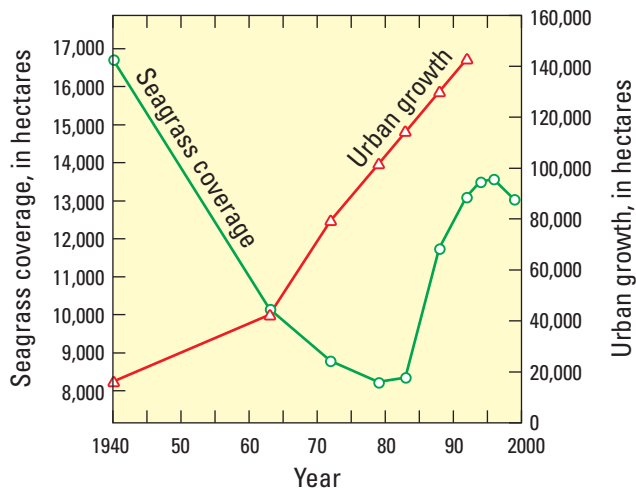


Figure 4-13. Comparison of urban growth and seagrass coverage in Tampa Bay. Data from Crane and Xian (2006).

adequate (at reduced light levels) to provide the oxygen necessary to maintain an oxidized zone adjacent to the plant roots (Calleja and others, 2006). Sediment organic content also seems to affect seagrass distribution, with healthy beds reported most frequently in sediment with <5 percent organic matter, perhaps due to elevated phytotoxin concentrations and/or reduced nutrient availability at levels >5 percent (reviewed by Koch, 2001).

Anthropogenic sediment contaminants may also affect seagrasses and their faunal associates. Levels of metals in sediments and seagrass tissue from 15 different locations throughout Old Tampa Bay, Middle Tampa Bay, and Lower Tampa Bay were examined by Fernandez and others (see Chapter 7, box 7-2). Concentrations of nickel, copper, and zinc in seagrass tissues exceeded concentrations in surrounding sediment at all 15 sample locations. Concentrations of lead in seagrasses exceeded sediment concentrations in 13 locations, chromium in 12 locations, and arsenic in 6 locations, indicating bioaccumulation of metals in seagrasses at these sites. The results suggest that more in-depth study of the potential impacts of metal contaminants on seagrass growth may be warranted.

Dredge and Fill

Dredge-and-fill activities in the bay, carried out to enlarge and maintain shipping channels and to develop the shoreline for residential, commercial, and industrial land uses (figs. 4-14 and 4-15), can impact seagrass meadows in two ways. The first is through direct physical impact, such as by removing the plants and the underlying sediments where they are growing, or by killing the plants by covering them with a thick layer of fill material. The second is through indirect effect, such as reducing water clarity by increasing turbidity (Lewis, 1977; Janicki and others, 1995).

As noted earlier, the human population of the Tampa Bay watershed began a period of rapid increase in the late 1940s and early 1950s (see Chapter 1, fig. 1-5). Accompanying this growth, several urban centers were constructed on or near the bay shoreline. Dredge-and-fill techniques were used to provide shoreline uplands for urban development and to construct port facilities, commercial shipping channels and their associated spoil disposal islands (see Chapter 2, fig. 2-1). For areas within and immediately



Figure 4–14. Residential development and finger canals along Boca Ciega Bay. Photo by Southwest Florida Water Management District.



Figure 4–15. Dredge operation in Tampa Bay port area. Photo by Nanette O'Hara, Tampa Bay Estuary Program.

adjacent to the bay, this construction activity peaked in the 1950s and 1960s, causing substantial changes to the bathymetry, tidal prism, and flushing characteristics of the bay (Goodwin, 1987, 1991) and the loss of substantial amounts of seagrass.

As a result of this activity, an estimated 12,800 acres of shallow-water habitats, whose preconstruction water depths were typically less than 2 m, were physically altered to such an extent that they are not likely to support seagrass beds in the future (Janicki and others, 1995). About 5,100 acres of Boca Ciega Bay (26 percent of that bay segment's historical shallow-water area) were filled for residential and commercial development. Large areas of the shallows in Old Tampa Bay (2,800 acres; 9 percent) and Hillsborough Bay (1,900 acres; 24 percent) were filled or channelized for urban and port development. From a practical perspective, these areas are considered nonrestorable (Janicki and others, 1995) and have been subtracted from the seagrass-restoration acreage targets developed for Tampa Bay (TBEP, 2006).

Currents and Wave Energy

Hydrodynamic forces, such as tidal currents and wind-driven waves, affect the physical structure and ecological functioning of seagrass habitats (Fonseca and Bell, 1998; Robbins and others, 2002; Koch and others, 2006b). They influence a number of habitat characteristics, such as the size, shape, and spatial patchiness of seagrass beds, and the biomass, shoot density, and percent cover of plants within beds. These habitat factors, in turn, affect the distribution and densities of benthic invertebrates and other fauna on a number of spatial scales, thus playing potential roles in macrofaunal community dynamics that have not yet received a great deal of study (Turner and others, 1999).

Hydrodynamic conditions also affect the ability of seagrass beds to become established and the susceptibility of existing beds to storm events (Fonseca and others, 2000, 2002; Robbins and Bell, 2000). High current velocity or wave energy can have a number of adverse impacts that produce site conditions less favorable for plant growth. These include direct physical effects, such as preventing plants from becoming established in constantly shifting sediments, damaging existing plants, or eroding beds, as well as indirect effects on characteristics, such as sediment grain size and the frequency of sediment resuspension. Although water clarity and light availability typically determine the maximum depth (Z_{\max}) to which seagrass beds can extend, a combination of aerial exposure effects, including desiccation and freezing, and wave energy usually determines the minimum depth (Z_{\min}). Z_{\min} is typically below the mean low water line, at a depth equal to the mean low water line, plus the depth of wave mixing (Koch, 2001).

For wind-driven waves, the relative exposure index concept, which combines the effects of wind direction, velocity, duration, and fetch, has shown promise for modeling the spatial distribution of seagrass patches within individual estuarine systems (Fonseca and Bell, 1998; Fonseca and others, 2002; Robbins and others, 2002). For estuaries with extensive shallow areas, however, where waves are simultaneously being generated by

wind and dissipated by interactions with vegetation and bottom sediments, modeling of wave energies and their impacts on seagrass habitats can be problematic (Koch, 2001).

Feedback effects are also present, in the sense that dense seagrass beds can reduce current velocities and wave energy as well as being affected by them, adding an additional level of complexity to the interactions that occur between hydrodynamic forces and seagrass habitats (Koch, 2001; Robbins and others, 2002). In areas where water flow is greatly reduced, either by dense seagrass beds or local geomorphological features, stagnant conditions can lead to elevated levels of sediment organics, sulfides, and other physical or chemical constituents that are detrimental to seagrasses (Robblee and others, 1991; Carlson and others, 2002). Given the deleterious effects that can occur in very high and very low energy regimes, areas that experience moderate current velocities and wave-energy levels seem to be optimal for seagrass growth and development (Koch, 2001; Koch and others, 2006b).

In Tampa Bay, Lewis and colleagues (Lewis and others, 1985; Lewis, 2002, 2008) have emphasized the potential importance of the bay's longshore bars as physical features that may help protect seagrass beds from excessive wave energy, providing relatively sheltered areas that are favorable for seagrass survival and growth (fig. 4–16). Lewis (2002) noted that the overall lengths of longshore bars have been reduced in recent decades and hypothesizes that such losses might act to slow or prevent the achievement of seagrass-restoration goals in some parts of the bay. Lewis (2002) also noted that reductions in seagrass cover in the bay between the late 1800s and early 1980s — particularly the loss of deeper seagrass beds from the bayward side of bars — may have been an initial step in allowing bars to become destabilized and then dispersed by wave energy (fig. 4–17).

To test these hypotheses, a multientity task force coordinated by the TBEP and funded by a number of local, State, and Federal partners is evaluating the status and evolution of the longshore bar system and the effects of wave energy and tidal-current velocities on the distribution and condition of seagrass beds in different parts of the bay. Those studies, currently in progress, have produced a number of preliminary findings. Fonseca and others (2002) used a relative exposure index model to demonstrate that a combination of regional wind patterns and local geomorphology makes the eastern side of the bay more exposed and vulnerable to wind-driven waves than the more sheltered western side. For the eastern side, the model predicted that the presence of protective structures, such as longshore bars, could increase the probability of seagrass cover on their lee sides, if the locations and sizes of the bars allowed them to produce sufficiently large reductions in relative exposure index values (Fonseca and others, 2002) (fig. 4–18).

Similarly, Robbins and others (2007) used arrays of submersible tide and wave recorders to measure the energy levels of natural (wind-driven) and anthropogenic (ship-generated) waves at five study locations in north-eastern Tampa Bay. Preliminary results indicate that, although ship wakes are not a dominant wave-generating factor in the bay, they periodically create abnormally high waves in some areas. These “rogue waves” (fig. 4–19) may impact seagrasses and increase shoreline erosion, as well as affect shorebird nests located in exposed areas. Initial wave-energy measurements indicated that, although extant longshore bars attenuate waves effectively, remnant



Figure 4-16. Location of longshore bars in 2004 in Tampa Bay. From Lewis (2008).

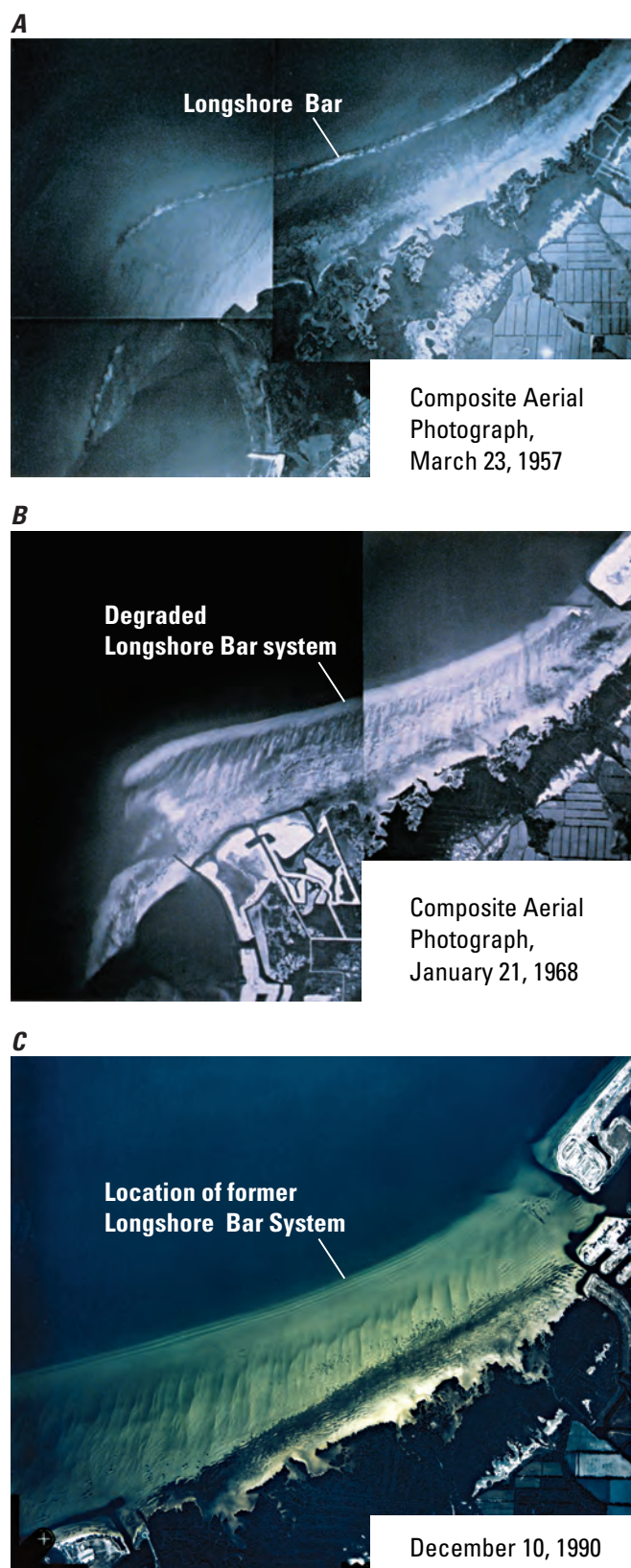


Figure 4-17. Longshore-bar degradation on the eastern shore of Tampa Bay near Apollo Beach on *A*, March 23, 1957; *B*, January 21, 1968; and *C*, December 10, 1990. From Lewis (2008).

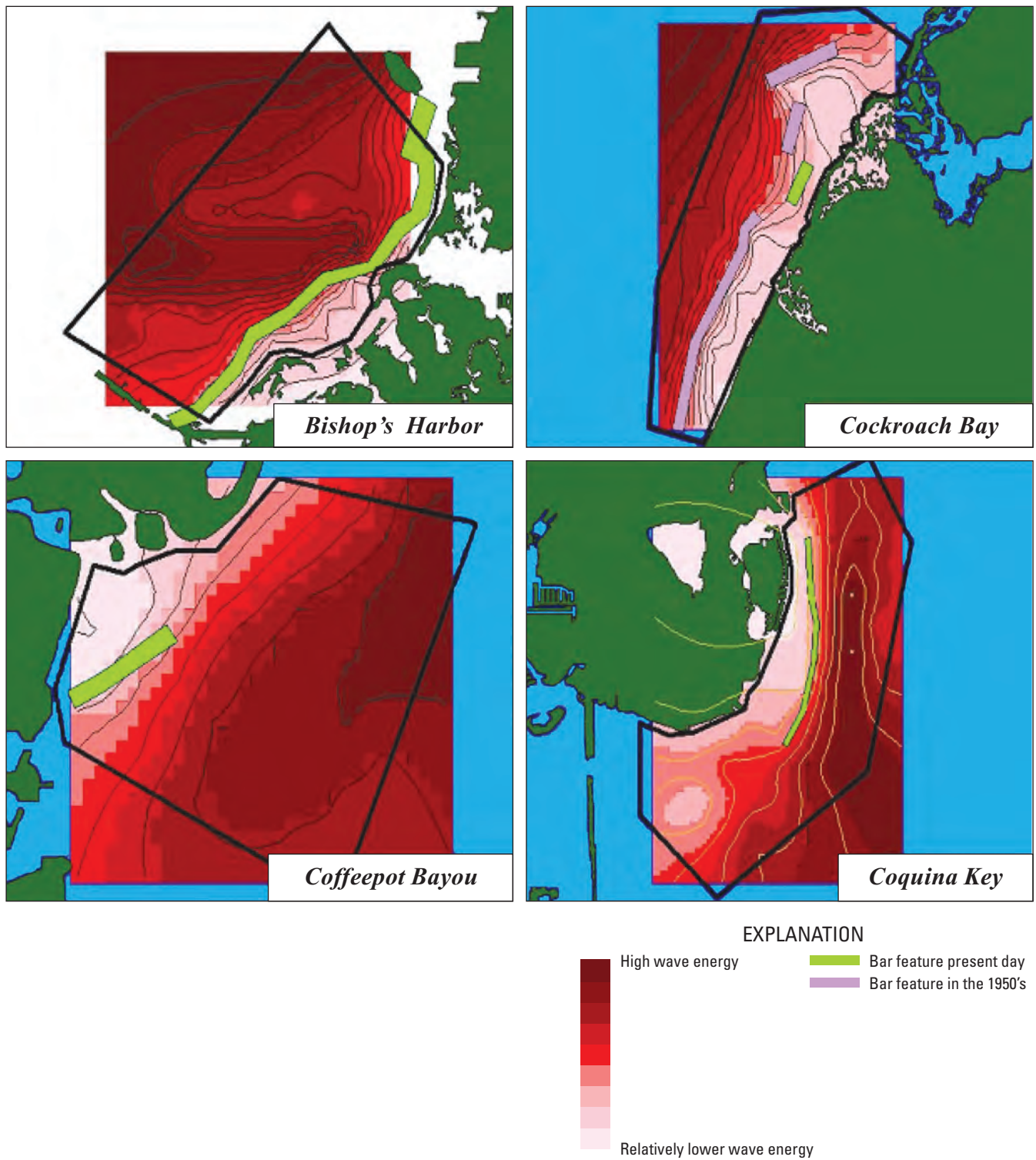


Figure 4-18. Relative exposure index value and bar locations at four Tampa Bay sites. From Fonseca and others (2002).



Figure 4–19. Rogue wave (about 2 feet high) observed offshore of MacDill Peninsula, 2003. Photo by Brad Robbins, Mote Marine Laboratory.

bars do not. In addition to wave energy, Robbins and others (2007) note that tidal scour may be a factor affecting longshore-bar condition and seagrass recovery in their study areas.

The hypothesis that seagrass beds located on the deeper, offshore side of longshore bars may help to increase bar stability and persistence (Lewis, 2002) is also being examined. Johansson and others (2007) initiated field experiments in 2006 to test the ability of planted *Syringodium* patches to trap and accumulate sediments and, thus, create a longshore bar in an area of Tampa Bay that currently lacks such a feature. Several years of observation will likely be needed (Bell and others, 2007) to determine the effects of the plantings.

Propeller Scarring

In Tampa Bay, propeller scarring is a frequent type of boating impact (Sargent and others, 1995; EPCHC, 2007), with aerial photography indicating that scarring intensity tends to be highest in shallow areas near boating channels, around passes between islands, and in shallow seagrass beds on the lee sides of islands and longshore bars (EPCHC, 2007) (fig. 4–20).

Regrowth of seagrasses into propeller scars occurs slowly in the bay, requiring periods of 3.6 to 6.4 years if no additional scarring occurs during the regrowth period (Durako and others, 1992) and more than 7 years for *Thalassia* regrowth in some cases (Dawes and others, 1997). Effects of propeller scarring on seagrass habitat use by several fish and invertebrate species in the bay were investigated by Bell and others (2002), who found no major reductions in faunal abundance in moderately scarred beds but suggested that intense propeller scarring over an expansive area could potentially impact the long-term stability of seagrass beds and the distributions and densities of their faunal inhabitants. When scarring becomes severe, habitat function can be lost, and the entire bed may lose the ability to regenerate and cease to exist (Stowers and others, 2002).

Sargent and others (1995) recommended a four-part management approach to reduce boating impacts in heavily scarred areas: increased boater education, channel marking, on-water enforcement, and implementation of



Figure 4–20. Propeller scarring in seagrass beds from recreational boating. Photo by Fish and Wildlife Research Institute.

special protective zones where internal combustion engine use is restricted. Several components of this strategy are being implemented in parts of Tampa Bay, particularly in Aquatic Preserve areas located in the vicinity of Weedon Island, Mullet Key, and Cockroach Bay (Stowers and others, 2002; EPCHC, 2007). Changes in scarring rates in these areas will be tracked as part of the bay-wide seagrass-management effort.

Planting and Transplanting

Seagrass planting and transplanting projects are often undertaken in an effort to restore cover in areas where coverage has been reduced or lost, or as mitigation for anthropogenic impacts to existing seagrass beds (Treat and Lewis, 2006). Although planting is not a technically complex process, it is costly, averaging \$240,000 to \$393,000 per acre, and has not produced high success rates (Fonseca, 2006). Based on a national survey, Fonseca and others (1998) noted that only about 50 percent of the projects whose outcomes could be documented had resulted in successful establishment of seagrass cover. Success was reported most frequently at locations where seagrass habitats existed, but had been disturbed by physical impacts that had been corrected or eliminated. Lack of success was most commonly due to site-selection issues, such as the presence of improper light regimes, bioturbation levels, wave energies, or other environmental conditions at the planting site (Short and others, 2006; Fonseca, 2006).

Bell and others (2007) used a replicated, multiplot experimental design to study success rates of transplanted *Halodule wrightii* in Boca Ciega Bay between 2002 and 2005. The study area was an unvegetated sand flat adjacent to existing *Halodule* beds, and was selected based on aerial photography showing that seagrass had previously been present. Four planting techniques were evaluated — three using manual planting methods, with units held together by rubber bands, enclosed in peat pots, or attached to wooden staples, and one using a mechanical (boat-based) planting procedure (fig. 4–21) applied to units held together by rubber bands.

Planting occurred in June 2002, and subsequent changes in seagrass cover in the planted plots and in unplanted control plots were measured annually through July 2005. Natural seagrass beds in the area were examined over the same period as reference sites. Patterns of seagrass-bed development in the planted plots were patchy and highly variable both within and between plots. By the second and third years of the study, however, statistically detectable differences in seagrass cover had developed in planting areas, with the highest levels of cover occurring in shallow water and within close proximity of natural seagrass beds. Significant differences in seagrass cover were detected among planting methods and over time (2002–2005); manual planting using units held together by rubber bands resulted in the highest cover, and the peat pot and mechanical planting methods produced substantially lower cover. Increases in *Halodule* cover in the planted plots occurred at lower rates than previously reported in the literature for natural beds. Bell and



Figure 4–21. Mechanical seagrass planting boat developed and operated by Seagrass Recovery, Inc. This boat is called a “GigaUnit Transplant System” and was designed for the removal, transport, and replanting of large (1.8 square meters) planting units (or sods) of submersed aquatic vegetation along with the roots, rhizomes and associated sediments. Photo by Jim Anderson, Seagrass Recovery, Inc.

others (2007) concluded that the time required for full restoration in planted beds may greatly exceed typical (for example, 3-year) monitoring periods required by many permitting agencies, and the development of large areas (tens to thousands of square meters) of cover for seagrass species that expand largely via rhizome elongation may be a process that requires decades.

Seagrass planting techniques continue to be revised for use in Tampa Bay, including the use of larger plugs with sediment intact (Johansson and others, 2007) (fig. 4–22), and excavating and transplanting seagrass sods (P. Hall, personal commun., 2007) (fig. 4–23). However, although planting is no longer an experimental process and seagrass beds can be restored with various planting and transplanting techniques, it is much more cost-effective to protect existing beds and restore environmental conditions, such as water quality and light regimes that promote their expansion (Fonseca and Bell, 1998; Treat and Lewis, 2006).



Figure 4–22. *Syringodium* transplant plug, MacDill Peninsula, 2007. Photo by Holly Greening, Tampa Bay Estuary Program.



Figure 4–23. Seagrass sod (1.8 square meters) being excavated by Seagrass Recovery GigaUnit Transplant System, prior to transplanting. Photo by Penny Hall, Fish and Wildlife Research Institute.

Sea-Level Rise and Other Components of Global Climate Change

Using information provided by the USEPA (Titus and Narayanan, 1995), the Tampa Bay Regional Planning Council (TBRPC, 2006) has projected future rates of sea-level rise in the bay region. The median projected increases are 13 cm by the year 2025, 24 cm by 2050, 37 cm by 2075 and 50 cm by 2100 (TBRPC, 2006). Currently, the average depths of the deep edges of seagrass beds range from about 31 cm in highly turbid upper Hillsborough Bay to about 2 m in clearer-water parts of Middle and Lower Tampa Bay (Johansson, 2002b). In addition, the shallow shelves on which these beds occur have very gradual shoreward slopes. If the increases in sea level projected by the TBRPC (2006) occur, and the water clarity and maximum depths of seagrass growth remain at present levels, beds in a number of areas may be required to migrate considerable distances during the next 25 to 100 years in order to remain in water depths that allow continued survival and growth (Glick and Clough, 2006). Neither the potential migration distances nor the potential effects of the projected sea-level rise on future acreages of seagrass cover in the various segments of the bay have been calculated. Concern exists, however, that the presence of seawalls, bulkheads, and other forms of hardened erosion-control structures that have been installed along many parts of the bay shoreline may have a “telescoping” effect on water depth zones as sea-level rises, potentially reducing the total area of the bay that has the appropriate water depths to support seagrass growth under the existing water clarity and light penetration regimes.

In addition to sea-level rise, increasing emissions of CO₂ and other greenhouse gases are projected to cause a number of global environmental changes that may impact seagrasses and other marine life forms. As summarized by Bjork and others (2008), these changes could potentially include:

- Increased CO₂ concentrations, which may cause increases in seagrass photosynthesis and growth rates;
- Lowered seawater pH, which could also increase seagrass production by countering the high pH (and reduced availability of inorganic carbon) that can occur as a result of photosynthetic activity in dense seagrass stands;
- Changes in water temperature, which can affect seagrass physiological processes, growth rates, and reproduction patterns, and alter the geographic distribution of species based on their temperature tolerances;
- Changes in PAR and UV radiation, whereby the PAR received by existing seagrass meadows may be reduced as a result of increased cloud formation, and harmful UV radiation may increase due to diminished ozone levels;
- Changes in the intensity of tropical storms, which can impact seagrasses by uprooting or burying plants and by decreasing light availability as a result of increased turbidity;
- Increased rainfall and flooding, which can impact seagrasses by producing rapid fluctuations in salinity and elevated turbidity levels due to increased terrestrial runoff; and

- Increased sediment hypoxia and anoxia, as a result of higher water temperature and more rapid decomposition of organic matter in the sediments, which could lead to more frequent episodes of sulphide toxicity and other stresses associated with reduced O₂ availability.

Status and Trends in Seagrass Cover

Seagrass acreage in Tampa Bay has been estimated for a number of time periods from the late 1800s through 2008 (fig. 4–24), using a variety of techniques. Estimates for 1950 and subsequent periods were developed using aerial photo-interpretation based on similar vegetation classification and interpretation methods, and are thought to be sufficiently comparable to support trend analyses (Haddad, 1989; Johansson and Ries, 1997; Kurz, 2002; TBEP, 2006). The 1950 and 1982 estimates were developed through a joint project conducted by the U.S. Fish and Wildlife Service and the Florida Department of Natural Resources (Haddad, 1989). Since 1988, seagrass maps have been produced about every 2 years by SWFWMD through a process described by Johansson and Ries (1997), Kurz (2002), and Tomasko and others (2005). The estimate for the late 1800s was developed using a combination of aerial photographs taken during the late 1930s and early 1940s and historical maps dating back to 1848 (Lewis and others, 1985). Because of these methodological differences, the late 1800s estimate is thought to be less comparable than the more recent estimates, but it provides helpful information on the general direction and possible magnitude of long-term changes in seagrass cover (Haddad, 1989).

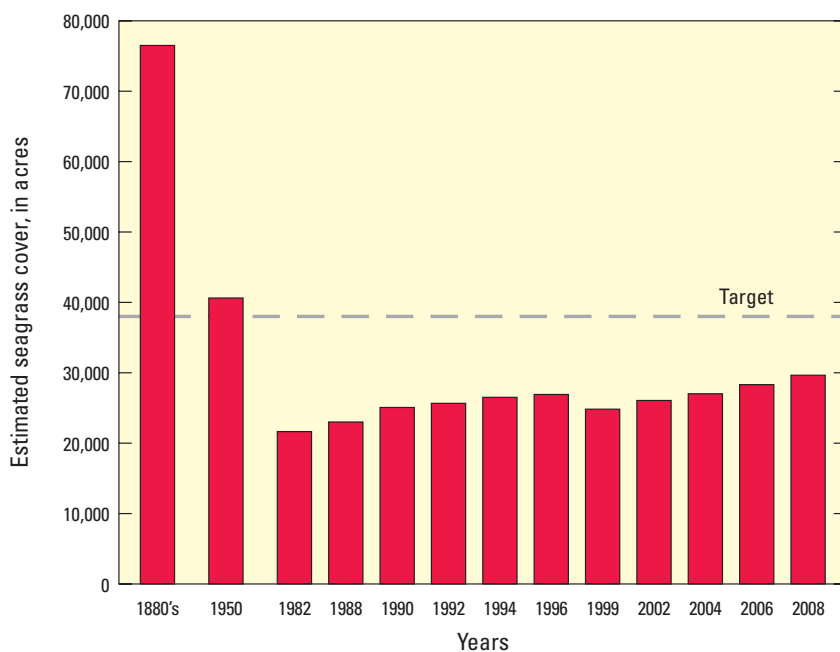


Figure 4–24. Estimated long-term changes in Tampa Bay seagrass cover and target cover established in living-resource strategy. Data from Lewis and others (1985) and Southwest Florida Water Management District.

Based on these estimates, seagrass cover in Tampa Bay decreased by about 19,000 acres between 1950 and 1982, followed by an increase of 5,200 acres between 1982 and 1996. Between 1996 and 1999, a period that included particularly heavy rainfall and stormwater runoff associated with a strong 1997–1998 El Niño event, seagrass cover decreased by about 2,000 acres, followed by recovery and an increase of about 2,200 acres between 1999 and 2004 (Tomasko and others, 2005; Greening and Janicki, 2006), and an additional increase of about 2,600 acres between 2004 and 2008 (fig. 4–24). Bay-wide, the 29,647 acres of seagrass mapped in 2008 was the highest value observed since 1950, but remained about 11,000 acres below the estimated 1950 coverage.

Seagrass Management Strategy

During the mid-1990s, the TBEP and its partners adopted a living resource-management strategy that linked the management of water quality and N loading to seagrass protection and restoration (Greening, 2002; Greening and Janicki, 2006; TBEP, 2006). The strategy was developed through the following series of steps (which are described in greater detail in Chapter 5):

Step 1: Set specific, quantitative seagrass acreage goals

In 1996, the local resource-management community adopted a minimum seagrass cover target goal of 38,000 acres (fig. 4–24), which represents 95 percent of the acreage that was estimated to have been present in the bay in the early 1950s. The early 1950s time period was selected as the baseline for seagrass cover, because it preceded the rapid population increases that have occurred in the watershed in more recent decades, and because aerial photographs were available for the entire Tampa Bay shoreline and adjacent shallow waters for that period (Greening, 2002; Greening and Janicki, 2006). Areas that have been rendered nonrestorable by dredging, filling, and the construction of causeways and other infrastructure since the early 1950s are not included in the restoration goal.

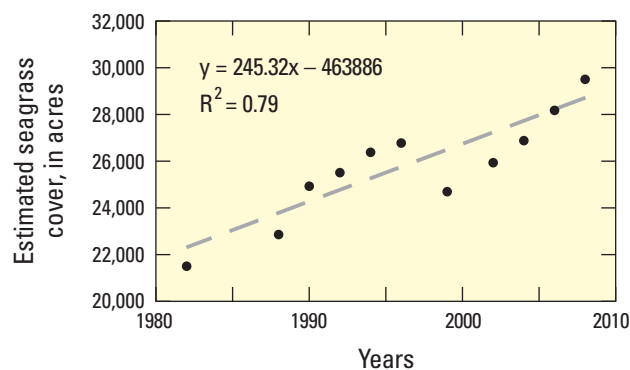


Figure 4–25. Seagrass-cover trend from 1982 through 2008. Data from Southwest Florida Water Management District.

Step 2: Determine the light requirements of the target seagrass species (*Thalassia testudinum*) in Tampa Bay

As noted, field studies carried out in stable *Thalassia* meadows in Lower Tampa Bay indicated that the deep edges of those seagrass beds corresponded to the depth at which about 20.5 percent of subsurface irradiance reached the bottom of the water column on an annual average basis (Dixon, 2002). Irradiance, I_o , is the sunlight that penetrates the water surface.

Step 3: Determine the water clarity levels necessary to provide adequate light to meet the seagrass acreage goals

Based on the 20.5 percent light requirement estimated in Step 2, the seagrass acreage restoration goal was restated as a light penetration and water clarity target. That is, to restore seagrass acreage to early 1950s levels in a given bay segment, water clarity in that segment should be restored to a point that allows at least 20.5 percent of I_o to reach the same average bottom depth that it reached during the early 1950s. Those depths were estimated to range from about 1 m in Hillsborough Bay to about 2 m in Lower Tampa Bay (Janicki and Wade, 1996; Greening and Janicki, 2006).

Step 4: Determine maximum chlorophyll *a* concentrations that allow water clarity to be maintained at appropriate levels

Water clarity and light penetration in Tampa Bay are influenced by a number of factors, including phytoplankton density (estimated using measured chlorophyll *a* concentrations), colored dissolved organic material, nonphytoplankton turbidity, and the water itself. Janicki and Wade (1996) used regression analyses applied to long-term EPCHC monitoring data to develop an empirical model summarizing water clarity variations in response to these factors in the four major bay segments. The results indicated that variation in observed depths at which 20.5 percent of surface irradiance penetrates could be explained by variation in observed chlorophyll *a* concentrations. Based on the regression models, annual average chlorophyll *a* targets ranging from 4.6 to 13.2 $\mu\text{g/L}$ in the four major bay segments were adopted by the TBEP partners (TBEP, 2006).

Step 5: Determine maximum nitrogen (N) loadings that allow chlorophyll *a* concentration targets to be achieved

Analyses of water-quality data and models indicated that, by managing the influx of N that enters the bay each year, chlorophyll *a* concentrations can be maintained within target levels. Examination of long-term water-quality data indicated that the water clarity conditions that existed in the bay during 1992–1994 were adequate to meet the water clarity targets that had been adopted in Step 3. A N load management strategy, based on “holding the line” at the annual loading levels estimated to have occurred during the 1992–1994 period, was, therefore, adopted by the TBEP and its partners (Greening and Janicki, 2006).

Step 6: Implement the management approach and assess its effectiveness

Successful long-term achievement of the water-clarity targets and the “hold-the-line” N management strategy could be prevented as a result of ongoing population growth in the watershed, which could potentially increase N discharges to the bay through increased wastewater and storm-water discharges. If estimated N loads are to be maintained at or below 1992–1994 levels, additional N management projects may be necessary to preclude expected load increases that would otherwise be generated as a result of population growth (Greening and Janicki, 2006). An average annual N load reduction of about 17 tons/yr would compensate for expected population growth in the watershed.

To achieve these additional load reductions, the Tampa Bay Nitrogen Management Consortium was formed, which is made up of electric utilities, other industries, agricultural interests, local governments and regulatory agencies and sponsored by the TBEP (TBEP, 2006). Together, the entities participating in the consortium developed a N management action plan and committed to implement projects that will contribute to meeting the adopted N management goal. The types of projects included in the plan range from traditional nutrient-reduction activities, such as wastewater- and stormwater-treatment upgrades, industrial retrofits, and agricultural best management practices, to actions that provide nutrient reductions as a secondary benefit, such as land acquisition and habitat restoration projects.

The effectiveness of the “hold-the-line” N-management strategy is assessed annually by evaluating chlorophyll *a* concentrations and water clarity levels measured by local monitoring programs in each bay segment during the previous calendar year, and comparing those values to the segment-specific targets (Greening and Janicki, 2006). During years when the water-quality targets are not met, a “decision-matrix” approach (see Chapter 5) is used to determine an appropriate level of management response. The ongoing effectiveness of the seagrass management strategy is evaluated every 2 years, when changes in seagrass cover are mapped and provided in GIS format by the SWFWMD. The GIS data layers produced by the mapping effort are available online (for example, SWFWMD, 2006).

Anticipated Challenges

The initial phase of the Tampa Bay seagrass management program, extending from 1996 through 2009, focused on relatively broad issues for each of the bay’s major segments: the overall water clarity and illumination levels required for seagrass growth, and the development of annual average light-related water-quality targets. Refinements to the approach are currently underway as part of the ongoing TBEP management effort, and in coming years are anticipated to include:

- Development of light attenuation targets for specific bands of wavelengths within the photosynthetically active part of the electromagnetic spectrum that may be particularly important in seagrass photosynthesis;

- Improved identification of other potential stressors, such as localized stormwater influxes, elevated water temperature in areas near powerplant discharge points, excessive hydrodynamic forces or bioturbation, reduced sediment quality, and boating impacts that may be inhibiting seagrass recovery in some areas of the bay; and
- Refinement of the existing seagrass management areas within each bay segment, to allow assessments of seagrass status and trends and discrimination of potential stressors to be made on finer spatial scales.

An additional set of questions that may face managers in coming years involves the speed of seagrass recovery and the adequacy of the existing water clarity and light targets to maintain or increase the recovery rate. During the 1982–2008 period, estimated seagrass acreage in the bay increased at a rate of about 245 acres/yr (fig. 4–25). Under best-case conditions — that is, if no substantial losses of existing seagrass acreage occurs, and if cover continues to increase linearly at the rate shown in figure 4–25 — the present trend indicates that the TBEP restoration goal of 38,000 acres will be met around the year 2050. That timeline could potentially be delayed, however, if future weather conditions, such as hurricanes or strong El Niño episodes, damage or eliminate a part of the existing seagrass acreage, or if the annual rate of increase is reduced as a result of sea-level rise or other factors. Also, it is not known if the current recovery rate will continue to be acceptable to the local populace. In coming decades, the public and its policymakers could undertake actions to change the rate of recovery. If a more rapid recovery were desired, increases in the water clarity targets (with concomitant decreases in the chlorophyll *a* targets and N loading goals) would presumably be necessary, since higher light levels would allow more rapid expansion of seagrass acreage. In locations where shoreward migration of seagrass beds is prevented by topography or by the presence of seawalls or other hardened shorelines, higher light levels may also prove necessary to maintain existing levels of seagrass cover by allowing beds to continue to exist in their current locations. On the other hand, if efforts to maintain the current recovery rate come to be seen as technically unrealistic or overly costly, managers could decide to lower water clarity targets in response to economic constraints.

Over the long term, the greatest upcoming challenges facing the seagrass and water-quality management programs appear to be associated with human population growth. As noted, the population in the three-county area immediately adjacent to Tampa Bay has more than quadrupled since the early 1950s, increasing from fewer than 500,000 persons in 1950 to more than 2 million in 2000. The population in these three counties reached 2.4 million in 2006 and is projected to reach 2.8 million in 2015, 3.2 million in 2030, and more than 4 million by 2050. Since the early 1980s, watershed management actions aimed at preventing N influx and phytoplankton densities from exceeding target levels have been successful in compensating for population growth. Those efforts will need to continue to compensate for the anticipated future growth. These issues are discussed in more detail, from a water-quality perspective, in Chapter 5.

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