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## SEAGRASS BIODIVERSITY IN THE INDIAN RIVER LAGOON

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### ABSTRACT

All six species of seagrasses known from the tropical western hemisphere as well as *Halophila johnsonii* grow in the Indian River Lagoon (IRL). *Halodule wrightii* is the most common species, but *Syringodium filiforme* may be locally more abundant. *Thalassia testudinum* is limited to the southern portion of the IRL. The three *Halophila* species, *H. decipiens*, *H. englemannii*, and *H. johnsonii*, form mixed or monotypic beds with the larger species, while *Ruppia maritima* is found in the most shallow areas and is the least common. A pronounced seasonal growth pattern is evident for all species with maximum growth in April–May and maximum biomass in June–July. Wide ranges in productivity have been reported for the three larger seagrasses with spring levels approaching those found in other coastal communities in Florida. The distribution and ecological importance of the three species of *Halophila* may be much greater than previously thought as shown by their abundance in deeper water, high productivity, and rapid turnover. The seagrass communities of the Indian River Lagoon have been shown to support a diverse epiphytic and drift macroalgal flora and function as a habitat, nursery and food source for epifauna, macrobenthos, and manatees. Given the ecological importance of seagrasses, a high priority for management of the Indian River Lagoon must include protection and enhancement of these communities. Selection of beds for management should consider ecological function of the species, biodiversity of associated fauna, and the level of continued anthropogenic impacts.

The ecological importance of the submersed aquatic vegetation, including seagrasses, in estuaries and coastal zones throughout the world is well established (Larkum et al., 1989). Healthy seagrass communities serve as habitats and nurseries for invertebrates, fish, turtles and birds; they stabilize sediments and recycle nutrients, and are highly productive equaling that of agriculture crops (Dawes, 1981). Seagrass communities have been described as the marine analog of tropical rain forests because of their structural complexity, biodiversity, and productivity (Simenstad, 1994). There is no doubt that the seagrass communities of the Indian River Lagoon (IRL) have contributed to the lagoon's designation as the most biologically diverse estuarine system in the continental U.S. (Gilmore et al., 1983). Our paper reviews the present knowledge of the IRL seagrasses and their significance in the biodiversity of the IRL. The goal of our review is to demonstrate the importance of seagrasses and to offer management recommendations for the preservation and expansion of their communities.

*Indian River Lagoon.*—The IRL is located on the east coast of Florida, being part of the longest barrier-island and tidal-inlet system (260 km) in the United States comprising 40% of the Florida Atlantic Coast from 27°N, 80°W to 29°N, 81°W (Barile, 1993). The richness of the IRL is attributed both to its geographical location, where the warm temperate and tropical flora and fauna overlap, and its diverse coastal and submerged vegetation including seagrass beds, salt marshes, mangrove forests, and macroalgal communities. The inlets provide access for numerous fauna which enter the lagoon as larvae, juveniles or adults.

With the closing of the Suwanne Straits in the late Paleocene, silico-clastic sands from the southern Appalachian Mountains were carried by longshore currents and deposited on the carbonate platform of Florida (Davis et al., 1992; Stauble and McNeil, 1985). The present IRL was formed in the past 6,000 years

when coastal retreat and sea level rise slowed, and the overwash barrier islands stabilized (Davis et al., 1992). The present seagrass communities in the IRL are a result of rapidly changing geological and oceanic conditions during the past 3,000 years.

Water depth throughout most of the IRL is between 1 and 3 m. The IRL proper has three openings to the Atlantic Ocean: Sebastian, Fort Pierce and St. Lucie Inlets. Tidal flushing is limited to a few km around each inlet. Tidal residence periods range from a few days in the southern part to months in the north, where non-tidal flushing dominates (Smith, 1993) and water levels are primarily controlled by wind driven circulation (Barile, 1993). While salinities around the inlets are typically oceanic, salinities elsewhere in the lagoon are highly variable and a function of seasonal patterns of freshwater inputs and evaporation.

According to recent estimates (R. Virnstein, pers. comm.) there are approximately 40,000 ha of seagrasses in the IRL. Spatial distribution of seagrass beds range from isolated patches to continuous coverage of square kilometers. The lagoon supports all of the seagrasses known from the Caribbean including turtle grass, *Thalassia testudinum* Banks ex König; manatee grass, *Syringodium filiforme* Kützing; shoal grass, *Halodule wrightii* Ascherson; star grass, *Halophila engelmannii* Ascherson; paddle grass, *Halophila decipiens* Ostenfeld (as *H. baillonis* Ascherson), and widgeon grass, *Ruppia maritima* L. (Eiseman, 1974, 1980; Phillips, 1960; Virnstein and Cairns, 1986). The seventh species, new to the tropical western Atlantic, *Halophila johnsonii* Eiseman was first reported by Eiseman and Benz (1975) and later described by Eiseman and McMillan (1980).

*Seagrass Distribution.*—Current and historical patterns in seagrass distribution and abundance within the IRL are not well documented; detailed mapping and quantification is needed to identify, protect, and manage this resource. Aerial studies in the past were limited to sections of the lagoon (Thompson, 1978; Downs, 1983). Groundtruthing of aerial photographs was also limited and is necessary to identify seagrass species, distinguish macroalgae, and locate smaller species (e.g., *Halophila* spp.). A more detailed and systematic effort at seagrass mapping involving aerial mapping (1:24,000) and groundtruthing was initiated by Virnstein and Cairns (1986) and is currently being undertaken by the St. Johns River Water Management District. This mapping effort has found that seagrass beds within cities (Vero Beach, Melbourne) and near major freshwater inputs (St. Lucie River, Sebastian River) have disappeared or are severely stressed (heavy epiphytization, low short shoot density). The most developed seagrass beds have been found in the northern lagoon where urbanization and freshwater inputs are minimal. North of Sebastian Inlet only *H. wrightii*, *S. filiforme*, *Halophila engelmannii* and *Ruppia maritima* occur while south of this inlet are found all seven species.

Based on distribution and biomass, *Halodule wrightii* is the most abundant species in the IRL although percent cover of *Syringodium filiforme* may be higher locally (Thompson, 1978; Virnstein and Carbonara, 1985; Gilbert and Clark, 1981). Virnstein and Cairns (1986) gave the following ranking of seagrass species in order of decreasing percent cover: *S. filiforme*, *H. wrightii*, *Halophila johnsonii*, *Thalassia testudinum*, *Halophila decipiens*, *Halophila engelmannii*, and *Ruppia maritima*. Kenworthy (1992a) found *H. wrightii* to be the most abundant species in Hope and Jupiter Sounds. Vertical zonation occurs with shoal grass frequently found in the most shallow (<0.5 m) and deeper regions (>1 m). Manatee grass occupies both intermediate and greater depths with *H. wrightii* while *T. testudinum* is found in depths less than 2 m (Kenworthy, 1992a).

The largest species of *Halophila*, *H. engelmannii*, occurs throughout the lagoon growing sparsely in the canopy beneath *Halodule wrightii* and *Syringodium filiforme* (Eiseman, 1974), and in monotypic stands in deeper water in the IRL (Gilbert and Clark, 1981; Short and Cambridge, 1984; Kenworthy, 1992a, 1992b). The distribution of *Halophila decipiens* in the IRL is restricted to locations south of Sebastian Inlet where it occurs primarily in monotypic stands in deeper water, or shallow turbid water. Unlike the other *Halophila* species, *H. decipiens* is rarely found under the canopy of larger seagrasses. *Halophila decipiens* regenerates annually by seed, growing during months of April to October (Continental Shelf Associates Inc. and Martel Laboratories, Inc., 1985; Kenworthy, 1992a, 1992b). Because the seeds of this species are light dependent for germination (McMillan, 1988), bioturbation may play a role in bringing seeds to the sediment surface, explaining the patchy distribution of *H. decipiens* in deeper water (Kenworthy, 1992a).

The geographical distribution of *Halophila johnsonii* is restricted to the east coast of Florida, known only between Sebastian Inlet and Biscayne Bay (Eiseman and McMillan, 1980; Kenworthy, 1992b). In the IRL, the species occurs between Sebastian and Jupiter Inlets. Unlike the other two species, *H. johnsonii* can be found in high irradiance intertidal zones, under the canopy of the larger seagrasses, and in deep water in association with *H. decipiens* (Kenworthy, 1992b). In laboratory studies, *H. johnsonii* was found to have broad tolerances to wide ranges of irradiance, temperature, and salinity (Dawes et al., 1989). Thus, although having a limited distribution, *H. johnsonii* shows broad tolerances and may be a recently evolved species. Because it is rare, the National Marine Fisheries Service has proposed *H. johnsonii* as a threatened or endangered plant in the Federal Register (1993).

*Seagrass Seasonality and Productivity.*—The seasonal growth pattern of seagrasses in the IRL has been studied only at a few sites, mostly emphasizing the three larger species (turtle, shoal, and manatee grass). Growth is continuous throughout the year with a bimodal cycle reported in all studies. Maximum growth is in the spring (April–May) and a lesser peak occurs in the fall (Rice et al., 1983; Virnstein and Carbonara, 1985). Maximum above-ground biomass was in June–July with highs of 300, 160, and 60 g dwt·m<sup>-2</sup> for *Thalassia testudinum*, *H. wrightii*, and *S. filiforme* respectively (Table 1), while minimum biomass was recorded in January–February in a bed near Link Port (Virnstein and Carbonara, 1985). Similar seasonal biomass changes were reported for *Halodule wrightii* near Link Port (French and Montgomery, 1983) with an August maximum of 245 and a September minimum of 140 g dwt·m<sup>-2</sup>. Latitudinal differences in growth have been noted for *Halodule wrightii* (Eiseman and Benz, 1975) and *Syringodium filiforme* (Gilbert and Clark, 1981); both species showed higher growth rates in the southern areas of the IRL.

Although the data are limited, seagrass productivity in the IRL, appears to be high when compared with Tampa Bay (Table 1). Rice et al. (1983) reported that net productivity was highest in spring and fall with rates of 1,185, 1,090, and 670 µg C fixed mg·chl<sup>-1</sup>·h<sup>-1</sup> for *Syringodium filiforme*, *Halodule wrightii*, and *Halophila engelmannii*, respectively.

In situ growth of seagrass leaves ranged from 0.86 to 3.11 cm·d<sup>-1</sup> for manatee grass (Fry, 1983) and up to 0.85 cm·d<sup>-1</sup> for shoal grass (Virnstein, 1982). Leaf growth during the spring and summer in the two species were 1.8 g dwt·m<sup>-2</sup>·d<sup>-1</sup> for *S. filiforme* (Fry and Virnstein, 1988) and 3.0 g dwt·m<sup>-2</sup>·d<sup>-1</sup> for *H. wrightii* (Virnstein, 1982). *Syringodium filiforme* showed high growth rates in the lagoon,

Table 1. Comparisons of biomass (g dwt m<sup>-2</sup>) and productivity (see footnote for units) of the three larger seagrasses common to the coasts of Florida (productivity ranges are shown for March and July, if available)

	Biomass			Productivity
	Total	Above	Below	
<i>Thalassia testudinum</i>				
Tampa Bay	49 <sup>1</sup>	25–180 <sup>2</sup>	600–900 <sup>2</sup>	1.0 <sup>3</sup>
Indian River				
Link Port <sup>3</sup>	98–160	1.3–128.1		
Link Port <sup>4</sup>	100–300			
Jim Island <sup>3</sup>	64–124	0.1–30.3		
Vero Beach <sup>3</sup>	85	8.4		
<i>Syringodium filiforme</i>				
Tampa Bay	22 <sup>1</sup>	50–170 <sup>2</sup>	160–400 <sup>2</sup>	0.5 <sup>3</sup>
Indian River				
Grant <sup>7</sup>			1.2	
Link Port <sup>3</sup>	49–58		0.1–2.7	
Link Port <sup>4</sup>	<10–40			
Jim Island <sup>3</sup>	12–19		1.8–20.4	
<i>Halodule wrightii</i>				
Tampa Bay	19 <sup>1</sup>	38–50 <sup>2</sup>	60–140 <sup>2</sup>	0.5 <sup>3</sup>
Indian River				
Grant <sup>7</sup>				1.1
Link Port <sup>3</sup>	20–72			0.2–2.6
Link Port <sup>5</sup>	20–140			
Jim Island <sup>3</sup>	124–198			1.2–47.0
Vero Beach <sup>3</sup>	45			13.2

<sup>1</sup> Jensen and Gibson, 1986. One year study in Tampa Bay (mg C m<sup>-2</sup> h<sup>-1</sup>).

<sup>2</sup> Lewis and Phillips, 1980. Aerial mapping of Tampa Bay (mg C m<sup>-2</sup> h<sup>-1</sup>).

<sup>3</sup> Heffernan and Gibson, 1983. One year study of IRL sites (mg C m<sup>-2</sup> h<sup>-1</sup>).

<sup>4</sup> Virnstein and Carbonara, 1985. 15 ha area monitored for 1 year (mg C g dwt<sup>-1</sup> h<sup>-1</sup>).

<sup>5</sup> Zieman and Zieman, 1989. Review (mg C g dwt<sup>-1</sup> h<sup>-1</sup>).

<sup>7</sup> Rice et al., 1983. Spring and fall measurements (mg C mg Chl a<sup>-1</sup> h<sup>-1</sup>).

with an annual leaf production of 572 g dwt·m<sup>-2</sup>·y<sup>-1</sup>, and rhizome growth being ¼ and root growth ⅙ of blade production (Short et al., 1993). An average of 47% of the total *S. filiforme* leaf production was exported out of the seagrass community during the summer growth period (Fry and Virnstein, 1988) linking seagrasses to decomposers and secondary producers outside the meadows.

Using <sup>14</sup>C labeled NaHCO<sub>3</sub>, primary productivity of the 3 larger species of seagrasses, epiphytic flora, benthic microalgae, and phytoplankton were compared at Jim Island, Link Port and Vero Beach (Heffernan and Gibson, 1983). In July, the photosynthetic rates were 2.63 to 47.0 for *Halodule wrightii*, 8.4 to 128.1 for *Thalassia testudinum*, and 2.7 to 20.4 mg C·m<sup>-2</sup>·h<sup>-1</sup> for *Syringodium filiforme* (Table 1). Relative to the other autotrophic components in the IRL (phytoplankton, macroalgae, benthic microalgae), there are large spatial and temporal variations in the contribution of seagrasses. Estimates of seagrass productivity range from 7% (Jensen and Gibson, 1986) to 86% (calculated from Heffernan and Gibson, 1983) of the fixed carbon. More studies are needed using standard procedures to clarify seagrass productivity.

Because of their small size, biomass of the three *Halophila* species is an order of magnitude less than any of the larger seagrasses. Total biomass including leaves, roots and rhizomes, for *H. decipiens* and *H. johnsonii* ranges between 3 and 20 g dwt·m<sup>-2</sup>, whereas the larger seagrasses may exceed 200–300 g dwt·m<sup>-2</sup>

(Gilbert and Clark, 1981; Kenworthy et al., 1989). Despite low biomass, the two *Halophila* species exhibit rapid growth rates, high fecundity and decompose more rapidly than *Halodule wrightii*, *Syringodium filiforme* or *Thalassia testudinum* (Kenworthy et al., 1989).

In water depths of 1.5–3 m, *Halophila decipiens* may cover up to 26% of the bottom (Kenworthy, 1992a, 1992b). When considered in the overall abundance of seagrasses in the IRL, which, under the best conditions is approximately 20–30% of the bottom, *Halophila* spp. must increase seagrass cover by a factor of 1.6–2.0. As with the other seagrasses, *Halophila* spp. stabilize sediments (Fonseca, 1989), provide habitat, and produce organic matter for secondary consumers and detritivores thereby increasing the potential for faunal diversity (Kenworthy et al., 1989).

*Management of the Seagrasses in the IRL.*—Seagrass monitoring can be used to monitor the health of an estuary (Dennison et al., 1993; Tomasko et al., in prep.) especially in light of their importance to the diversity of the IRL (Steward et al., 1992). The communities serve as habitats for invertebrates (Virnstein, 1987), macrocrustaceans (Gore et al., 1981), fish (Stoner, 1983), and manatees (Lefebvre and Powell, 1990). In addition, the associated drift macroalgae (Eiseman and Benz, 1975; Fry, 1984) and seagrass blade epiphytes (Howard and Short, 1986; Virnstein and Howard, 1987) serve as food and habitat within these communities.

Of the seven general threats to reduction of biodiversity worldwide (Suchanek, 1994), habitat loss and degradation has caused the greatest amount of damage to seagrass communities in the IRL. The decline in coverage of the IRL seagrass communities has been documented since the 1950's, with estimated losses as high as 100% in certain areas (Haddad, 1985). These declines have been correlated with decreases in water transparency and a general decline in water quality (Morris and Tomasko, 1993). Diminished water transparency has resulted from increased suspended particulate loads, growth of plankton and seagrass epiphytes due to nutrient loading, and from the modification of freshwater discharges that release colored water from drainage canals (Kenworthy and Haurert, 1991; Morris and Tomasko, 1993; Tomasko et al., in prep.).

Decline of the IRL seagrasses probably has resulted in further decreases in water transparency because they stabilize the sediments and recycle nutrients while supporting primary and secondary production. Once seagrass beds are lost it is difficult and expensive to restore them. In fact, there has never been a seagrass restoration effort that has resulted in the recovery of 100% of the lost resources (Fonseca, 1992). For these reasons, it is imperative that every step be taken to conserve and protect the existing beds and to establish biologically and economically feasible enhancement and restoration goals in the IRL.

Given the continued pressures of urbanization of the surrounding land, maintaining the high biodiversity of the seagrass communities is a challenge. Effective management of the IRL requires an understanding of how ecological processes control seagrass biodiversity, how seagrass biodiversity affects the rest of the IRL system, and how species and habitat diversity are affected by both human activities and natural phenomena. The selection of seagrass communities with high priority for protection and enhancement should include the following considerations: 1. Ecological function. An important consideration in seagrass conservation should be maintenance and enhancement of seagrass function. Information is needed on what is the minimal size of seagrass beds to support specific flora and fauna. Studies are needed to determine the critical size and importance of different types of seagrass communities. Recent studies suggest specific associations exist;

for example *Halodule wrightii* is the preferred habitat for juvenile spotted seatrout (*Cynoscion nebulosus*) in the IRL (Gilmore, pers. comm.). No information is available regarding specific associations for species of *Halophila*. 2. Faunal diversity. Seagrass bed complexity probably enhances faunal biodiversity. Habitat edges or transition zones are generally richer in species than their adjacent habitats and thus should be included in preservation and enhancement plans. 3. Anthropogenic impacts. Selection of any region of the IRL for protection of seagrass communities must consider continued anthropogenic impacts. Modifications to natural habitats (storm water run off, drainage canals, dredging, seawalls) degrade the value of adjacent seagrass communities and may exceed the natural resiliency of the system. Seagrass management must therefore go beyond the permitting process (for dredging, dock construction) and include control of boat traffic (propeller scaring), and monitoring water quality from upland areas. Seagrass enhancement projects must be correlated with water quality monitoring to identify regions where management is ecologically and economically feasible. Recent optical water quality models (Gallegos, 1994) may be useful tools for achieving this goal.

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