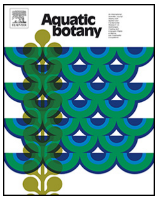
Aquatic Botany 124 (2015) 45–53

Contents lists available at ScienceDirect

Aquatic Botany

journal homepage: www.elsevier.com/locate/aquabot



Relative effects of physical and small-scale nutrient factors on the distribution of tropical seagrasses in the Great White Heron National Wildlife Refuge, Lower Florida Keys

Lauri Green[a,*], Dale E. Gawlik[b,1], Leonardo Calle[b,2], Brian E. Lapointe[a,3]

[a] *Harbor Branch Oceanographic Institute at Florida Atlantic University, 5600 US Hwy 1 Fort Pierce, FL 34982, USA*
[b] *Environmental Science Program, Florida Atlantic University, 777 Glades Rd. Boca Raton, FL 33431, USA*

ARTICLE INFO

Article history:
Received 14 October 2014
Received in revised form 18 March 2015
Accepted 19 March 2015
Available online 26 March 2015

Keywords:
Florida
Habitat suitability
Model selection
Nutrient availability
Physical
Seagrass

ABSTRACT

We tested the relative effects of physical factors such as exposure time and water depth as well as nutrient availability on *Thalassia testudinum*, *Halodule wrightii* and *Syringodium filiforme* distribution within the Great White Heron National Wildlife Refuge, Florida Keys. We quantified the percent cover of each seagrass species in 1-m^2 plots ($n = 325$) along intertidal and shallow subtidal flats adjacent to Upper Harbor Key Water Keys and Howe Key. We used model selection to evaluate the effects of physical parameters and water column nutrients on the percent cover and composition of seagrass species within plots. Best models were selected based on lowest Akaike's information criteria (AIC) values and maximum model weights (ω_i). We found that the presence of the other species, distance to nearest island and time of exposure during diurnal low tides best explained the distribution of *T. testudinum* ($\omega_i = 0.44$). Model averaged parameter estimates (β) showed that *H. wrightii* and *S. filiforme* had the greatest negative influence on *T. testudinum* ($\beta = -0.396$, -0.278, respectively). *H. wrightii* distribution was affected strongly by the presence of the other species, distance to Pine Channel, exposure time and mean lower low water (MLLW) ($\omega_i = 0.56$) with *T. testudinum* and *S. filiforme* exerting the greatest negative influences ($\beta = -0.450$, -0.184, respectively). The best model indicated that *S. filiforme* was strongly influenced by the other species, distance to Pine Channel and MLLW ($\omega_i = 0.5$). Model averaging indicated that *S. filiforme* was associated with deep water ($\beta_{MLLW} = -28.0.018$). Our study showcased that small scale (<100 m) habitat heterogeneity influenced the composition of seagrass communities.

1. Introduction

Seagrass ecosystems are critically important because many species of threatened fish, reptiles and birds depend on them during some stage of their life cycle (Hughes et al., 2009). Understanding the factors that alter the abundance and distribution of species is an important focus of ecology particularly for foundation species such as seagrasses. For example, light as a function of water depth is an established factor that determines habitat suitability for seagrasses (Robbins and Bell, 2000; Hale et al., 2004). Whereas, the risks associated with desiccation during low tide (Leuschner and Rees, 1993; Vermaat et al., 1993; Seddon and Cheshire, 2001) in structuring seagrass communities (Lan et al., 2005; Campbell et al., 2006) is less well understood. Fonseca and Bell (1998) showed that total seagrass percent cover tended to decrease while the ratio between *Halodule wrightii* and *Zostera marina* tended to increase with increased relative exposure index (REI) values. On the other hand, it is less clear how factors such as current speed (Fagherazzi and Wiberg, 2009) and erosion (Erftmeijer and Lewis III, 2006) may reduce habitat suitability for some seagrass species but not others. Understanding the relative influence of physical forces that act on seagrass community composition will provide resource managers with ecological insight that can be applied to maximize the success of seagrass restoration efforts and to predict the response of seagrasses by natural and anthropogenic changes in the physical environment over time.

---

[*] Corresponding author. Current address: US Environmental Protection Agency, OSU Hatfield Marine Science Center, 2111 SE Marine Science Dr., Newport, OR 97366, USA. Tel.: +1 541 867 4035; fax: +1 541 867 0100.

E-mail addresses: ruiz-green.lauri@epa.gov (L. Green), dgawlik@fau.edu (D.E. Gawlik), leonardo.calle@msu.montana.edu (L. Calle), blapoin1@fau.edu (B.E. Lapointe).

[1] Tel.: +1 561 297 3333.

[2] Ecosystem Dynamics Laboratory, Department of Ecology, Montana State University Bozeman, MT 59717, USA.

[3] Tel.: +1 772 242 2276.

http://dx.doi.org/10.1016/j.aquabot.2015.03.005
0304-3770/

Degradation of seagrass beds is occurring worldwide (Lotze et al., 2006) with devastating impairment to ecosystem function (Coll et al., 2011) and reduction in coastal protection from waves (Koch et al., 2009). Nutrient enrichment is among the top threats to the ecosystem function of seagrass beds (McGlathery et al., 2013). Excessive input from sources such as fertilizers (Howarth et al., 2002) and sewage (Ward-Paige et al., 2005) stimulate phytoplankton blooms (Ferreira et al., 2011), macroalgae (Hauxwell et al., 2001; Kopecky and Dunton, 2006), and periphyton (Tomasko and Lapointe, 1991; Lapointe et al., 1994). These factors block light and reduce seagrass productivity (Nelson, 2009). Nutrient enrichment has been known to decrease habitat suitability for a theoretical climax community dominated by *Thalassia testudinum* (e.g., Thayer et al., 1994) to one dominated by *H. wrightii* (Fourqurean et al., 2005). Such changes also affect seagrass-dependent species at higher trophic levels. For example, Ferguson (2008, 2009),) demonstrated that nutrient-driven changes in seagrass structural composition altered benthic invertebrate community composition and resulted in loss of larger bodied invertebrates. Studies that evaluate the role of nutrient enrichment and the extent of changes in seagrass community composition have wide application to the conservation of marine fauna and are critically needed for marine sanctuaries.

We applied Akaike's information criteria (AIC) to generalized linear models to evaluate the importance of physical factors and nutrient availability and on three species of seagrass: *T. testudinum*, *H. wrightii* and *Syringodium filiforme* within the Great White Heron National Wildlife Refuge, Florida Keys. We hypothesized that (i) seagrass plots closest to elevated concentrations of water column of nitrogen and phosphorus would be dominated by *H. wrightii* and *S. filiforme* (ii) seagrasses plots farther from elevated nutrient concentrations would be dominated by *T. testudinum* and affected most strongly by physical factors.

2. Methods

2.1. Study area

This study was conducted in intertidal and shallow subtidal (<3 m) seagrass flats adjacent to Upper Harbor Key (24°48′40.19″N, 81°26′26.67″W), Water Keys (24°46′40.28″N, 81°27′26.91″W) and Howe Key (24°43′48.22″N, 81°25′51.37″W) within the Great White Heron National Wildlife Refuge (hereafter referred to as: the refuge) and the Florida Keys National Marine Sanctuary (FKNMS), Florida, U. S. A. (Fig. 1). Data on water column nutrients, chlorophyll *a* concentrations, seagrass percent cover and periphyton percent cover were collected in April and May 2012 (spring), July 2012 (summer), October 2012 (fall), and February 2013 (winter). Additional seagrass and periphyton data were collected in April–May 2013 (spring). Assessments of sediment organic content were performed in October 2012 (fall) and February 2013 (winter). Furthermore, previous research suggested possible anthropogenic nitrogen and phosphorus concentration gradients from Howe Key to Upper Harbor Key (local sewage discharges) or from the Everglades (agricultural runoff) (Lapointe et al., 2004).

2.2. Quantification of seagrass abundance and distribution

From July 2012 to April 2013, a total of 325 plots were assessed (Fig. 1) and the percent cover of *T. testudinum* (turtle grass), *H. wrightii* (shoal grass), and *S. filiforme* (manatee grass) were quantified for each plot. In summer 2012, plots were selected by a snorkeler that swam approximately 15 meters from a boat. At each plot, the percent cover of each seagrass species was quantified using a 1 m^2 quadrat, gridded into 100, 10 cm cells. The presence of each species was evaluated for each cell and percent cover was the sum of all cells. The snorkeler then haphazardly chose a direction and swam another ~15 m. This process was repeated five times per site. Additionally in summer 2012, we assessed percent cover on a wider area than could be covered by a swimming snorkeler. Plots were selected by anchoring a boat approximately every 200 m along four parallel transects which began 50–100 m from the mangrove islands. Snorkelers placed a quadrat on the benthos immediately off the starboard side. To map data, the coordinates of each plot were taken from the boat using a handheld Garmin GPSMap® 78 sc centered above the quadrat. Sixteen plots were assessed for Upper Harbor Key and Howe Key only. Due to time constraints, Water Keys was not assessed using a grid; only haphazard plots were collected during that time period. In fall 2012 and winter 2013, 30 GPS points spaced 50 m apart and forming a grid 250 m by 300 m were established using Google™ Earth. A boat was driven as close to the point as possible. Data and coordinates within each plot were collected as above.

To increase our sample size for seagrass and periphyton percent cover and take into account physical factors known to affect seagrass distribution we changed our seagrass plot selection methods in spring 2013. We sought to compare relative water depth across sites, thus we used mean lower low water (MLLW) from a digital elevation map (DEM) created in 1997 by the NOAA as part of the Florida Keys Benthic Habitat Mapping Project. Since updated maps were not available, we assumed that elevation did not change in the interim. Positive values represent points above sea level; whereas, negative values were below sea level. One hundred and twenty one plots along 21 transects were established along MLLW gradients. The origins of each transect were selected randomly from points that fell within 50 m of the channel and ran 400–700 m along depth gradients. Plots were spaced 100 m apart and percent cover of seagrasses and periphyton were quantified as above.

2.3. Quantification of periphyton percent cover

We quantified the percent cover of large periphyton (greater than 2 cm) and small periphyton (less than 2 cm long) by counting the number of cells within each 1 m^2 plot (see above) where large and small periphyton on seagrasses were observed even once within each cell. For 122 of the 325 total plots, percent cover was calculated as the sum of the number of cells for each periphyton class. We distinguished the two categories because large periphyton have been shown to uproot seagrass plants (Borowitzka and Lethbridge, 1989 as cited in Fong et al., 2000). In summer and fall 2012 and winter 2013 a subset of five random seagrass plots were selected and the percent cover of periphyton for those plots were quantified. In spring, 2013 at 102 plots where seagrass was quantified percent cover of large and small periphyton were also quantified.

2.4. Quantification of water column dissolved nutrients and chlorophyll *a*

2.4.1. Water column dissolved nutrients

To obtain a snapshot of nutrient concentrations for each site we quantified water column nutrients concentrations. The study area has four possible nutrient sources; the Gulf of Mexico (Gibson et al., 2008), Everglades discharges (Brand, 2002; Lapointe et al., 2004), sewage and aerobic treatment units near the inhabited Big Pine Key (Lapointe et al., 2004) and bird guano from roosting and nesting islands (Calle et al., 2014). We quantified the instantaneous nutrient concentrations of each site during four sampling seasons but we did not quantify nutrient concentrations with respect to distance to known roosting and nesting sites. Water samples were collected in May 2012 (spring), July 2012 (summer) October 2012 (fall), and