Gulf and Caribbean Research

2013

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Recommended Citation

ASSESSMENT OF SEAGRASS FLORAL COMMUNITY STRUCTURE FROM TWO CARIBBEAN MARINE PROTECTED AREAS

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ABSTRACT: Seagrass communities represent spatially complex and biomass producing systems comprised of intermixed seagrass and algal species. We investigated shallow water communities from two Marine Protected Areas (MPAs) in the Caribbean: St. John, United States Virgin Islands and Cayos Cochinos, Honduras. St. John sites (4) lie within the Virgin Islands National Park and the Coral Reef National Monument and are designated within an UNESCO Biosphere Reserve. Honduran sites (4) lie within the designated Marine National Monument. Our results indicate that both MPAs were dominated by *Thalassia testudinum* with spatial coverage and shoot density significantly greater in Honduras. Many sites also showed substantial cover of *Syringodium filiforme*, which was significantly greater in St. John. Most major algal groups showed significant differences between MPAs and among sites within locations. Specifically, *Halimeda, Penicillus, Udotea, Galaxaura,* and *Dictyosphaeria* were significantly more abundant in Honduras, while *Padina* and *Avrainvillea* were significantly greater from St. John. Additionally, only Honduran sites showed the presence of coral colonies (*Montastrea* and *Porites*) within their seagrass beds. Floral community level analyses demonstrated significant differences among almost all site comparisons suggesting relatively distinct floral communities exist within each of these regions, but both MPAs maintain high spatial coverage of seagrasses providing critical ecosystem services.

KEY WORDS: Cayos Cochinos, Honduras, MPA, *Syringodium, Thalassia*

INTRODUCTION

Global seagrass decline is well documented (Orth et al. 2006, Waycott et al. 2009) and is often a result of anthropogenic activities. While seagrasses are recognized as important primary producers and serve as refuge from predation, our ability to protect and stop the decline is limited for many species (Short et al. 2011). Consistently, coastal eutrophication has been a significant factor in seagrass decline as excess nitrogen fuels phytoplankton and ephemeral algal populations, causing reductions in light and diurnal hypoxia as blooms spread. While this scenario is well documented in temperate systems, the decline in some tropical grass beds is more difficult to link directly to eutrophication. Often tropical seagrass beds have shown declines associated with physical damage (e.g., boat anchors, mooring, prop scars), light attenuation from terrestrial sediment runoff and from the generalized collapse and destruction of the associated mangrove and coral reef communities. Both mangrove and coral reef systems have also seen significant global declines from a variety of direct anthropogenic sources (e.g., coastal development, aquaculture, run–off, eutrophication, algal overgrowth) as well as widespread diseases, overfishing, loss of herbivores, and climate change (Valiela et al. 2001, Pandolfi et al. 2003). Essentially, when any of these communities is negatively impacted, secondary impacts are often felt in the adjacent systems. For example, mangrove deforestation leads to elevated run–off (Thampanya et al. 2006) and resuspension causing sediment loading to seagrass and coral reef communities; thereby increasing light stress and physical smothering of organisms. These secondary ripple effects eventually lead to overall declines in entire coastal ecosystems and a vicious circle of ever increasing declines occur. Rivera–Monroy et al. (2004) describe a framework in which to assess the current status of the Caribbean marine systems and identify potential stressors and areas of research necessary to develop management strategies. Wilkinson and Salvat (2012) argue that this unprecedented decline rises to the moral imperative of addressing social issues driving the anthropogenic stresses negatively impacting habitats and living resources, while Unsworth and Cullen (2010) advocate for the conservation of seagrasses as fundamental to the health of tropical marine communities.

Habitat restoration is often proposed as a mechanism to reverse the trends of habitat loss, but it has met with varying levels of success, and restoration of mangrove and coral reef communities may take decades to become established and provide biological and ecosystem services (Bosire et al. 2008). As such, one potential avenue to stem the local declines in these systems is to enact policies which protect them from further decay and allow the natural resilience of the communities to reestablish and flourish. The most frequent policy employed is the development of a Marine Protected Area (MPA). While this may seem to be a relatively straightforward answer, MPA implementation often means the displacement of local residents and elimination of their livelihood. This is most evident in the implementation of fishing restrictions, which may impact both local economies and basic food gathering for the affected, displaced individuals (Brondo and Bown 2011). As such, development of MPAs frequently takes substantial negotiations to ensure local economic viability or subsistence, but local communities can

often find themselves marginalized with limited input, understanding, and opportunities associated with MPA designation and implementation (Camargo et al. 2009, Ban et al. 2011). However, when appropriately protected, the regional value of the improved ecological functionality of the protected ecosystem can have far reaching positive impacts ranging from fish spawning sanctuaries increasing regional fish populations (Babcock et al. 2010) to ecotourism providing a new economic viability feature to local economies (Brondo and Bown 2011). However, this is not without risk to the living resources, as evidenced by novice snorkelers negatively impacting seagrass beds in a Mexican MPA (Herrera–Silveira et al. 2010).

MPAs are now established in tropical regions worldwide, but the MPA designation is frequently tied to coral reefs (Gaines et al. 2010, Ban et al. 2011). As these communities are the most prone to debilitating anthropogenic threats, especially climate change, they necessarily need the elevated designation and protection. However, it is the combined and integrative nature of coral reefs, mangroves, and seagrass beds which often provide the regional recovery and resilience of fish and invertebrate populations (Aguilar–Perera and Appeldoorn 2008), but resilience may differ between Caribbean and Pacific reef systems based on functional redundancy and species richness (Roff and Mumby 2012).

Since Caribbean communities seem to lack this system redundancy, the potential for negative human impacts to living resources increases as development and encroachment increase. In St. John, USVI, an increase in housing and commercial development has led to increased land erosion (Macdonald et al. 1997) and a dramatic shift in sediment inputs from natural processes to ones dominated by human activity (Brooks et al. 2007). These external pressures, coupled with natural disturbances and diseases, have led to the degradation of marine communities and negatively impacted fish populations (Rogers and Beets 2001) and coral reefs (Rogers and Miller 2006). Similar trends in coral reef degradation were shown for Honduras (Garcia–Salgado et al. 2008) and may be related to increased erosion and industrial activity (Harborne et al. 2001, Prouty et al. 2008), but limited data in either system exist regarding the potential impacts in affiliated seagrass communities.
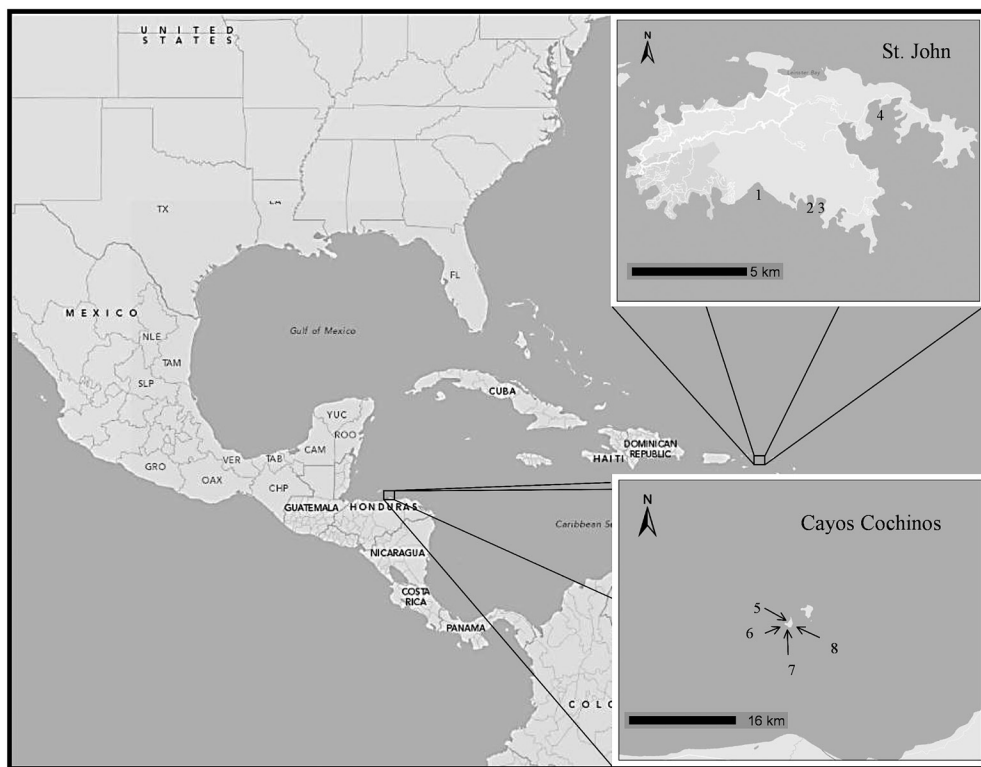

Figure 1. Location of the study sites in St. John, USVI and Cayos Cochinos, Honduras. Site designations are as follows: 1, Reef Bay (REEF); 2, Great Lameshur (GL); 3, Little Lameshur (LL); 4, Hurricane Hole (HH); 5, Jena's Cove (JC); 6, Menor West (MW); 7, Menor South (MS); 8, Menor East (ME).

Tropical seagrass systems represent a complex community composed of a few vascular plant species with numerous algal and marginal coral species. The shared role of primary production and habitat structure provide important resources to fish and invertebrates and are directly linked to elevated fish populations on the adjacent coral reefs (Dorenbosch et al. 2004). Therefore, determining the basic floral community structure is essential to addressing and assessing their role in these integrated communities. However, one of the challenges to determining this structure is the generalized legal nature of MPAs, which restrictively or explicitly prohibit destructive sampling. We present a comparative, non–destructive assessment of seagrass floral communities in Caribbean MPAs in St. Johns. USVI and Hondoras.

Materials and Methods
Study sites

Seagrass beds were sampled in two MPAs in the Caribbean. Data were collected in 2009 from the Cayos Cochinos Marine National Monument in Honduras and in 2010 from the Virgin Islands National Park and Coral Reef Monument in St. John, USVI (Figure 1). Each of these sites has substantial governmental protection and restrictions on fishing and other human activities. St. John is an UNESCO designated Biosphere Reserve, while Cayos Cochinos is a Government protected MPA managed by the Honduran Coral Reef Foundation.

Floral community structure was assessed at 4 shallow water seagrass beds in each MPA. Honduran sites surrounding Cayos Menor include Jena's Cove (15°57'30"N, 86°30'14"W), Menor South (15°57'17"N, 86°30'24"W), Menor West (15°57'28"N, 86°30'26"W), and Menor East (15°57'10"N, 86°29'52"W). St. John, USVI sites included Hurricane Hole (18°21'11"N, 64°42'13"W), Great Lameshur (18°19'07"N, 64°43'23"W), Little Lameshur (18°19'11"N, 64°43'40"W), and Reef Bay (18°19'22"N, 64°44'47"W).

Floral assessment

To conduct non–destructive research, a quadrat methodology was employed to determine seagrass demographics and algal species abundance, richness, and diversity. In Honduras, a 50 cm x 50 cm (0.25 m²) quadrat was used in shallow (<2 m depth), sub–tidal seagrass dominated communities. All algal species were identified to lowest practical taxonomy in the field, with a photographic reference for unidentifiable species during the dive. Algal species were then enumerated in the total quadrat, while seagrass species presence and coverage was assessed visually within the sample quadrat. This technique did not allow for an exact quantification of spatial coverage, but provided a relative dominance assessment among the seagrass species (e.g., monoculture or mixed species beds). A small rectangular sub–grid (8 cm x 9 cm, 0.072m²) was used to determine shoot density for seagrass species. Research in Honduras was supervised by the Operation Wallacea Dive Program. Dive regulations permit a maximum of 50 minutes for dives, so each site was sampled haphazardly for this time period and as many quadrat samples were collected as possible. Specifically, 14 samples were collected from Menor West (MW), 17 from Menor South (MS), 19 from Jena's Cove (JC), and 18 from Menor East (ME). For research conducted in St. John, a collapsible, portable quadrat was manufactured so that all future sampling protocols would be standardized. To assess the floral characteristics we used 0.09 m² quadrats (30 cm x 30 cm) divided into 9 grids of 10 cm x 10 cm. Percent cover of seagrass species was assessed by using a presence–absence ratio of occurrence in the 9 grids. Algal species were identified and counted in the entire quadrat to get a density estimate of each as they comprised part of the community but are not dominant coverage species. The central grid was then investigated and the shoot abundance of each potential seagrass species was counted to calculate a shoot density for each of the quadrats. Total number of samples collected at each site varied in St. John based on logistical considerations. Specifically, 22 samples were collected from Great Lameshur (GL), 25 from Little Lameshur (LL), and 20 from both Reef Bay (REEF) and Hurricane Hole (HH).

Statistical Analysis

While the quadrat sampling methodology differed slightly between Honduras and the St. John sites, the data collected are comparable as coverage and density information were standardized based on the sample quadrat used. Data were standardized for both sites on a per m² basis and analyzed using 2–Way Nested ANOVA (SAS®) with site as the independent variable and locations within sites nested in the analysis. Dependant variables related to the spatial coverage and shoot density of turtle grass (*Thalassia testudinum* Banks ex König), manatee grass (*Syringodium filiforme* Kütz), shoal grass (*Halodule wrightii* Ascherson) and density of the abundant algal groups. To assess significant differences among locations within each MPA, we used an LSMeans procedure for discrimination. In all cases, degrees of freedom are $F_{1,147}$ unless otherwise noted. Additionally, to assess floral community structure, we conducted a Similarity of Percentages (SIMPER), an Analysis of Similarities (ANOSIM), and used the non-metric multi-dimensional scaling (MDS) for discrimination outlined in the software program Primer®. Data were 4th root transformed to minimize the bias associated with the numerically dominant seagrass species and abundant algal taxa present in samples. We conducted 2 different analyses, one directed at assessing the total floral community (seagrass and algal taxa) and a second analysis only assessing the community differences associated with algal taxa. The reason the second analysis was carried out was to discriminate the algal community differences among sites from the vascular plant community which dominated the initial analyses.

Results

Characterization of shallow water floral communities showed that both MPAs were dominated by *T. testudinum* with lower quantities of *S. filiforme* (Table 1). Numerous significant differences were seen in seagrass demographics between systems including significantly greater *T. testudinum* spatial coverage (F = 6.04, p < 0.02) and shoot density ($F_{1,146}$ = 123.2, p < 0.0001; df reduced because one shoot count was smudged on a dive slate) in Honduras, while *S. filiforme* spatial coverage (F = 71.5, p < 0.0001) and *H. wrightii* shoot density (F = 7.4, p < 0.008) were significantly greater in St. John. Differences in the density of several algal taxa were also present with significantly higher means for *Dictyota* spp. (F = 6.6, p < 0.02), *Halimeda* spp. (F = 126.2, p < 0.0001), *Valonia* spp. (F = 4.3, p < 0.04), *Penicillus* spp. (F = 12.1, p < 0.001), *Udotea* spp. (F = 8.2, p < 0.005), *Dictyosphaeria cavernosa* (Forsskål) Børgesen (F = 16.0, p < 0.0001), and *Galaxaura* spp. (F = 8.2, p < 0.005) in Honduras. In contrast, *Padina* spp. (F = 6.7, p < 0.01), *Avrainvillea* spp. (F = 10.8, p < 0.002), and *Wrangelia argus* (Montagne) (F = 4.3, p < 0.04) were significantly greater in St. John. The corals *Porites* spp. (F = 7.5, p < 0.007) and *Montastrea* spp. (F = 4.7, p < 0.03) were significantly greater in Honduras, as they were not found in samples from St. John.

Taxonomic richness did not differ between MPAs (Table 1), suggesting that despite the size differences in the quad-

TABLE 1. *Summary of floral and coral demographics for each of the 8 sites investigated. Data represent mean values (± se) of the percent cover of seagrass species and the density of the floral and coral species (#/m^2). Asterisks following the taxonomic category indicate significant differences between MPAs (* 0.05, ** 0.01, *** 0.001), while letters following the means indicate significant differences among sites within the MPAs (alpha = 0.05). Site designations are as follows: Reef Bay (REEF); Great Lameshur (GL); Little Lameshur (LL); Hurricane Hole (HH); Jena's Cove (JC); Menor West (MW); Menor South (MS); Menor East (ME).*

| MPA Region | Honduras | | | | St. John | | | |
Site	MW	MS	JC	ME	HH	GL	LL	REEF
Seagrass taxa								
Thalassia testudinum percent cover*	95.7±2.3	81.8±6.1	86.8±5.1	99.4±0.56	99.4±0.6^a	68.2±8.8^b	60.4±9.5^b	97.2±1.6^a
T. testudinum shoot density***	873.0±53.3^a	653.6±70.2^b	577.5±45.4^b	678.1±35.5^b	515.0±50.4^a	231.8±45.3^c	184.0±38.6^c	385.0±27.4^b
Syringodium filiforme percent cover***	84.3±4.3^a	55.9±10.8^b	0^c	0^c	95.6±3.9^a	49.5±9.7^b	99.1±0.9^a	35.0±10.1^b
S. filiforme shoot density	763.9±108.7^a	392.2±97.6^b	0^c	0^c	315.0±40.6^b	209.1±72.6^b	616.0±47.8^a	160.0±52.0^b
Halodule wrightii shoot density**	0	0	0	0	0^b	6.8±4.8^b	21.6±7.8^a	0^b
Algal taxa								
Halimeda***	228.2±88.1^c	277.8±70.4^{bc}	394.7±77.5^b	709.9±89.0^a	0.5±0.5	4.1±2.6	19.2±5.1	15.0±5.4
Penicillus***	248.0±73.1^b	776.1±317.8^a	87.7±41.4^b	30.8±14.0^b	98.0±17.4	7.7±4.2	6.4±3.0	0
Dictyosphaeria cavernosa***	9.9±9.9^b	392.2±189.5^a	307.0±91.1^a	15.4±10.6^b	1.0±0.7^b	0^b	0^b	5.0±2.5^a
Dictyota*	0^b	57.2±43.0^{a,b}	102.3±46.1^a	23.1±12.6^b	0	11.4±4.4	10.4±3.2	17.5±6.4
Galaxaura**	0^c	57.2±39.6^b	73.1±37.5^a	7.7±7.7^{b,c}	0	0	0	0
Caulerpa	0^b	81.7±57.2^a	14.6±14.6^b	0^b	3.0±2.1	0	0	2.0±1.4
Udotea**	29.8±15.8^{a,b}	40.8±25.9^a	0^b	30.9±23.9^{a,b}	0	0.5±0.5	0	0
Valonia*	9.9±9.9	40.8±40.8	43.8±30.1	0	0	0	0	0
Avrainvillea**	0	0	0	0	2.5±1.4^b	2.3±1.1^b	6.4±2.4^a	1.5±1.5^b
Padina**	0	0	0	0	0^b	2.3±1.6^b	7.6±3.1^a	0.5±0.5^b
Wrangelia argus*	0	0	0	0	0^b	7.3±3.7^a	0^b	5.5±4.1^{a,b}
Coral								
Porites**	19.8±19.8^{a,b}	73.5±58.4^a	65.8±26.8^a	0^b	0	0	0	0
Montastrea*	0	16.3±16.3	7.3±7.3	15.4±10.6	0	0	0	0
Taxa Richness	3.5^a	3.9^a	3.3^a	2.5^b	3.35^{a,b}	2.4^c	3.64^a	2.7^{b,c}
Floral Diversity***	1.05^a	1.11^a	1.02^a	0.77^b	0.84^a	0.41^c	0.68^{a,b}	0.59^{b,c}

rats used it had no impact on species richness. However, significant differences were observed among sites within MPAs ($F_{6,147}$ = 5.9, p < 0.0001). Taxonomic diversity was significantly greater from Honduras ($F_{1,147}$ = 47.9, p < 0.0001), but also differed among sites within MPAs (F = 5.5, p < 0.0001). Within MPAs, significant differences among sites were seen for *T. testudinum* shoot density, *S. filiforme* shoot density and spatial coverage, and density of *Dictyota*, *Halimeda*, *Caulerpa*, *Penicillus*, *Udotea*, *Galaxaura*, and *Porites* in Honduras. *Thalassia testudinum* shoot density and spatial coverage, *S. filiforme* shoot density and spatial coverage, *H. wrightii* shoot density, *Padina*, *Dictyosphaeria cavernosa*, *Avrainvillea*, and *Wrangelia argus* density differed among sites in St. John (Table 1).

When floral communities were assessed using ANOSIM (Global R = 0.387, p < 0.001), significant differences existed for all site combinations except Honduras MW and MS (Table 2, upper right). Since all sites were numerically dominated by two seagrass species, discrimination of site differences associated with the algal species was difficult to interpret and the SIMPER analysis documented the dominance of *T. testudinum*, which contributed > 50% to the site similarity (Table 3). A second ANOSIM addressed this by assessing only the algal species present (Global R = 0.373, p < 0.001) and significant differences were seen among almost all sites (Table 2, lower left). When the floral communities were plotted in the MDS, distinct differences can be visualized between Honduras and St. John (Figure 2), but substantial overlap is present due to the dominance of *T. testudinum* with discrimination due to the relative amount of *Halimeda*, *S. filiforme*, and *Penicillus* (Table 3).

TABLE 2. *Results of the ANOSIM comparing floral community structure among all locations investigated. Top right section of the table represents the combined seagrass and algal floral community results (R statistic, p value: * = 0.05, ** = 0.01, *** = 0.001), while the lower left section of the table shows the results of the algal only taxa community assessment. Site designations are as follows: Reef Bay (REEF); Great Lameshur (GL); Little Lameshur (LL); Hurricane Hole (HH); Jena's Cove (JC); Menor West (MW); Menor South (MS); Menor East (ME).*

	St. John GL	St. John LL	St. John REEF	St. John HH	Honduras MW	Honduras ME	Honduras MS	Honduras JC
St. John GL		0.25***	0.07*	0.26***	0.15*	0.35***	0.20***	0.37***
St. John LL	0.06		0.37***	0.40***	0.25***	0.80***	0.43***	0.80***
St. John REEF	0.1*	0.1		0.51***	0.41***	0.44***	0.37***	0.40***
St. John HH	0.48***	0.54***	0.86***		0.44***	0.87***	0.47***	0.84***
Honduras MW	0.22**	0.29**	0.55***	0.58***		0.70***	0.06	0.66***
Honduras ME	0.44***	0.34***	0.45***	0.80***	0.36**		0.29***	0.09*
Honduras MS	0.31***	0.26***	0.37***	0.64***	0.04	0.18***		0.25***
Honduras JC	0.42***	0.30***	0.30***	0.84***	0.26**	0.09*	0.08*	

Discussion

Global seagrass loss necessarily means that there is a loss in the productivity and ecosystem services which they provide. One of the greatest challenges we face is that while seagrass systems are well studied in many tropical and temperate regions, just as many, if not more, exist in locations where limited data exist. Our findings build upon initial work carried out by Bologna et al. (2008) in St. John and Ogden (1998) and Michot et al. (2002) from Honduras. While Bologna et al. (2008) focused on seagrass community structure, Ogden (1998) focused on the algal communities present on shallow reef sites during the designation of the Cayos Cochinos MPA and Michot et al. (2002) assessed the response and impacts of Hurricane Mitch. Results from our current work demonstrate high spatial coverage of seagrasses in both systems, but substantial differences between MPAs and among sites within the MPAs in regards to both seagrass and algal communities. In particular, *T. testudinum* shoot density and spatial coverage were significantly greater from Honduras compared to St. John, while *S. filiforme* and *H. wrightii* were greater from St. John (Table 1). Additionally, Honduran sites had greater species richness, diversity, and seven algal species with significantly greater densities, while St. John had only four species showing significantly greater densities. As a consequence, the overall floral community structure between sites was significantly different, but the initial SIMPER analysis demonstrated that both Honduras and

TABLE 3. *Contributing taxa defining and discriminating the MPAs floral composition based upon SIMPER analysis. Values represent the individual percent contribution to defining the flora responsible for the relationship. Top panel results are based on combined seagrass and algal taxa present in samples, while the lower panel analyses are based only upon the algal taxa present in samples.*

All Flora	St. John		Honduras	
St. John	Average St. John Similarity	49.2%	Average Dissimilarity	58.3%
		Individual Contribution		Individual Contribution
	Thalassia testudinum	51.9%	*Halimeda*	22.6%
	Syringodium filiforme	34.7%	*Syringodium filiforme*	19.3%
	Penicillus	4.9%	*Thalassia testudinum*	13.0%
			Penicillus	12.9%
			Dictyosphaeria cavernosa	8.1%
			Dictyota	7.6%
			Udotea	2.6%
			Galaxaura	2.5%
			Avrainvillea	2.4%
Honduras			Average Honduras Similarity	55.9%
				Individual Contribution
			Thalassia testudinum	54.5%
			Halimeda	28.9%
			Penicillus	6.4%
			Syringodium filiforme	5.9%

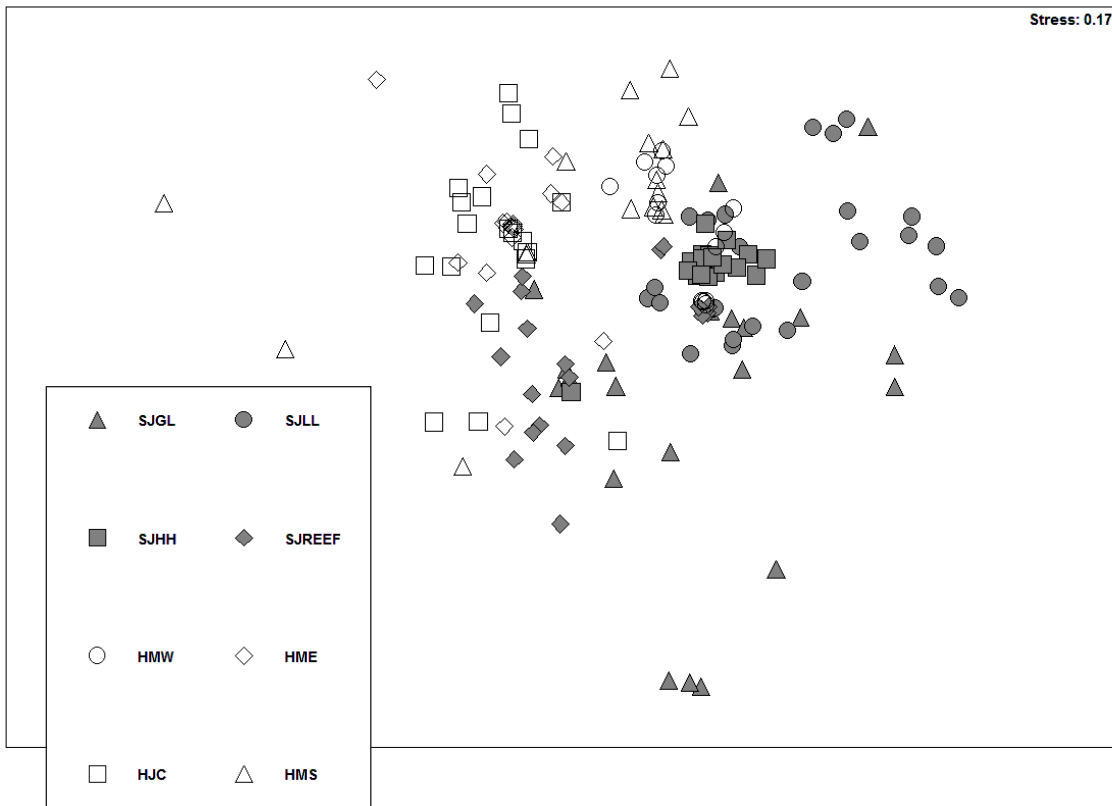| Algae Only				
St. John	Average St. John Similarity	24.3%		
		Individual Contribution		
	Penicillus	44.9%		
	Dictyota	22.1%		
	Halimeda	19.2%		
	Avrainvillea	7.0%		
Honduras	Average Dissimilarity	79.85%	Average Honduras Similarity	43.98%
		Individual Contribution		Individual Contribution
	Halimeda	34.5%	*Halimeda*	74.0%
	Penicillus	20.5%	*Penicillus*	17.3%
	Dictyota	11.5%		
	Dictyosphaeria cavernosa	10.6%		
	Avrainvillea	4.3%		
	Udotea	4.1%		
	Galaxaura	3.2%		
	Padina	3.1%		

Figure 2. *MDS plot of floral community structure between Honduras and St. John. Open symbols represent samples from Honduras, filled symbols represent samples from St. John. Specific sites designated with prefixes of H for Honduras and SJ for St. John in front of location abbreviations. Site designations are as follows: Reef Bay (REEF); Great Lameshur (GL); Little Lameshur (LL); Hurricane Hole (HH); Jena's Cove (JC); Menor West (MW); Menor South (MS); Menor East (ME).*

St. John communities were dominated by *T. testudinum*. In general, our results concur with the previous studies demonstrating dominance by *T. testudinum* with lesser amounts of *S. filiforme*, while *Halimeda* and *Penicillus* dominate the algal groups. Michot et al. (2002) documented *H. wrightii* growth in shallow disturbed regions of Honduras, but it was absent from our current assessment of seagrass community structure. When the seagrass species were eliminated from the analyses, the pattern was dominated by *Penicillus* in St. John and *Halimeda* in Honduras. Consequently, while these systems showed remarkable similarities in species presence, they maintained unique characteristics with an average dissimilarity of 79.8%. Beyond algal differences, the presence of small coral colonies present in Honduras suggests substantially different shallow benthic community structure.

The *Porites* and *Montastrea* colonies present in Honduras suggest healthier reef structure, which corresponds to the differences in MPAs. St. John is predominately a United States National Park with designated protected regions associated with the coastal marine systems, but many residents and visitors inhabit the island. This leads to substantially greater visitation and infrastructure. In fact, the development on St. John has lead to considerable erosion (Brooks et al. 2007), potentially impacting both reef and seagrass communities. Additionally, much of the island is not on a public sewage system, which creates the potential for nutrient loading in the system, and large desalinization plants exist associated with the tourist developments to serve the high water de-

mands. The Honduras MPA has extremely limited development and infrastructure with only limited access to tourists. As such, the anthropogenic pressures are considerably less on the adjacent marine communities, but coastal development in Honduras is progressing (Harborne et al. 2001) and stressors (e.g., sediments, heavy metals) from terrestrial and industrial sources will continue (Prouty et al. 2008). Supporting evidence for greater reef degradation on St. John comes from Rogers and Miller (2006) who demonstrated low coral spatial coverage (10–12%) on reefs in that region compared to the relatively higher coverage seen in Honduras (10–29%, García–Salgado et al. 2008, Bologna et al. 2012). Rogers and Miller (2006) suggest that the coral decline was most likely due to natural and human induced impacts such as hurricanes, over–fishing, and continued development. While these stressors impact Honduran reefs, the reduced anthropogenic stress may play a substantial role in the resilience of the Cayos Cochinos reefs (Carilli et al. 2009), and therefore provide greater recruitment potential for corals or greater environmental stability for recruitment, survival, and growth of *Porites* and *Montastrea*. However, Green et al. (2008) suggest elevated *Porites astreoides* (Lamarck) abundance may be due to the declining reefs creating open space for this species. This argument is not plausible for *Porites* found among the dense grass beds and the greater health of the Honduran reefs provides a proximal mechanism for greater potential recruitment and consequent presence in these grass beds.

The differences observed between MPAs is not unex-

pected, however the differences in flora communities seen among sites within each MPA was. For St. John, 3 of the 4 sites are relatively exposed, but HH is quite protected and had significantly greater *T. testudinum* shoot density compared to the other sites, as well as an order of magnitude greater abundance of *Penicillus* spp., but little *Halimeda* and no *Dictyota* or *Padina*. Site LL was distinguished by its low abundance of *T. testudinum* and significantly greater *S. filiforme*, *H. wrightii*, *Padina*, and *Avrainvillea*. The last 2 sites are principally distinguished by their relative abundances of *T. testudinum* and *S. filiforme*. Why these major differences exist among sites might be explained to a limited degree by relative oceanic exposure, but our data from adjacent bays showing such defined flora communities needs further investigation. Bologna et al. (2008) showed differences in flora communities and organic carbon among GL, LL, and HH, but REEF was not sampled, nor was sediment size structure and composition analyzed. In Honduras, some of the within–MPA differences were seen in the relative cover and shoot density of *T. testudinum*, but a major distinguishing factor was that only half the sites had *S. filiforme* present. Sites with *S. filiforme* also had high densities of *Penicillus*, but lower densities of *Halimeda*. Collectively, these differences are evident at the community level (Figure 2), but additional research is needed to address community structure and the relatively low species richness observed in both MPAs.

One broad challenge in assessing communities within MPAs relates to the ability to collect quantitatively defensible data. The use of quadrats provides essential information regarding the plant density, species richness, and floral diversity. However, biomass, shoot structure, presence of rare algal taxa and other quantitative data are only obtainable through destructive methods, frequently prohibited in many MPAs. This work provides essential information necessary to describe a baseline of seagrass community structure, while we develop long–term strategies for the monitoring of these habitats from these and other MPAs. This type of assessment is currently being completed through Seagrass.net, but different protocols may be needed to cover greater areas to develop robust region–wide data sets from areas which do not have established long–term monitoring plans in place or do not have the organized local communities trained to carry out consistent and long–term monitoring. We recognize the limitations of non–destructive sampling, but affirm the efficacy of using this method to gather quantifiable and comparable data sets among numerous sites.

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