INTRODUCTION

Culebra Island coral reef epibenthic and fish communities have been thoroughly documented (Rogers et al., 1978; Hernández-Delgado, 1994b,c, 2000, 2001, in review; Pagán-Villegas et al., 1999; Garrison et al., 2000; Hernández-Delgado and Sabat, 2000; in preparation; Hernández-Delgado et al., 2000; Hernández-Delgado and Rosado-Matías, in preparation a,b). The ecological significance of its coral reef communities and fisheries, and the presence of a resident population of the endangered Green Turtle, *Chelonia mydas* (Pagán-Villegas et al., 1999), lead the P.R. Department of Natural and Environmental Resources (PRDNER) to designate the Luis Peña Channel Natural Reserve on June 1, 1999. Further, PRDNER Administrative Order No. 99-15 prohibited all fishing and collecting activities within the Reserve on September 30, 1999, becoming a *de facto* Marine Fishery Reserve (MFR). Also, the coastal waters of Culebra Island support an extensive seagrass bed system (Figure 1) that was designated in October 2, 1998 as a critical habitat (Resource Category 1) for *C. mydas* by the National Marine Fisheries Service (*Fed. Reg*. 63(170):46693-46701; Sept. 2, 1998). Part of the Culebra's Resource Category 1 seagrass habitats are located within the Luis Peña Channel No-Take Natural Reserve (LPCNR). According to Pagán-Villegas et al. (1999) and to Hernández-Delgado et al. (2000), seagrasses are the most abundant and one of the most significant types of habitats within the LPCNR. However, these habitats have never been quantitatively described.

Seagrasses are marine angiospermae, which have true roots, stems and leaves containing vascular tissues, and inconspicuous flowers that form spiny seeds (Littler et al., 1989). They form extensive meadows in shallow marine unconsolidated bottoms (Odum, 1974). Seagrasses can significantly modify the physical, chemical and geological properties of shallow coastal bottoms (Vicente, 1992). They posses a high economic value because they can stabilize unconsolidated sediments, thus damping wave action and reducing shoreline erosion rates (Fonseca and Calahan, 1992). Also, seagrasses function as a natural water filter, thus can reduce the concentration of solid suspended material in the water column. In addition, seagrass beds are a major source of primary productivity in the tropical marine environment (Odum et al., 1959; Odum, 1974; Vicente, 1992: Wilzbach et al., 2000) and constitute a nursery ground for many commercially-important fish and invertebrate species (Pollard, 1984; Hettler, 1989), thus supporting important marine fishery resources (Lewis and Stoner, 1983; Connolly, 1994). Seagrass beds also support a high faunal (Glynn, 1964; Holmquist et al., 1989) and algal diversity (Littler et al., 1989), thus constituting also foraging grounds for several endangered marine species (Rathbun et al., 1985). These habitats are also major sites of human recreation and scientific studies (Kemp, 2000)

According to Vicente (1992), a total of seven seagrass species have been informed from Puerto Rico, including: 1) Turtle Grass, *Thalassia testudinum* (Banks ex König); 2) Sea Vines, *Halophila decipiens* (Ostenfeld); 3) *H. baillonis* (Ascherson); 4) *H. engelmannii* (Ascherson); 5) Manatee Grass, *Syringodium filiforme* (Kützing); 6) Shoal Grass, *Halodule wrightii* (Ascherson); and 7) Widgeon Grass (Ditch Grass), *Ruppia maritima* (Linnaeus). Seagrasses have become a widely used indicator of the relative health and ecological condition of temperate (Dennison et al.,1993) and tropical estuarine ecosystems (Kemp, 1983; Bortone, 2000).

According to the Caribbean Fishery Management Council (CFMC, 1998a), seagrass beds support important conch fisheries throughout the wider Caribbean region, thus are considered to represent a significant essential fish habitat (EFH). During the last two decades, conch fisheries within the U.S. Caribbean have been considered to be severely overfished and declining (Wood and Olsen, 1983; Appeldoorn, 1991, 1992; Valle-Esquivel, 2002). Queen Conch (*Strombus gigas*) landings in Puerto Rico, which peaked up to 400,000 lbs in 1983, have declined to only 29,000 lbs in 2000 (Valle-Esquivel, 2002), a 93% decline. This species was designated as overfished in the 2000 Report to Congress on the Status of U.S. Fisheries and a total prohibition of harvest and possession of conch in the U.S. Caribbean EEZ is under consideration as an amendment to the Queen Conch FMP (CFMC, 1996, 2001). Currently, there is a seasonal closure of the Queen Conch fishery in Puerto Rico during its reproductive peak, between July 1 and September 30 of each year (PRDNER Executive Order 97-09, June 12, 1997).

Large populations of juvenile and young adult Queen Conch, *S. gigas*, and of the Fighting Conch, *S. pugilis*, have been observed and quantified within the LPCNR (Hernández-Delgado, unpublished data), but have never been quantified since its designation until the present study. Previously, Rosario (1995), Appeldoorn (1996), and Mateo-Rabelo (1997) documented the importance of Culebra Island for Queen Conch. However, all of those efforts consisted of estimating the Queen Conch densities on deep waters (>20 m) using rapid survey methods, but did not provide any insight on the abundance and size class distributions of the three Conch species which are being landed in Culebra Island, namely the Queen Conch, *S. gigas*, the Fighting Conch, *S. pugilis*, and the Milk Conch, *Strombus costatus*. In addition, there is a lack of information concerning the status of spawning stocks and juvenile stocks of these species in the region. Finally, and from the EFH standpoint, there is also a lack of information regarding the status of their habitats. One of the most important tasks of the CFMC in relation to EFH for each of the life stages of Conch species is to identify the location and distribution of individuals, as well as to assess the condition of these habitats (CFMC, 1998a).

The main objective of the designation of the LPCNR was to restore its overfished reef fish populations and to protect important juvenile fish nursery grounds (Pagán-Villegas et al., 1999). But the role of MFRs to help restore Conch populations has been poorly documented. The 475 hectares of the LPCNR is the only location under the Puerto Rican state waters jurisdiction that Queen Conch fishing is permanently prohibited. Therefore, the LPCNR can become a permanent control monitoring station to address natural fluctuations of Conch populations. Also, it can be used as a model management tool to help restore shallow water populations of the Queen Conch within the U.S. Caribbean. Thus, it is important to establish a baseline data bank regarding the status of Conch populations and their EFHs within the LPCNR to which future stock and EFH assessments can be compared.

The first objective of this study was to provide a baseline quantitative description of the actual status of the LPCNR shallow-water seagrass bed communities. These constitute part of the Conch species EFH. Our second objective was to document the actual status of the three Conch species populations within the LPCNR. Data collected within the LPCNR boundaries was compared to two nearby control sites outside the Reserve.

METHODS

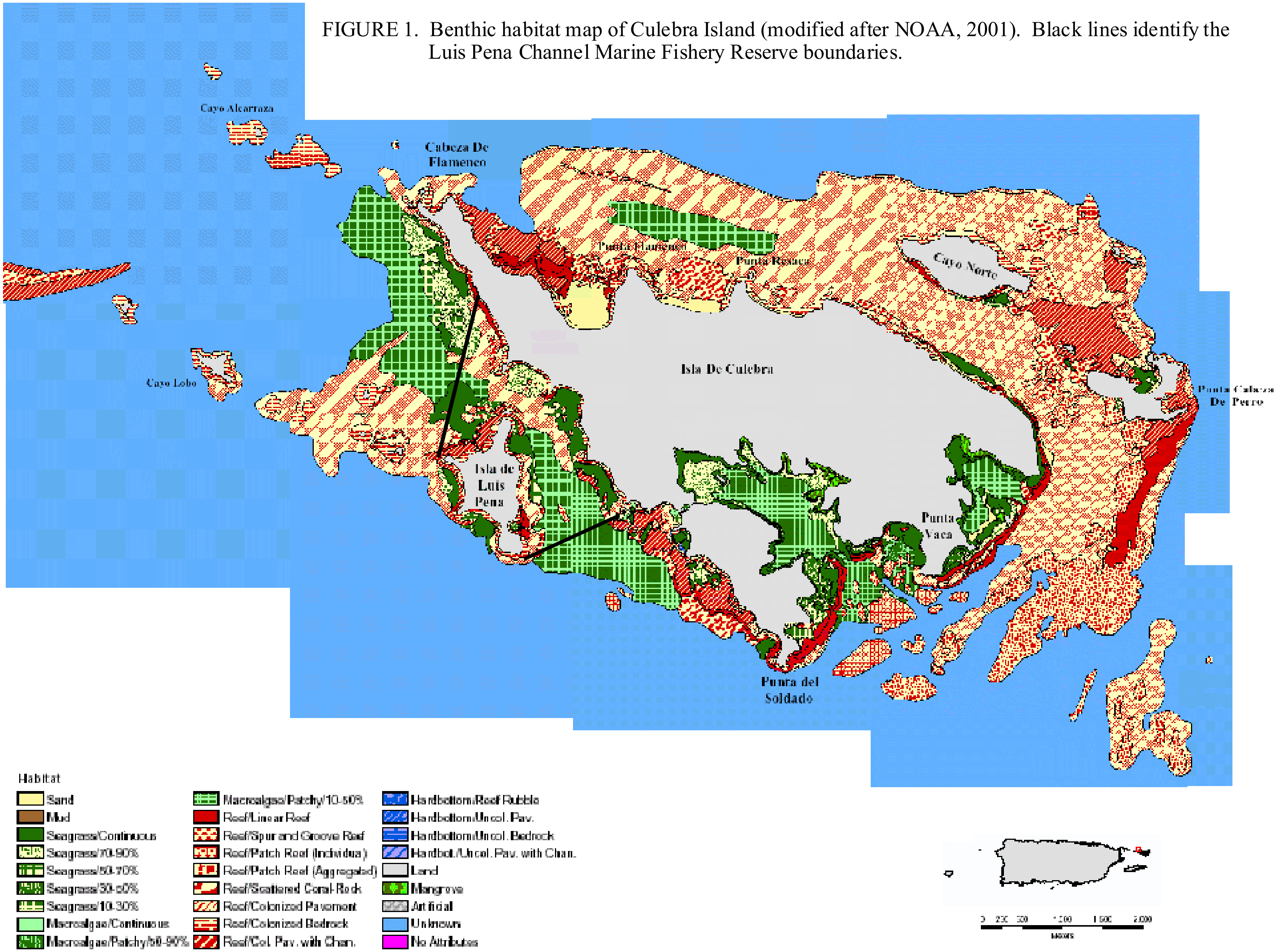
Study sites.

This study was carried out within the 475-hectare LPCNR, Culebra Island, P.R. (Figure 1). Culebra Island is located at approximately 28 km off Fajardo, northeastern Puerto Rico (Figure 2). The LPCNR benthic communities were originally described by Pagán et al. (1999) and by Hernández-Delgado et al. (2000). A dot grid matrix (n=74 dots) was overlayed on top of Figure 2 (modified after NOAA, 2001) to obtain a rough estimate of the area of each benthic category type. Major benthic categories included:

Seagrass (continuous)	102.7 hectares	21.62%
Sesgrass (70-90%)	19.2 hectares	4.05%
Seagrass (50-70%)	12.8 hectares	2.70%
Seagrass (30-50%)	19.2 hectares	4.05%
Macroalgal plain (10-50%)	89.9 hectares	18.92%
Linear reef	19.2 hectares	4.05%
Scattered coral rock	32.1 hectares	6.76%
Colonized bedrock	51.4 hectares	10.81%
Colonized pavement with channels	44.9 hectares	9.46%
Colonized pavement	83.5 hectares	17.57%

From this analysis, nearly 49% of the benthic habitats within the LPCNR are coral reefs or colonized pavement type of habitats, suggesting that previous estimates by Pagán et al. (1999) and by Hernández-Delgado et al. (2000) largely underestimated the area of coralline communities. This was most probably due to the absence of data from deepwater habitats. Approximately 32.4% of the benthic habitats within the LPCNR are seagrass habitats, an estimate which is fairly similar to the previous estimates of Pagán et al. (1999) and Hernández-Delgado et al. (2000). Also, nearly 19% of the benthos was identified as macroalgal plains.

FIGURE 1. Benthic habitat map of Culebra Island (modified after NOAA, 2001). Black lines identify the Luis Pena Channel Marine Fishery Reserve boundaries.



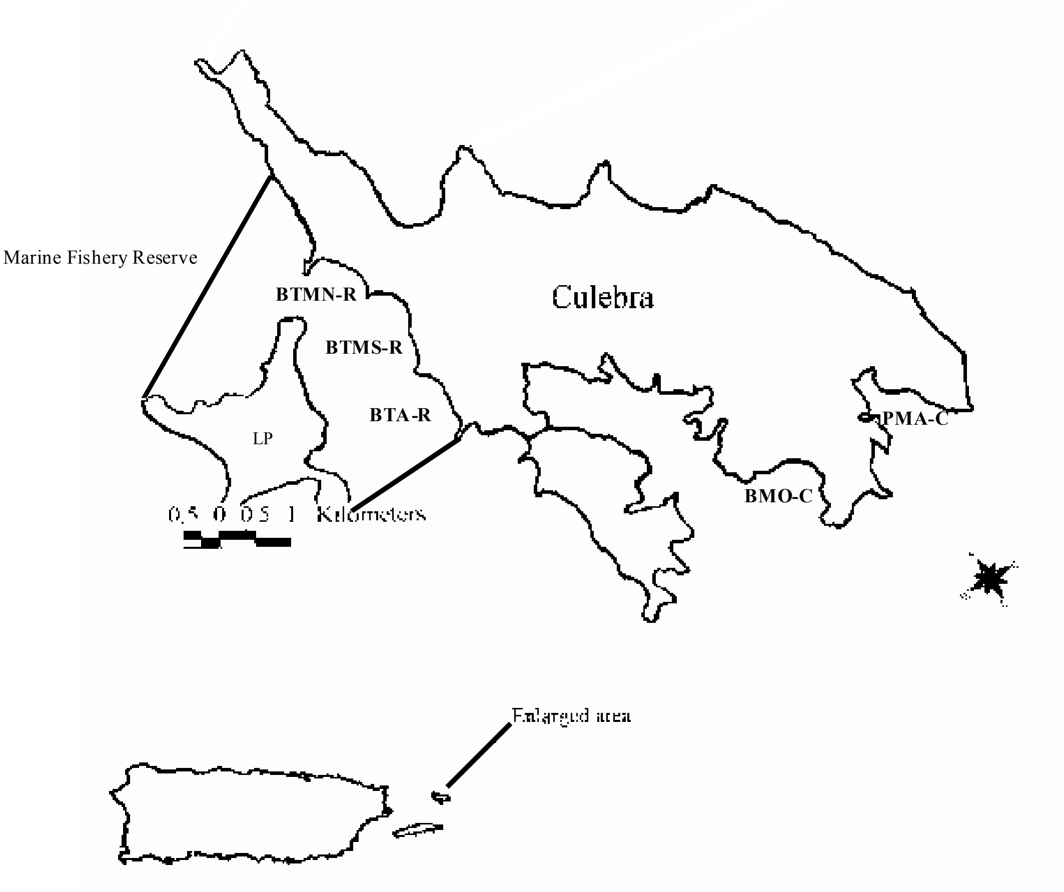


FIGURE 2. Study sites: A) Luis Pena Channel Marine Fishery Reserve: BTMN-R= Bahia Tamarindo (North); BTMS-R= Bahia Tamarindo (South); BTA-R= Bahia Tarja. B) Control sites outside the Reserve: BMO-C= Bahia Mosquito; PMA-C= Puerto de Manglar.

Maroalgal plains and colonized pavements are the predominant types of benthic habitats within the LPCNR, accounting for over 36% of the total benthos. Considering the fact that Conch species EFHs mostly include seagrass beds, macroalgal plains and colonized pavements (=hard grounds) (CFMC, 1998), it has been estimated that approximately 327.3 hectares (68.9%) could be considered as a potential Conch EFH within the LPCNR.

A total of five sampling stations were studied (Figure 2). Three seagrass bed sampling stations were studied within the LPCNR: BTMN-R) Bahía Tamarindo-North (18°19'21.87"N; 65°19'36.55"W); BTMS-R) Bahía Tamarindo-South (18°19'04.92"N; 65°19'08.28"W); and BTA-R) Bahía Tarja (18°18'15.26"N; 65°18'44.79"W). Two control (outside the LPCNR) seagrass bed sampling stations were selected for comparative purposes: BMO-C) Bahía Mosquito (18°17'44.60"N; 65°15'53.78"W); and PMA-C) Puerto de Manglar (18°18'21.54"N; 65°15'26.00"W).

All data was collected during the months of July and August, 2002, during the peak of the reproductive season of different Conch species (Reed, 1992; Valle-Esquivel, 2002).

Benthic components cover.

The structure of seagrass communities was characterized at all of the study sites. The percentage of benthic components (i.e, seagrasses, algae, other components) cover was determined along three (3) replicate 30 m-long line transects per sampling station and per depth contour, parallel to the shoreline. Transects were placed approximately at 15-20 m apart of each other Five (5) replicate 1.0 m^2 quadrats were haphazardously sampled along every transect line at fixed 6 m intervals. The “other components” category included sponges, isolated coral fragments, tunicates and open substrate. All surveys were limited to depth ranges of 1.5 to of 7 m. We tested the hypotheses that there were no significant differences in the benthic cover of different components among sites (LPCNR vs. control sites) by using the Kruskal-Wallis test (Kruskal and Wallis, 1952), a non-parametric analysis of variance (ANOVA) by ranks. This method was selected over a parametric one-way ANOVA because part of the data was not normal. This was most probably due to the patchy nature of seagrass bed habitats and the probably low number of samples relative to the patchy and discontinuous nature of some of the sampling stations (i.e., PMA-C). According to Andrews (1954), under such circumstances, the Kruskal-Wallis test is the more powerful test. Data on proportions (% relative cover of each benthic category) was arcsine-(√)-transformed (Zar, 1984). The statistical power of the samples was calculated as $1-\beta$, where β= sampling error which was calculated as [Standard Error/Mean] x 100 (Zar, 1984). Data on statistical power was summarized in the Appendix.

Benthic components density.

Seagrass density (# shoots/m^2) and calcareous erect algal density (# stalks/m^2) were determined along three (3) replicate 30 m-long line transects as described above. Five (5) replicate 0.0225 m^2 (0.15 X 0.15 m) quadrats were haphazardously sampled along every transect line at 6 m intervals. We tested the hypotheses that there were no significant differences in the seagrass or calcareous algal densities among sites (LPCNR vs. control sites). Density data on seagrass density was individually analyzed for each one of the three seagrass species present at the study sites: *Thalassia testudinum*, *Syringodium filiforme*, and *Halodule wrightii*. Calcareous algae were pooled and included *Halimeda* spp., *Penicillum* spp., and *Udotea* spp. Data was log-transformed (Zar, 1984) for statistical analysis by means of a Kruskal-Wallis test (Kruskal and Wallis, 1952). The statistical power of the samples was calculated as described above. Data on statistical power was summarized in the Appendix.

Epibenthic components standing crop biomass.

Seagrass and algal samples samples were haphazardously sampled using five (5) replicate 0.0225 m^2 (0.15 X 0.15 m) quadrats along each replicate 30-m long transect as described above. Samples were collected by cutting them off the substrate surface using scissors. Excess sediment and any living epizootic was removed in situ and samples were placed in individually identified zip lock bags. Excess water was decanted and bags were place on ice until analyzed within 24 h. Seagrass samples were sorted by species and decalcified using diluted 10% v/v HCL to remove incrusting calcareous algae (CARICOMP, 1994). Calcareous algae and macroalgae were separated and grouped by functional group (Steneck and Dethier, 1994). All samples were washed with freshwater before and after the HCL treatment, and dried in a drying oven at 60°C/24 h. Standing crop biomass data was converted to g/m^2. Also, data was converted to kg/ha and to ton/acre in order to facilitate comparison to data obtained from the literature. We tested the hypotheses that there were no significant differences in the benthic components standing crop biomass among sites (LPCNR vs. control sites). Data was log-transformed (Zar, 1984) for statistical analysis by means of a Kruskal-Wallis test (Kruskal and Wallis, 1952). The statistical power of the samples was calculated as described above. Data on statistical power was summarized in the Appendix.

Thalassia testudinum mean leaf length and width.

Thalassia testudinum leaves from shoots collected above were measured using a plastic ruler for length and width to the nearest mm. Length was measured from the base to the tip of the seagrass blade. Width measurements were consistently made at approximately 1-2 cm off the base of each leaf. A total of 1,205 leaves were measured as follows: BTMN-R (n=197); BTMS-R (n=375); BTA-R (n=419); BMO-C (n=391); and PMA-C (n=214). We tested the hypotheses that there were no significant differences in the *T. testudinum* length and width values among sites (LPCNR vs. control sites). Data was log-transformed for statistical analysis by means of a One-Way ANOVA (Zar, 1984) following a Shapiro-Wilk test for data normality (Shapiro and Wilk, 1965; Shapiro et al., 1968). The statistical power of the samples was calculated as described above. Data on statistical power was summarized in the Appendix. In addition, size class distribution of leaf length and width were documented based on the following categories for length: <50 mm, 51-100 mm, 101-150 mm, 151-200 mm, 201-250 mm, 251-300 mm, and >300 mm. The following categories were used for leaf width: <2 mm, 3-4 mm, 5-6 mm, 7-8 mm, 9-10 mm, 11-12 mm, and >12 mm.

Leaf Area Index (LAI).

Thalassia testudinum leaves from shoots were measured as described above. Leaf length (L) and width (W) data was used to calculate leaf area: L x W. These data were used to calculate the average leaf surface area and the Leaf Area Index (LAI):

$$LAI = (\text{leaf surface area})/1\ \text{m}^2$$

We tested the hypothesis that there were no significant differences in the *T. testudinum* LAI among sites (LPCNR vs. control sites). The statistical power of the samples was calculated as described above. Data on statistical power was summarized in the Appendix.

Multivariate analysis of seagrass bed communities

Multivariate analysis techniques were tested for their ability to discriminate patterns of geographical variation in the structure of seagrass bed communities. The seagrass community data set, based on the proportional cover of each major benthic component category, was compiled into a matrix and imported into PRIMER ecological statistics software package (Clarke, 1993; Clarke and Warwick, 1994, 2001) for multivariate analysis. Raw proportional cover values were square-root transformed in order appropriately weight the less abundant benthic categories (Clarke and Warwick, 2001; McField et al, 2001). Study sites were first classified with hierarchical clustering (CLUSTER) using the Bray-Curtis group average linkage method (Bray and Curtis, 1957; Simboura et al., 1995) and then ordinated using a non-metric multidimensional scaling (MDS) plot (Kenny and Rees, 1994; Clarke and Warwick, 2001). Site labels corresponded to each of our study sites (Figure 2). Significant differences between groups of sites were tested using PRIMER's multivariate equivalent of an ANOVA called ANOSIM, which means "analysis of similarities" (Clarke and Green, 1988; Clarke, 1993; Clarke and Warwick, 2001). Both, global and pairwise tests were carried out by means of ANOSIM. This test was based on 5000 permutations and had no built in assumptions about the data distribution (Mcfield et al., 2001). They key taxa responsible for the differences between groups of sites were determined using PRIMER's SIMPER routine (Clarke, 1993; Clarke and Warwick, 2001).

Conch densities and size class distribution.

Conch densities were determined by using the belt transect method. A total of 94 randomly placed 50 x 4 m long belt transects were surveyed throughout the study sites at depths ranging from 1.5 to 7 m as follows: BTMN-R (n=16); BTMS-R (n=11); BTA-R (n=26); BMO-C (n=16); and PMA (n=25). Ten replicate transects was originally set as the sample size, but given the low densities of Conch, sampling effort was increased as feasible in order to reduce sampling error and increase statistical power. Each transect was established parallel to the shoreline at each sampling site. Conch were counted, identified, and measured using a cm-calibrated metric tape and a mm-calibrated caliper. Two types of measurements were obtained: 1) maximum shell length (mm); and 2) lip width (mm). We tested the hypotheses that there were no significant differences in the conch densities, maximum shell length and lip width among sites (LPCNR vs. control sites). Data was log-transformed (Zar, 1984) for statistical analysis by means of a Kruskal-Wallis test (Kruskal and Wallis, 1952). The statistical power of the samples was calculated as described above. Data on statistical power was summarized in the Appendix. Size class distribution of the maximum shell length for the Queen Conch, *Strombus gigas*, was based on the following categories: <50 mm, 50-100 mm, 100-150 mm, 150-200 mm, 200-250 mm, 250-300 mm, and >300 mm. The following categories were used for *S. gigas* lip width: <1 mm, 1-2 mm, 2-3 mm, 3-4 mm, 4-5 mm, and >5 mm. Size class distribution of the maximum shell length for the Fighting Conch, *S. pugilis*, and of the Milk Conch, *S. costatus*, were based on the following categories: <80 mm, 80-100 mm, 100-120 mm, 120-140 mm, 140-160 mm, 160-180 mm, 180-200 mm, and >200 mm. The following categories were used for lip width on the latter two species: <0.5 mm, 0.5-0.7 mm, 0.7-0.9 mm, 0.9-1.1 mm, and >1.1 mm.

Relationships between maximum shell length and lip width.

A linear regression was carried out to determine the relationship between maximum shell length vs. lip width to establish a baseline data bank regarding the trends in the protected fishery within the LPCNR.

Status of Conch populations before and after the LPCNR designation.

Conch densities (#/ha) at BTMS-R and at BTA-R were compared between years 1998 (one year prior to the LPCNR designation) and 2002 (3 years after designation) during the peak of the reproductive season (=seasonal fishery closure). Similar methods (see above) were used to estimate Conch densities in both years. A total of 8 replicate transects were assessed at BTMS-R during 1998, vs. 11 during 2002. A total of 10 replicate transects were surveyed at BTA-R during 1998, vs. 26 during 2002. Data was log(+1)-transformed (Zar, 1984) and analyzed for year effects by means of a Kruskal-Wallis test (Kruskal and Wallis, 1952).

Water transparency.

Horizontal water transparency was determined at each sampling station in triplicates using a 51-cm diameter Secchi disk. Lectures were carried out at a depth of 1 m below surface.

RESULTS

Benthic components cover.

There were highly significant differences in the three major benthic components cover among study sites (Table 1). Mean values were summarized in Table 2. The highest % of seagrass cover was observed at BTMN-R (97%), followed by BTM-S (93%) and BMO-C (85%) (Figure 3). A comparison of means analysis ($p=0.05$) demonstrated that these three sites were significantly different from other sites (Kruskal-Wallis; $p<0.0001$). Thus, although the highest % of seagrass cover was observed in 2 out of the 3 study sites within the LPCNR, the lowest % cover was also observed at BTA-R (57%). Therefore, the null hypothesis of no significant difference in the % of seagrass cover among the LPCNR and the control sites outside was not rejected. A higher % of seagrass cover was not exclusive of the LPCNR. A mean statistical power of 94% for seagrass cover was the highest among all of the parameters measured (Appendix, Table A1).

Similarly, there was a highly significant difference (Kruskal-Wallis; $p<0.0001$) in the % of algal cover (Table 1). Mean values were summarized in Table 2. The highest % of algal cover was observed at PMA-C (26%), followed by BMO-C (4%) and BTA-R (3%) (Figure 4). A comparison of means analysis ($p=0.05$) demonstrated that only PMA-C was significantly different from other sites. Although the highest % of algal cover was observed one of the control sites, there were no differences among the remaining control site and among the LPCNR. Therefore, the null hypothesis of no significant difference in the % of algal cover among the LPCNR and the control sites outside was not rejected. Low % of algal cover was not exclusive

TABLE 1. Comparison of major seagrass bed parameters among sites.

Parameter	Kruskal-Wallis Statistic	*p*	Comparison of means (*p*=0.05)
% Seagrass cover**	36.92	<0.0001	142 425 53
% Algal cover	36.50	<0.0001	5 3421
% Other cover	36.69	<0.0001	3 2541
Density T. testudinum (#/m^2)***	42.22	<0.0001	41 325
Density S. filiforme (#/m^2)	10.87	0.0281	2351 3514
Density H. wrightii (#/m^2)	22.33	0.0002	54123 (N.S.)****
Density Calcareous algae (#/m^2)	4.74	0.3152	35421 (N.S.)
Biomass T. testudinum (g/m^2)	17.66	0.0014	14 4235
Biomass S. filiforme (g/m^2)	9.85	0.0431	1523 5234
Biomass H. wrightii (g/m^2)	7.71	0.1028	45213 (N.S.)
Biomass Calcareous algae (g/m^2)	14.64	0.0055	5413 4132
Biomass Macroalgae (g/m^2)	21.11	0.0003	51234 (N.S.)

*Kruskal-Wallis non-parametric ANOVA; d.f.=4,70.

**Data on proportions were Arcsin(Square-root)-transformed.

***Log-transformed data.

****N.S.= Not Significant.

1=BTMN-R; 2=BTMS-R; 3=BTA-R; 4=BMO-C; 5=PMA-C.

TABLE 2. Percentage cover of major epibenthic categories at each study site.

Site	% Seagrass	% Algae	% Other
BTMN-R	96.9±0.9	0.8±0.2	1.3±0.7
BTMS-R	93.1±1.6	2.5±0.6	4.5±1.5
BTA-R	57.0±3.3	2.9±0.5	40.1±3.2
BMO-C	84.9±5.6	4.4±1.6	10.7±4.2
PMA-C	66.0±8.9	26.0±6.1	8.0±3.2

*Mean ± one standard error.

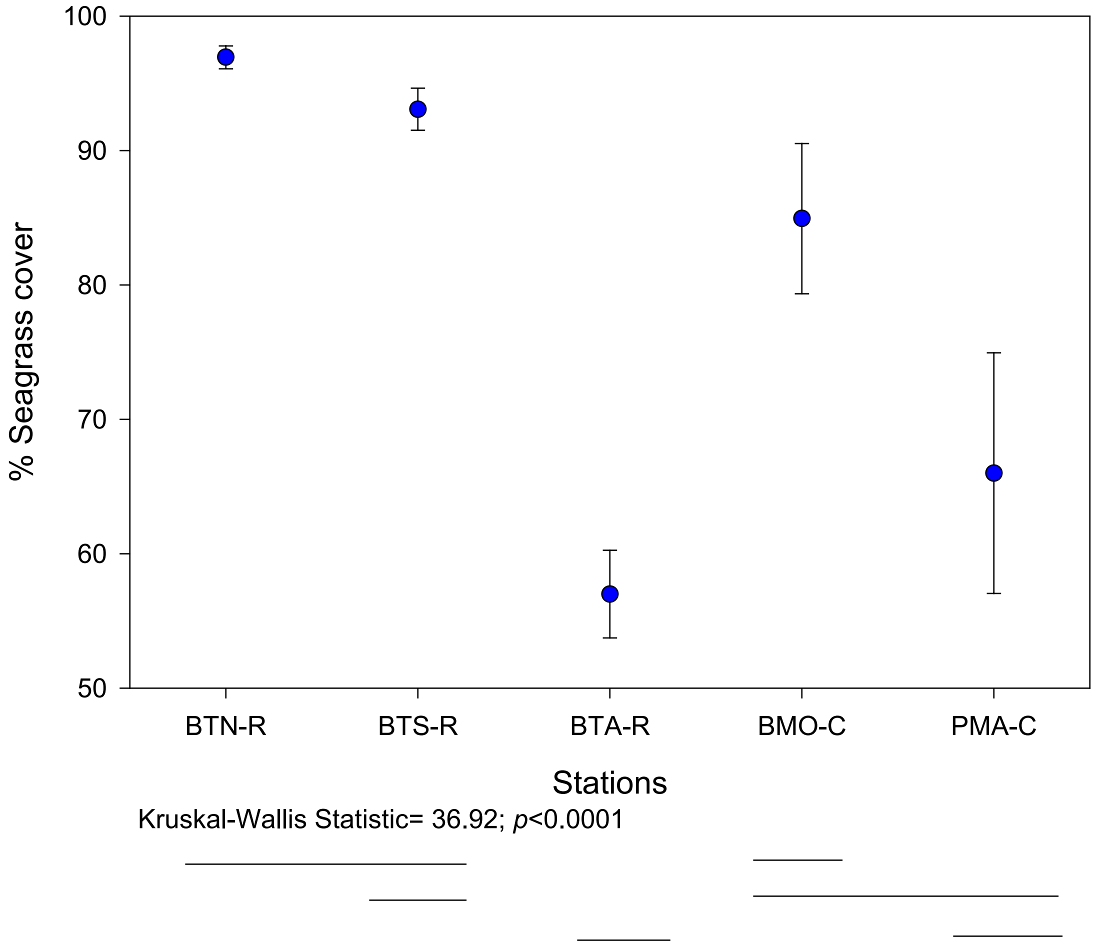
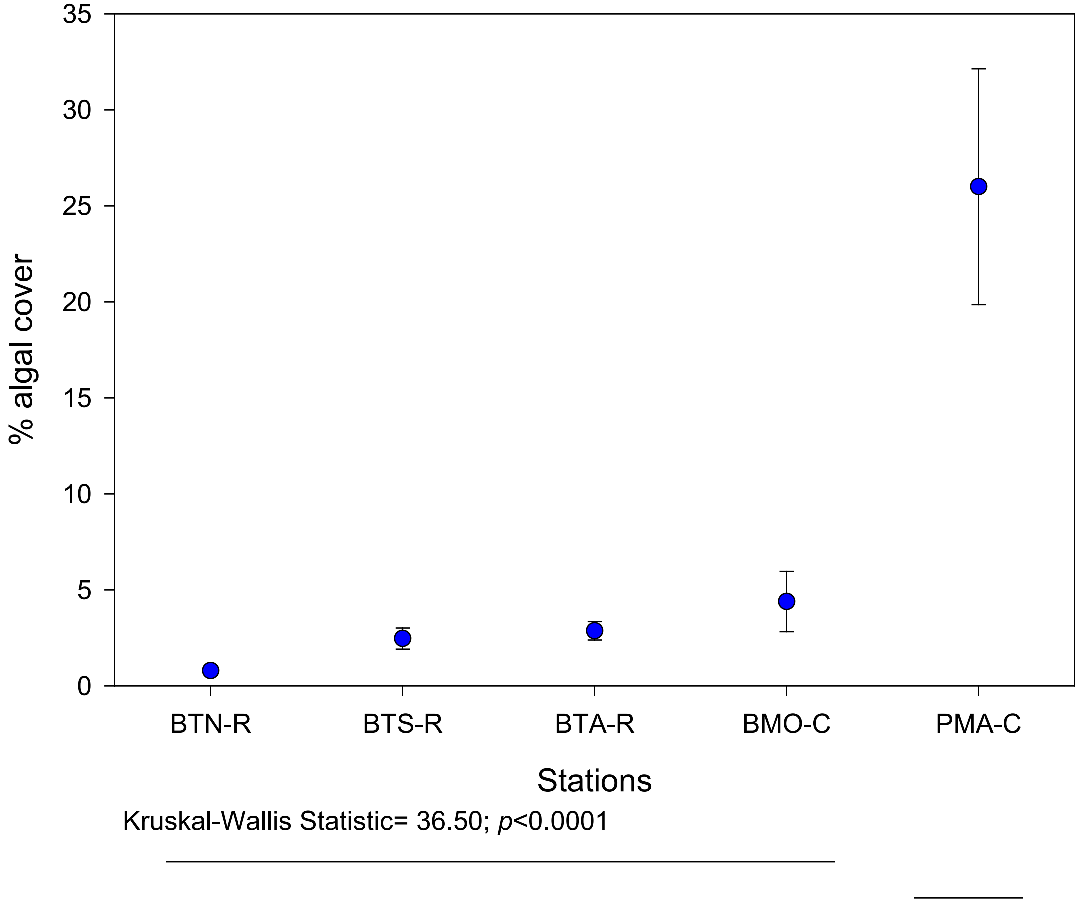


FIGURE 3. Percentage of seagrass cover (mean±one standard error). Horizontal bars summarize post hoc analysis results.



Kruskal-Wallis Statistic= 36.50; *p*<0.0001

FIGURE 4. Percentage of algal cover (mean±one standard error). Horizontal bars summarize post hoc analysis results.

of the LPCNR. A mean statistical power of 74% for algal cover was considered marginally high (Appendix, Table A1).

There was also a highly significant difference (Kruskal-Wallis; $p<0.0001$) in the % of cover of other benthic components (Table 1). Mean values were summarized in Table 2. The highest % of cover of other components was observed at BTA-R (40%), followed by BMO-C (11%) and PMA-C (8%) (Figure 5). A comparison of means analysis ($p=0.05$) demonstrated that only BTA-R was significantly different from other sites. Similarly, the highest % of cover of other components was observed one of the LPCNR sites. But, there were no differences among the remaining LPCNR and control sites. Therefore, the null hypothesis of no significant difference in the % of other components cover among the LPCNR and the control sites outside was not rejected. Low % of cover of other components were not exclusive of the LPCNR. However, a mean statistical power of 65% for other components cover was considered low (Appendix, Table A1).

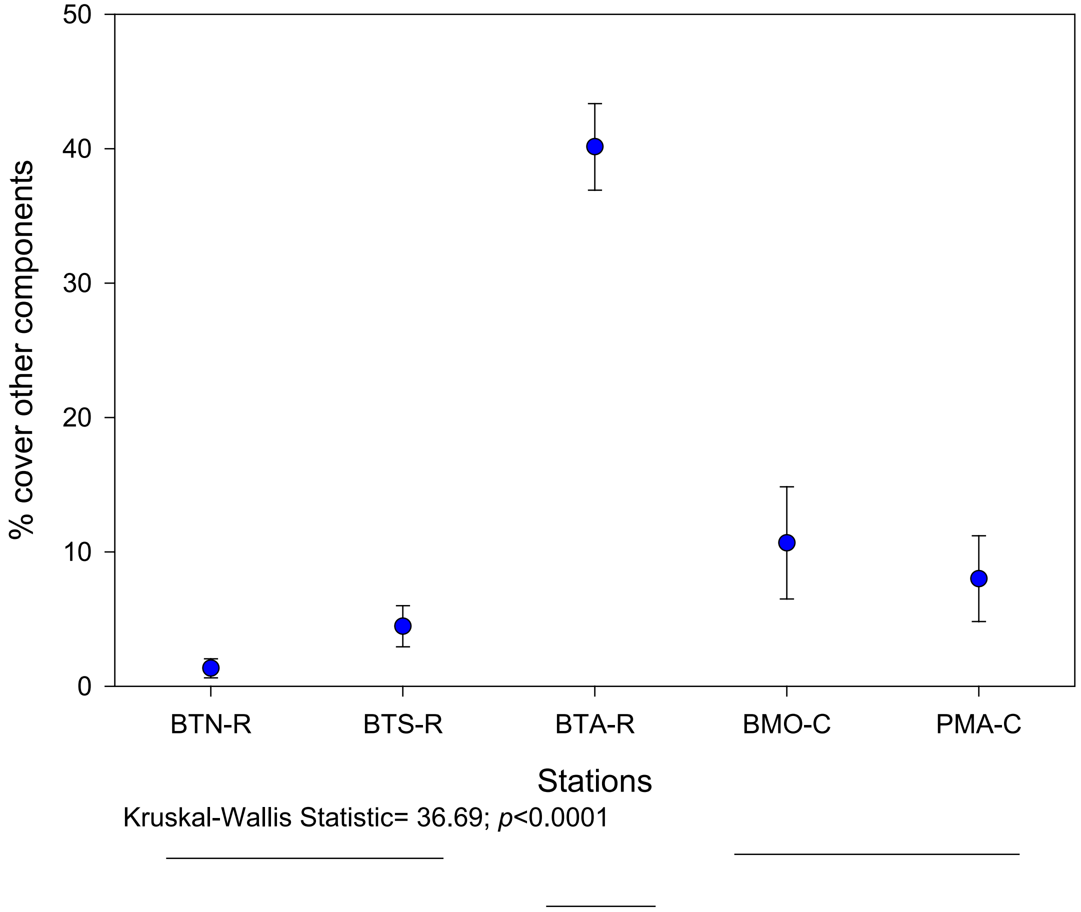
Benthic components density.

The density data of all benthic components was summarized in Table 3. There was a highly significant difference (Kruskal-Wallis; $p<0.0001$) in the density of *Thalassia testudinum* among sites (Table 1), with BMO-C averaging 1847 shoots/m^2 and BTMN-R a total of 1820 shoots/m^2 (Figure 6). The lowest density was observed at PMA-C, 660 shoots/m^2. A comparison of means analysis showed that these two sites were significantly different from the remaining LPCNR and control sites. Thus, the null hypothesis of no significant differences in the density of *T. testudinum* among sites within and outside the LPCNR was not rejected.

TABLE 3. Density of major epibenthic categories at each study site (# shoots/m^2).

Site	T. testudinum	S. filiforme	H. wrightii	**Calcareous algae**
BTMN-R	1,820±151	447±130	0±0	29±12
BTMS-R	1,100±95	533±88	0±0	53±17
BTA-R	1,113±75	567±152	0±0	179±61
BMO-C	1,847±132	200±106	400±243	80±31
PMA-C	660±112	653±191	1,300±485	100±40

*Mean ± one standard error.



Kruskal-Wallis Statistic= 36.69; *p*<0.0001

FIGURE 5. Percentage of other epibenthic components cover (mean±one standard error). Horizontal bars summarize post hoc analysis results.

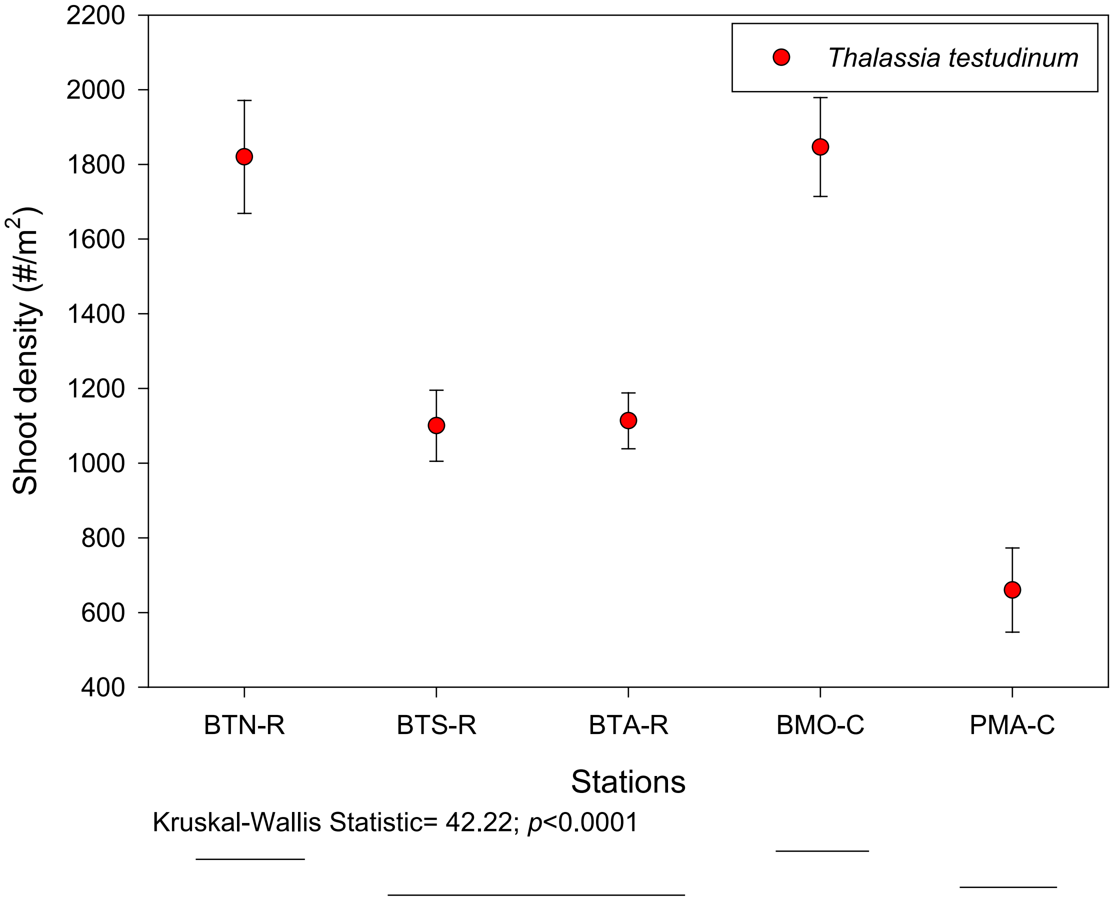


FIGURE 6. Shoot density of *Thalassia testudinum* (mean±one standard error). Horizontal bars summarize post hoc analysis results.

There were highly dense *Thalassia testudinum* stands both at the Reserve and control sites. The mean statistical power of this analysis was 90% (Appendix, Table A1).

There was a significant difference (Kruskal-Wallis; p=0.0281) in the density of *Syringodium filiforme* among sites (Table 1), with PMA-C averaging 653 shoots/m^2 and BTA-R a total of 567 shoots/m^2 (Figure 7). The lowest density was observed at BMO-C, 200 shoots/m^2. A comparison of means analysis showed that only BTMS-R and BMO-C were significantly different from the remaining LPCNR and control sites. We failed to reject the null hypothesis of no significant differences in the density of *S. filiforme* among sites within and outside the LPCNR. But the mean statistical power of this analysis was only 69% (Appendix, Table A1).

The Kruskal-Wallis test suggested that the density of *Halodule wrightii* (Figure 8) was significantly different among sites (Table 1), but a comparison of means analysis confirmed that this difference was not significant. Also, there were no significant differences in the densities of calcareous algal stalks among sites (Figure 9). The statistical power of these two analyses, however, was rather low (51% and 63%, respectively) (Appendix, Table A1).

Benthic components standing crop biomass.

Seagrasses were the most important benthic component at each study site (Tables 4a,b,c). The highest mean total seagrass standing stock biomass (Figure 10) was documented at BTMN-R (314 g/m^2), followed in the distance by BTMS-R (165 g/m^2). The lowest value was observed at BTA-R (90 g/m^2). These mean values are equivalent at BTMN-R to 3139 kg/ha (Table 4b) or to 7.63 ton/acres (Table 4c) of total seagrass standing crop biomass. At BTMS-R, it is

TABLE 4a. Standing stock biomass (g/m^2) of major epibenthic categories at each study site.

Site	T. testudinum	S. filiforme	H. wrightii	Calcareous algae	Macroalgae
BTMN-R	151.8±16.0	162.1±57.0	0±0	39.7±21.7	0±0
BTMS-R	143.2±15.9	11.6±2.2	10.3±6.0	15.9±7.7	0±0
BTA-R	83.1±8.5	6.5±1.5	0±0	14.5±7.7	0±0
BMO-C	126.0±16.2	2.5±0.9	3.7±2.4	52.3±17.6	0±0
PMA-C	76.3±17.8	19.7±6.9	3.3±2.8	306.9±113.3	11.2±6.8

*Mean ± one standard error.

TABLE 4b. Standing stock biomass (kg/ha) of major epibenthic categories at each study site.

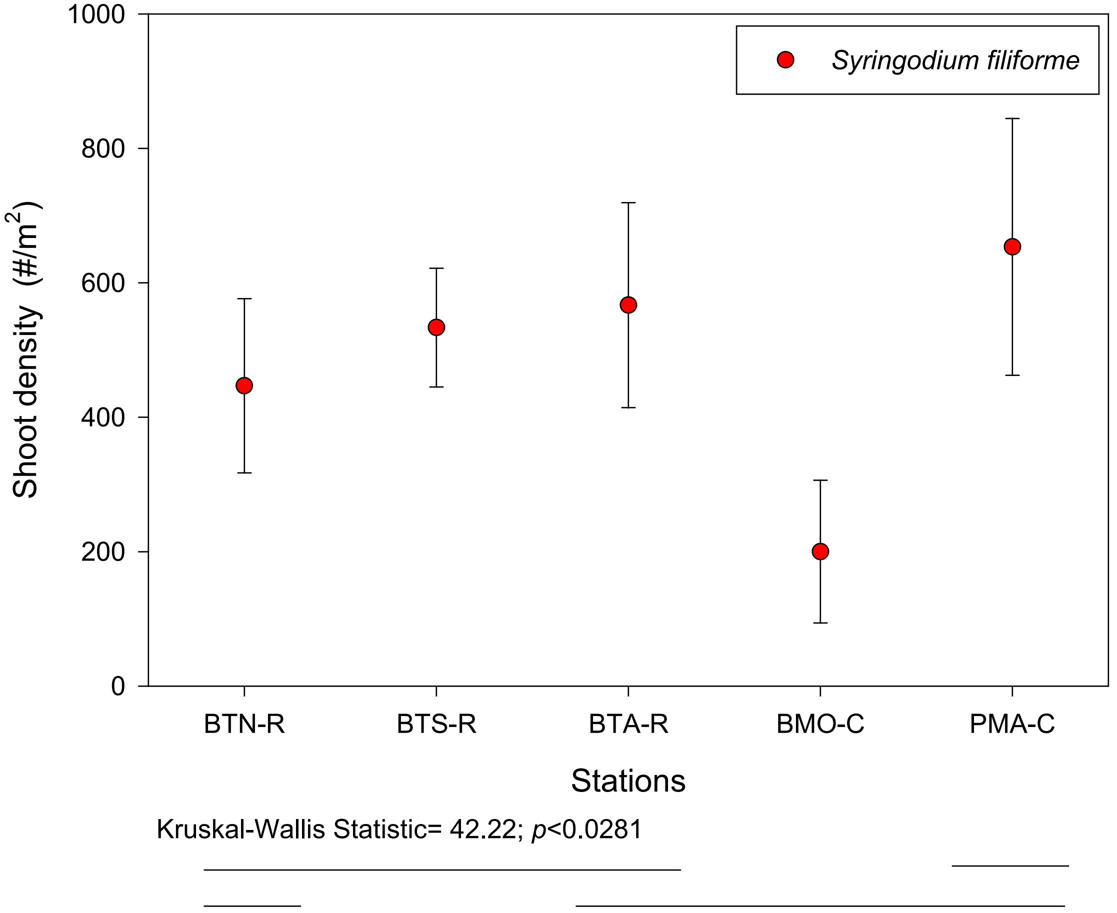
Site	T. testudinum	S. filiforme	H. wrightii	Calcareous algae	Macroalgae
BTMN-R	1518±160	1621±570	0±0	397±217	0±0
BTMS-R	1432±159	116±22	103±60	159±77	0±0
BTA-R	831±85	65±15	0±0	145±77	0±0
BMO-C	1260±162	25±9	37±24	523±176	0±0
PMA-C	763±178	197±69	33±28	3069±1133	112±68

*Mean ± one standard error.

TABLE 4c. Standing stock biomass (ton/acres) of major epibenthic categories at each study site.

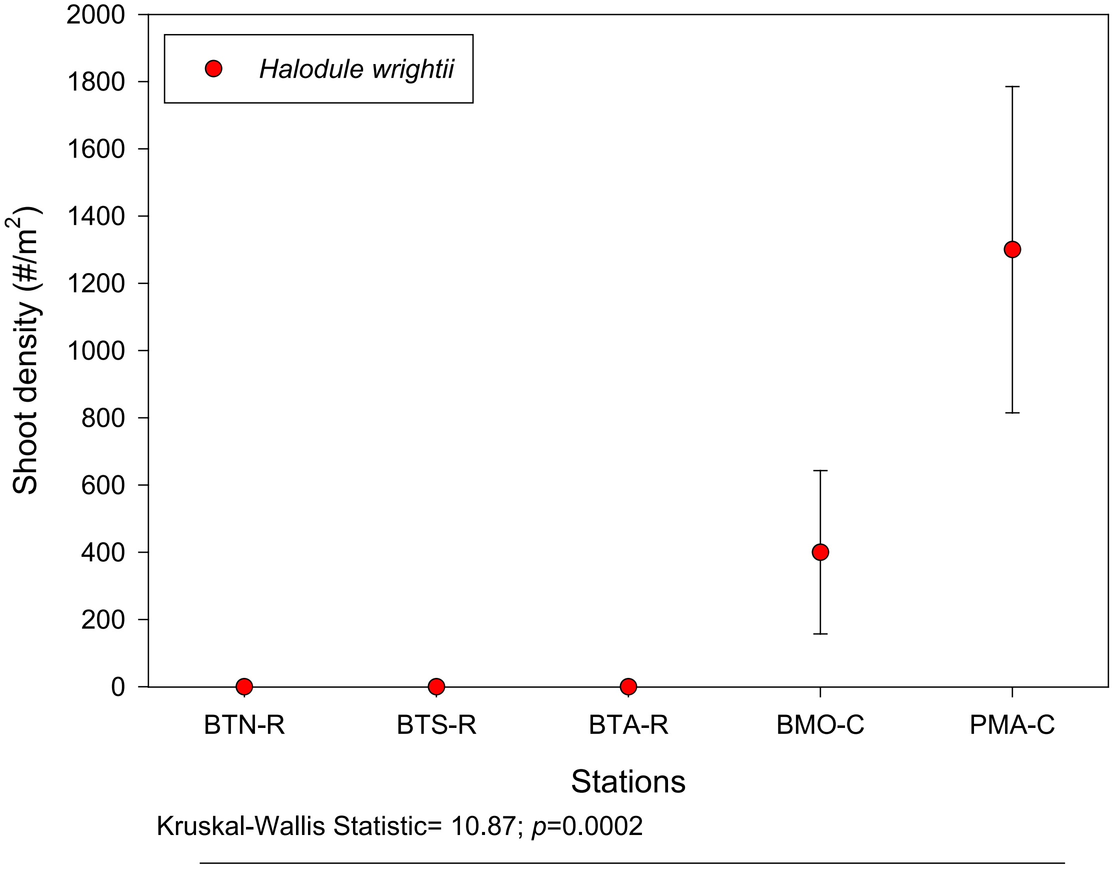
Site	T. testudinum	S. filiforme	H. wrightii	Calcareous algae	Macroalgae
BTMN-R	3.69±0.39	3.94±1.39	0±0	0.97±0.53	0±0
BTMS-R	3.48±0.39	0.28±0.05	0.25±0.15	0.39±0.19	0±0
BTA-R	2.02±0.21	0.16±0.04	0±0	0.35±0.19	0±0
BMO-C	3.06±0.39	0.06±0.02	0.09±0.06	1.27v0.43	0±0
PMA-C	1.86±0.43	0.47±0.17	0.08±0.07	7.46±2.76	0.27±0.17

*Mean ± one standard error.



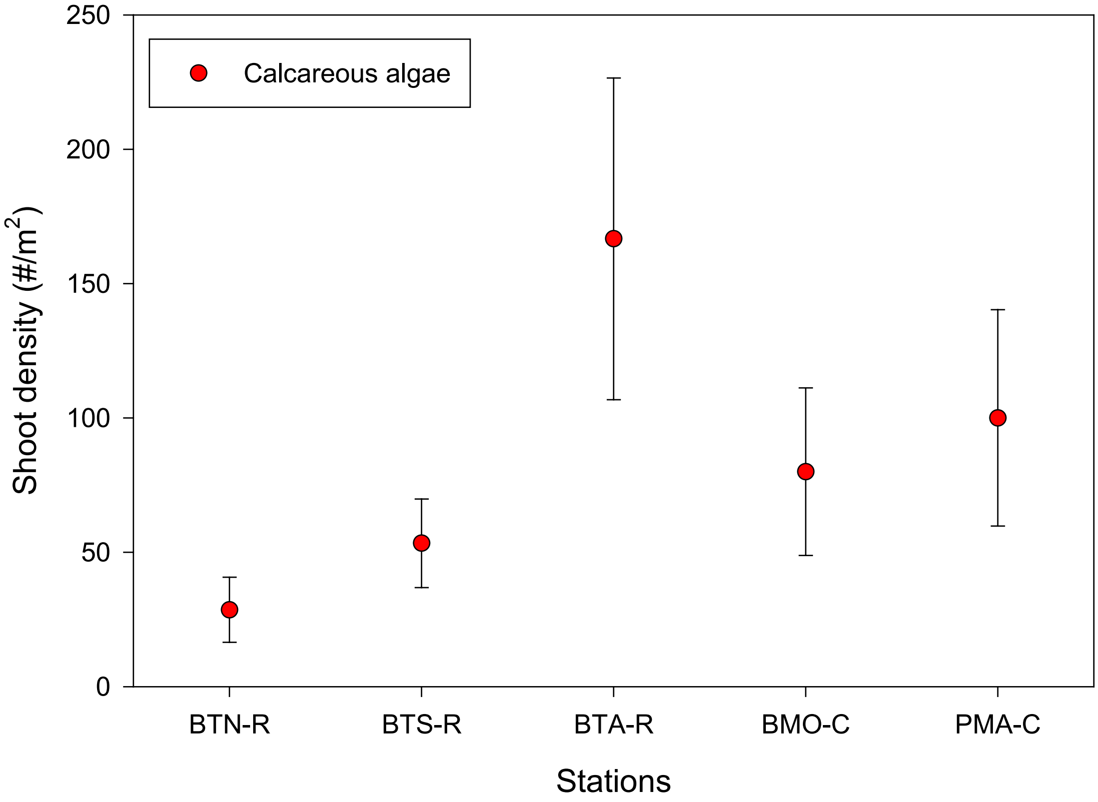
Kruskal-Wallis Statistic= 42.22; $p<0.0281$

FIGURE 7. Shoot density of *Syringodium filiforme* (mean±one standard error). Horizontal bars summarize post hoc analysis results.



Kruskal-Wallis Statistic= 10.87; *p*=0.0002

FIGURE 8. Shoot density of *Halodule wrightii* (mean±one standard error). Horizontal bars summarize post hoc analysis results.



Kruskal-Wallis Statistic= 4.74; *p*=0.3152

FIGURE 9. Shoot density of calcareous algae (mean±one standard error). Horizontal bars summarize post hoc analysis results.

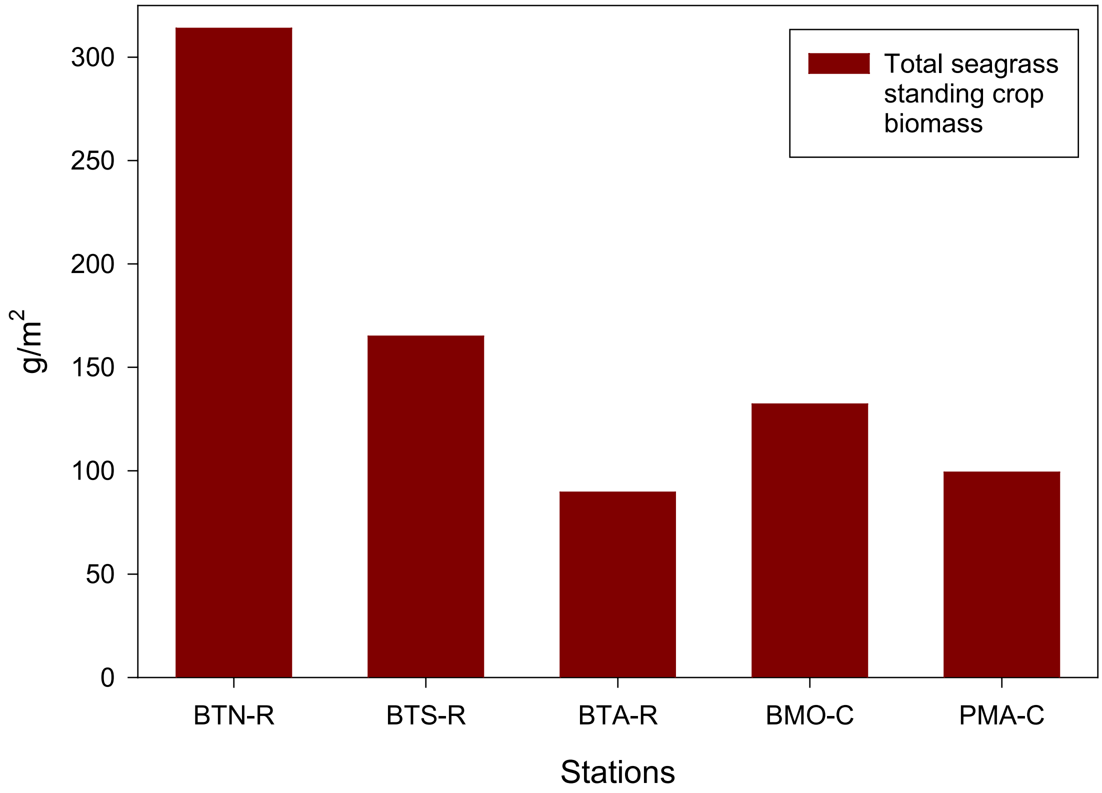


FIGURE 10. Mean total seagrass standing crop biomass (g/m^2).

equivalent to 1651 kg/ha or to 4.01 ton/acres. At BTA-R, it equals only 896 kg/ha or 2.18 ton/acre. Based on an average total seagrass biomass of 189.7 g/m^2 for seagrass bed habitats within the LPCNR (153.9 ha or 380.3 acres), is estimated that the potential total seagrass standing crop biomass for the LPCNR would be approximately 291,692 kg, which equals 1752 tons.

Thalassia testudinum standing crop biomass (Figure 11) was significantly different (Kruskal-Wallis, p=0.0014) among BTMN-R and, the group of sites composed by BTMS-R, BTA-R, and PMA-C (Table 1). The highest mean value was observed at BTMN-R (152 g/m^2) and the lowest at PMA-C (76 g/m^2). The statistical power of this test was 86% (Appendix, Table A1). *Syringodium filiforme* standing crop biomass (Figure 12) was also significantly different (Kruskal-Wallis, p=0.0431) among BTMN-R and BMO-C (Table 1). The highest mean value was observed at BTMN-R (162 g/m^2) and the lowest at BMO-C (2.5 g/m^2). The statistical power of this test was about 70% (Appendix, Table A1).

Standing crop biomass of *Halodule wrightii* (Figure 13) was not significantly different among sites (Table 1). This species was present only at sites that showed evidence of phyisical disturbance on the bottom (i.e., wave action, anchoring). But the calcareous algal standing crop biomass (Figure 14) was significantly different (Kruskal-Wallis, p=0.0055) among BTMN-R and BMO-C (Table 1). The highest mean value was observed at PMA-C (307 g/m^2) and the lowest at BTA-R (14.5 g/m^2). Finally, although the Kruskal-Wallis test suggested that macroalgal standing crop biomass (Figure 15) was significantly different among sites (Table 1), a comparison of means analysis demonstrated no statistical differences. However, the statistical

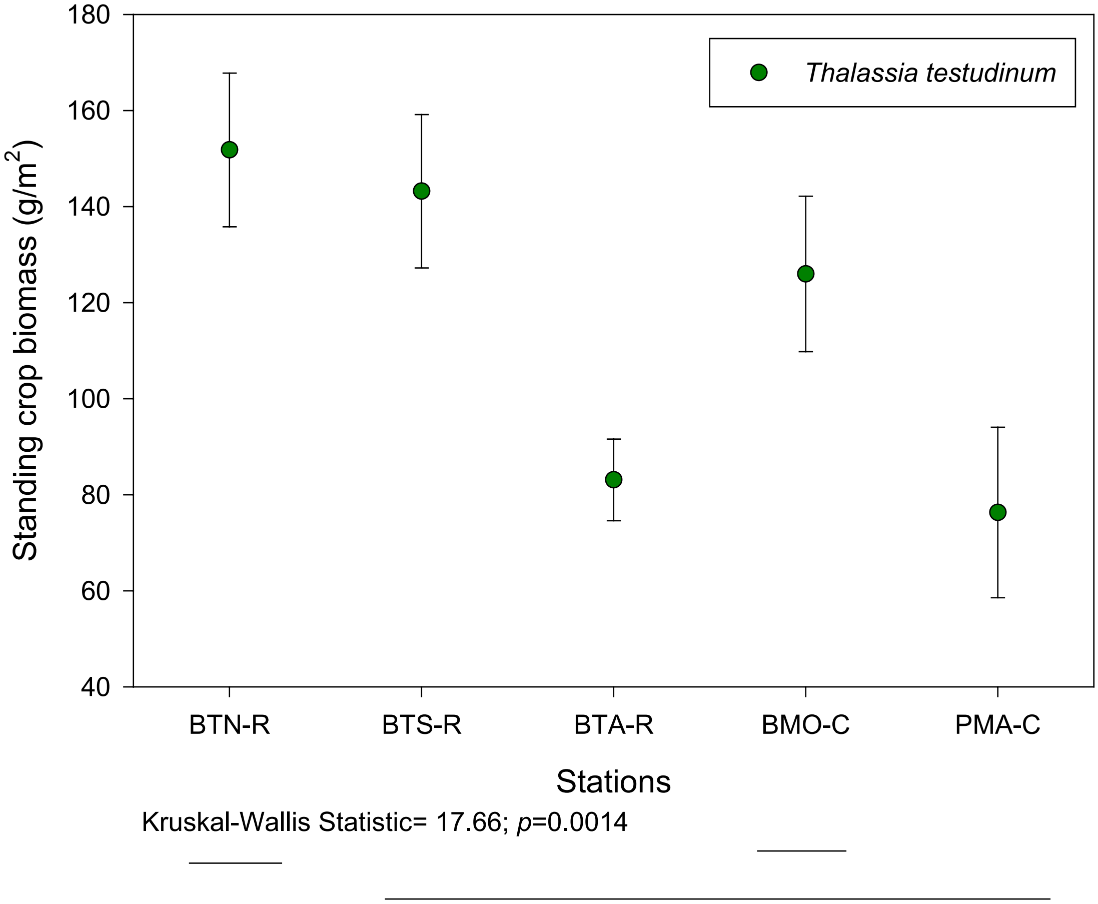


FIGURE 11. *Thalassia testudinum* standing crop biomass (g/m^2) (mean±one standard error). Horizontal bars summarize post hoc analysis results.

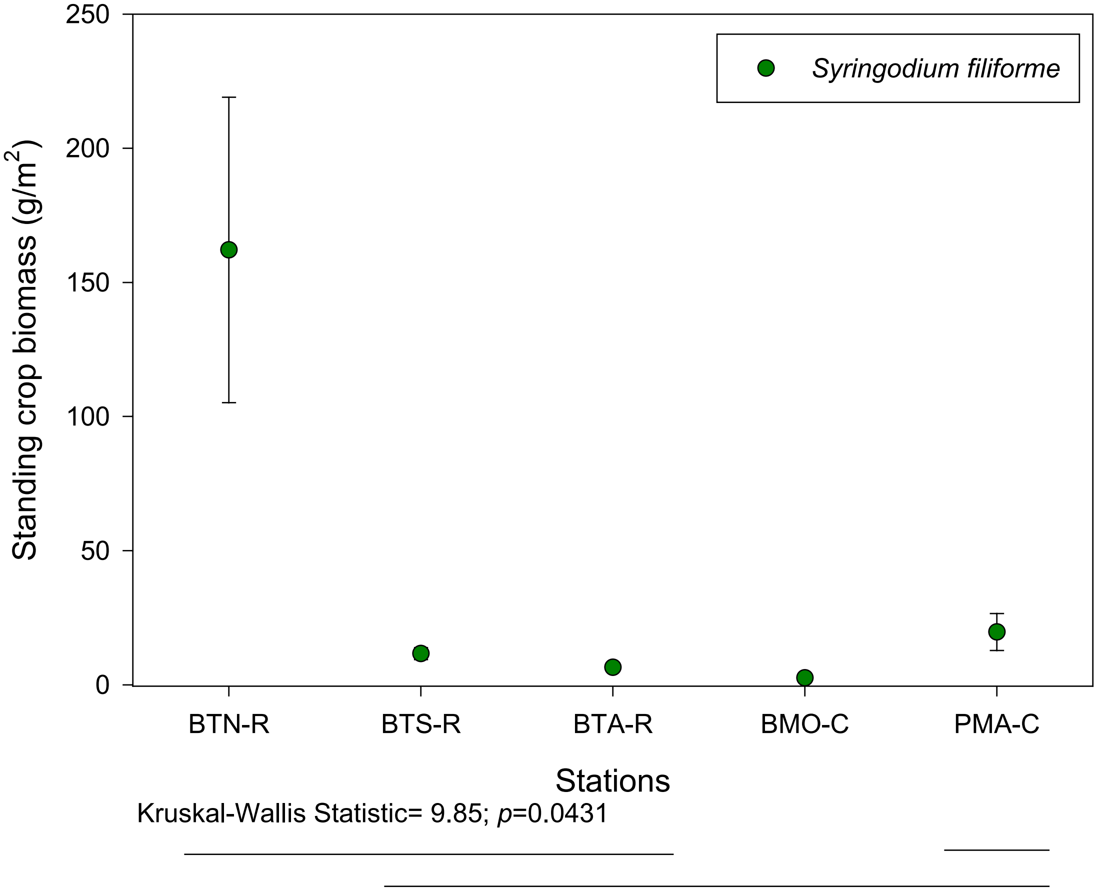
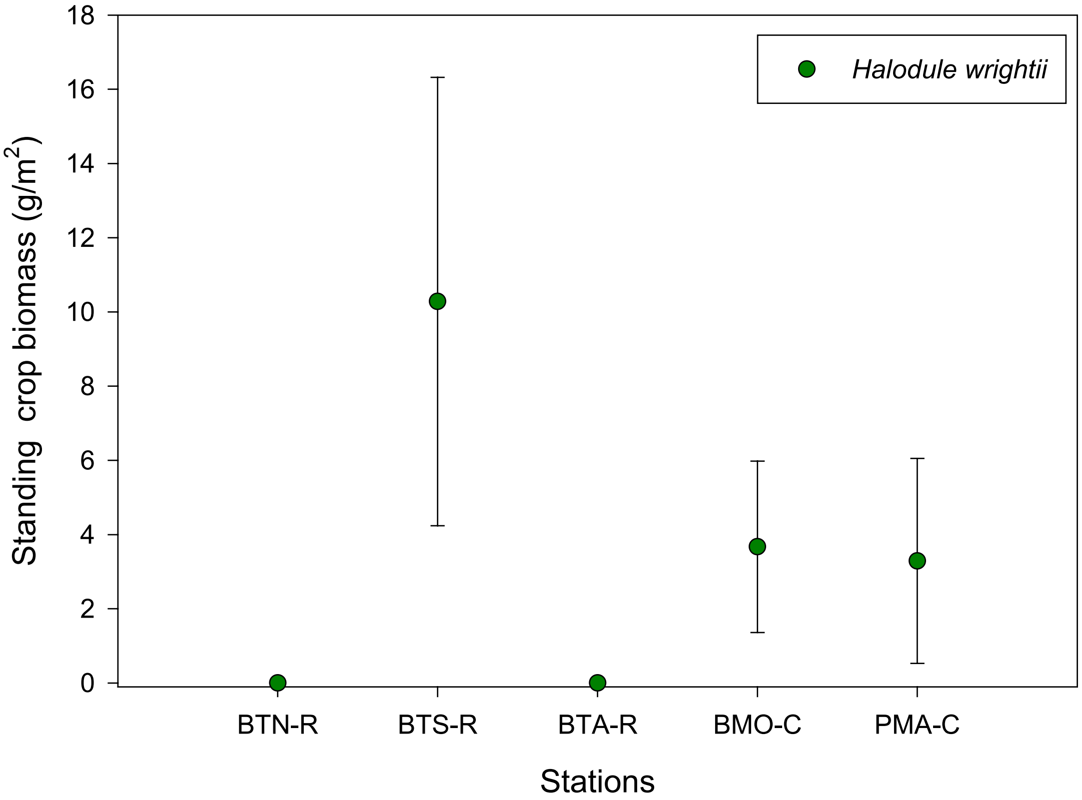


FIGURE 12. *Syringodium filiforme* standing crop biomass (g/m^2) (mean±one standard error). Horizontal bars summarize post hoc analysis results.



Kruskal-Wallis Statistic= 7.71; *p*=0.1028

FIGURE 13. *Halodule wrightii* standing crop biomass (g/m^2) (mean±one standard error). Horizontal bars summarize post hoc analysis results.

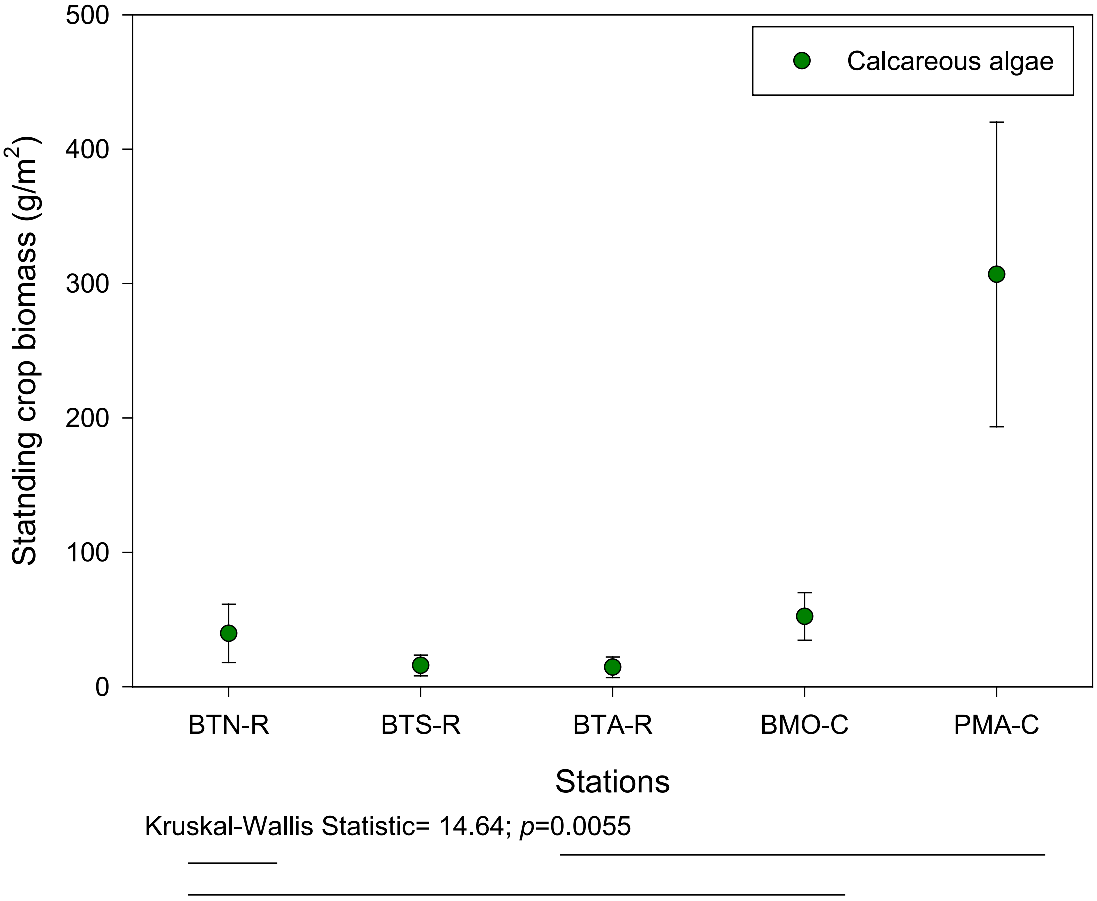


FIGURE 14. Calcareous algae standing crop biomass (g/m^2) (mean±one standard error). Horizontal bars summarize post hoc analysis results.

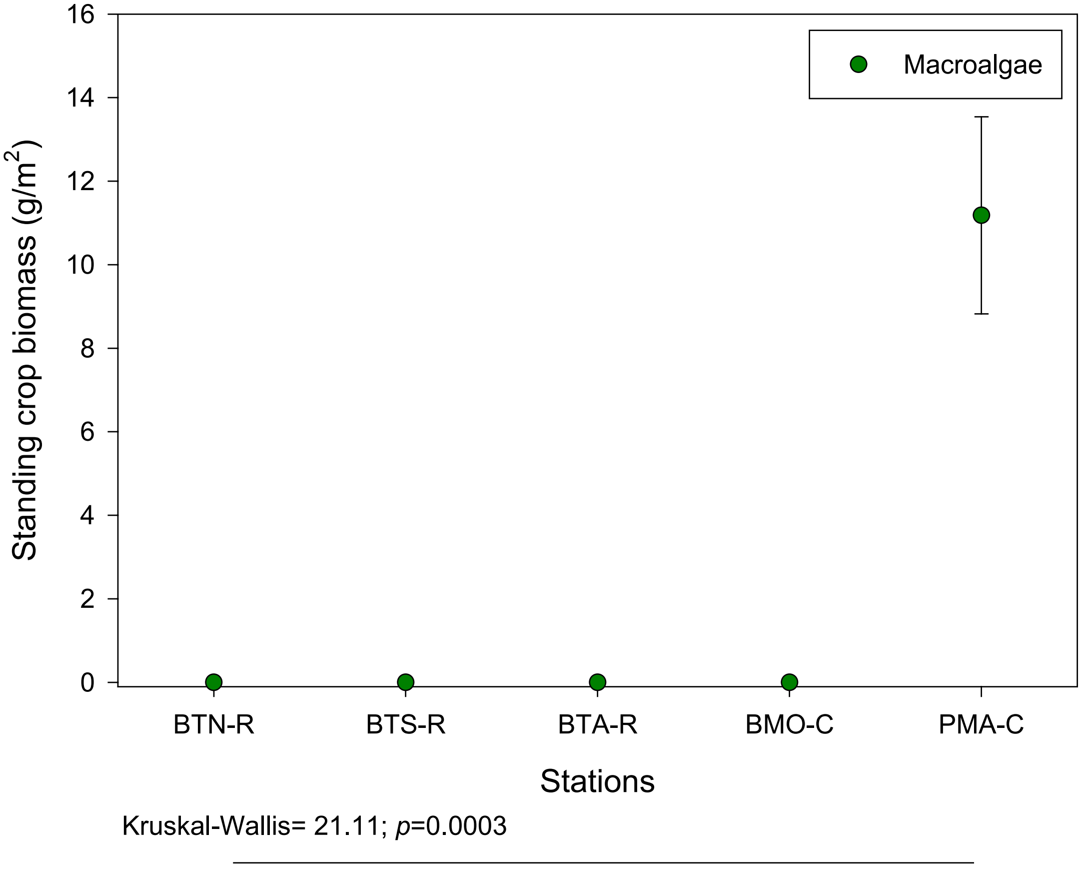


FIGURE 15. Macroalgal standing crop biomass (g/m^2) (mean±one standard error). Horizontal bars summarize post hoc analysis results.

power of the latter three tests ranged only between a weak 31 and 55% (Appendix, Table A1).

Thalassia testudinum parameters.

The *T. testudinum* mean LAI (Figure 16) was significantly different among sites (one-way ANOVA; *p*=0.0006) (Table 5). Mean data was summarized in Table 6. The highest LAI value was documented at BTMN-R (1.62 m^2/m^2) and the lowest at PMA-C (0.40 m^2/m^2). A Bonferonni comparison of means analysis (*p*=0.05) showed that BTMN-R was significantly different than BTA-R, BMO-C and PMA-C. Also, that BTMS-R was significantly different than PMA-C. The was a tendency towards declining LAI as we moved from the core seagrass habitat of the LPCNR (BTMN-R) towards the farthest control site PMA-C. The mean statistical power of this test was 79%, a marginally-high value (Appendix, Table 2).

Mean leaf length in *T. testudinum* was significantly different (one-way ANOVA; *p*<0.0001) among sites (Table 5). Mean data was summarized in Table 6. The highest mean values were obtained from BTMS-R (164 mm), followed by BTMN-R (137 mm). The lowest mean value was registered at BTA-R (96 mm) (Figure 17). A Bonferonni comparison of means analysis (*p*=0.05) showed that BTMS-R was significantly different than BMO-C, PMA-C and BTA-R. Thus, the null hypothesis of no significant differences of *T. testudinum* leaf length was not rejected. The mean statistical power of this test was 93% (Appendix, Table 2). Leaf length size class distribution was summarized in Table 7. The most abundant size class category within the LPCNR was the 151-200 mm category at BTMN-R and at BTMS-R, with the 51-100 mm category being the most abundant at BTA-R (Figure 18). On the control sites, the 101-150 mm category was the most abundant at BMO-C, while the 51-100 mm category was the most common at PMA-C. This size class distribution was partially skewed towards smaller size

TABLE 5. Comparison of *Thalassia testudinum* parameters among sites*.

Parameter	**D.F.**	**SS**	**MS**	**F**	***p***	**Bonferroni Comparison of means (*p*=0.05)**
Leaf length x Site						
Between	4	0.50	0.12	10.09	<0.0001	21
Within	54	0.67	0.01			1453
Total	58	1.16				
Leaf width x Site						
Between	4	0.05	0.01	4.06	0.0060	2415
Within	54	0.18	0.003			4153
Total	58	0.24				
LAI x Site**						
Between	4	6.73	1.68	5.73	0.0006	12
Within	54	15.87	0.29			234
Total	58	22.60				345

*One-Way ANOVA.

** LAI= Leaf Area Index (m^2/m^2).

1=BTMN-R; 2=BTMS-R; 3=BTA-R; 4=BMO-C; 5=PMA-C.

TABLE 6. Summary of *Thalassia testudinum* leaf parameters at each study site.

Site	Leaf length (mm)	Leaf width (mm)	Leaf Area Index
BTMN-R	136.7±12.6	7.1±0.3	1.62±0.45
BTMS-R	163.5±6.4	8.3±0.2	1.05±0.17
BTA-R	95.5±5.6	6.8±0.2	0.64±0.12
BMO-C	107.4±6.5	7.2±0.3	0.63±0.11
PMA-C	110.1±11.7	7.1±0.4	0.40±0.11

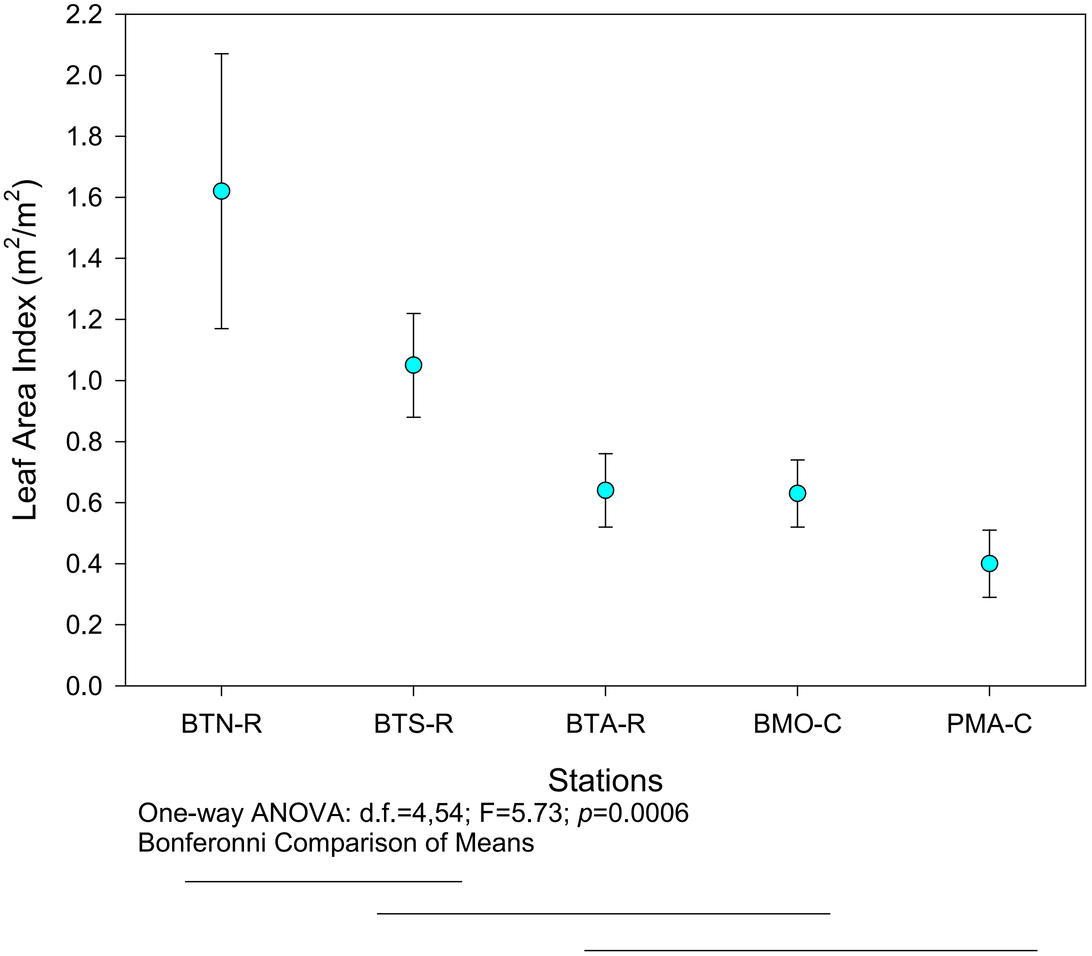
*Mean ± one standard error.

TABLE 7. Size class distribution (leaf length % relative frequency) of *Thalassia testudinum*.

Size class (mm)	**BTMN-R (n=197)**	**BTMS-R (n=375)**	**BTA-R (n=419)**	**BMO-C (n=391)**	**PMA-C (n=214)**
<50	11.79	10.72	23.26	15.94	13.68
51-100	18.97	13.14	30.70	25.96	32.55
101-150	25.13	17.16	28.78	35.99	24.06
151-200	27.18	23.06	14.87	16.71	19.81
201-250	9.23	23.06	2.40	4.63	7.08
251-300	5.13	8.85	0	0.77	2.83
>300	2.56	4.02	0	0	0

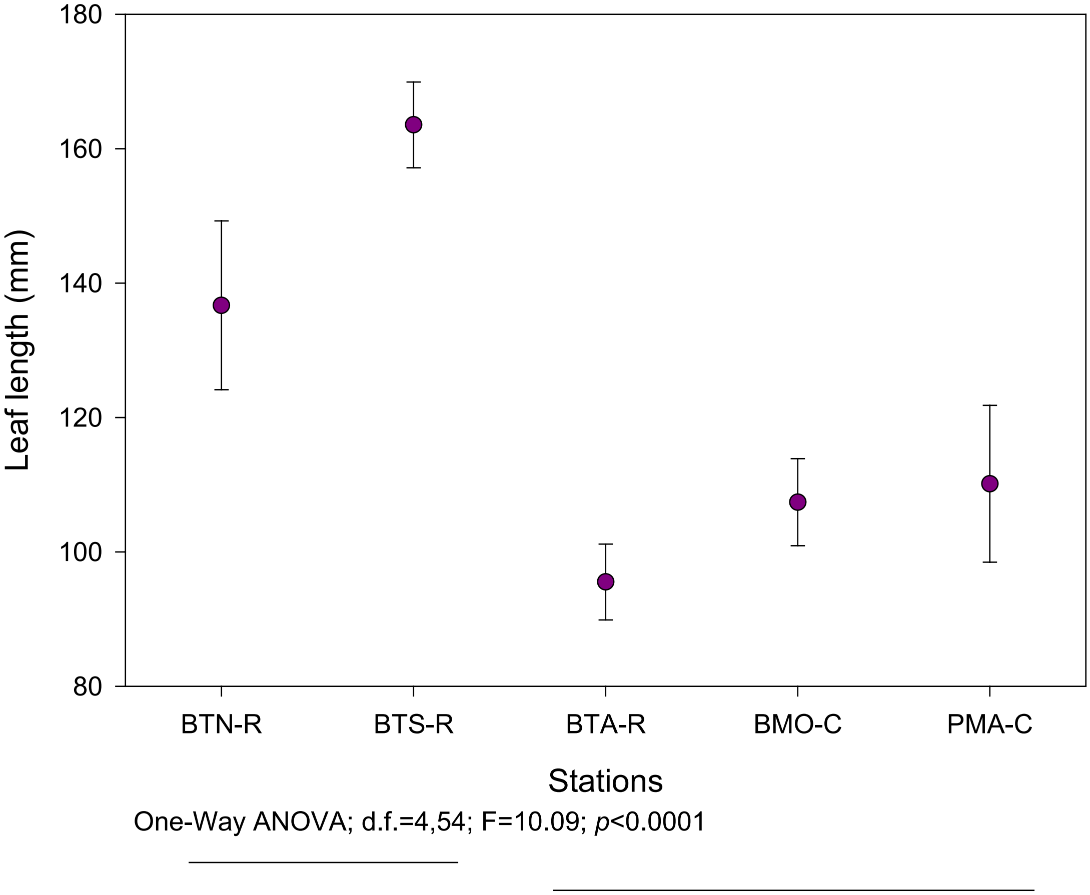
TABLE 8. Size class distribution (leaf width % relative frequency) of *Thalassia testudinum*.

Size class (mm)	**BTMN-R (n=197)**	**BTMS-R (n=375)**	**BTA-R (n=419)**	**BMO-C (n=391)**	**PMA-C (n=214)**
<2	1.03	0.27	0.72	0	0
3-4	5.13	0.80	5.76	4.11	8.49
5-6	28.71	7.51	31.89	29.56	26.42
7-8	44.62	45.58	52.28	44.99	35.85
9-10	18.46	43.97	9.35	17.99	26.89
11-12	2.05	1.88	0	2.57	2.36
>12	0	0	0	0.77	0



One-way ANOVA: d.f.=4,54; F=5.73; *p*=0.0006
Bonferonni Comparison of Means

FIGURE 16. *Thalassia testudinum* Leaf Area Index (m^2/m^2). Mean±one standard error. Horizontal bars summarize post hoc analysis results.



One-Way ANOVA; d.f.=4,54; F=10.09; $p<0.0001$

FIGURE 17. *Thalassia testudinum* leaf length (mm). Mean±one standard error. Horizontal bars summarize post hoc analysis results.

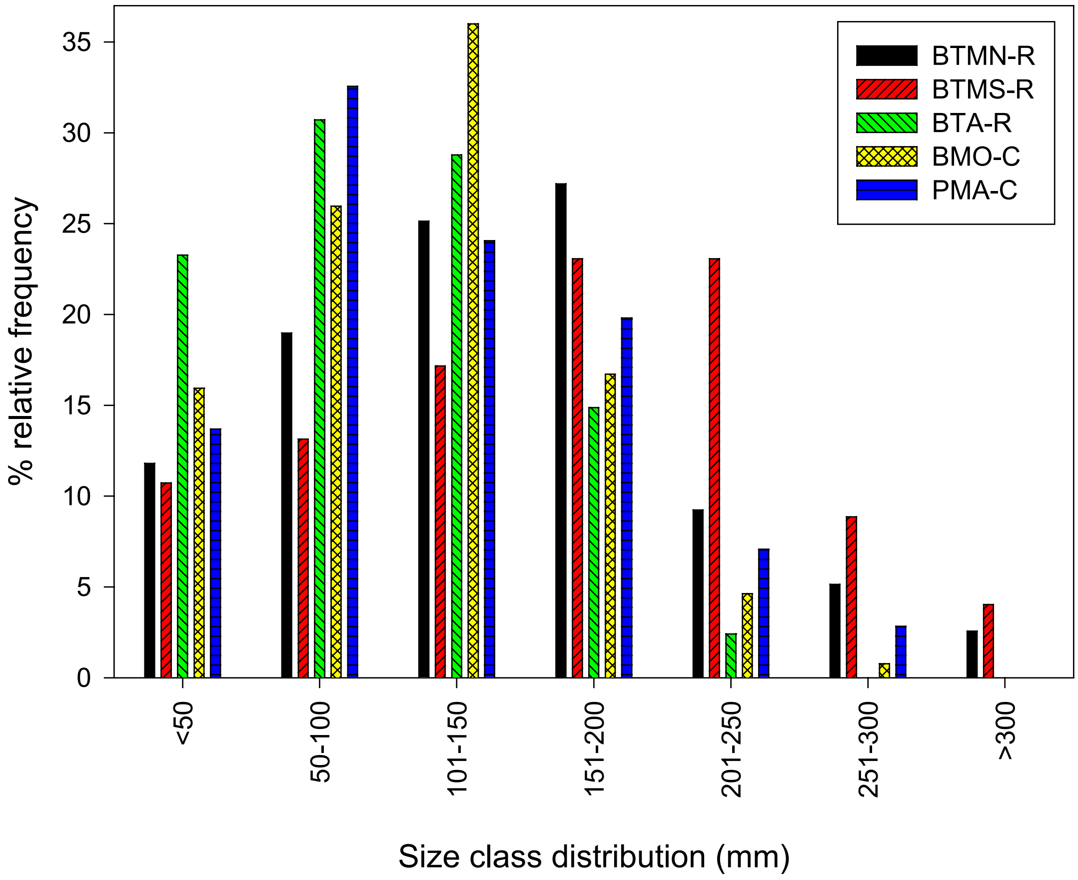


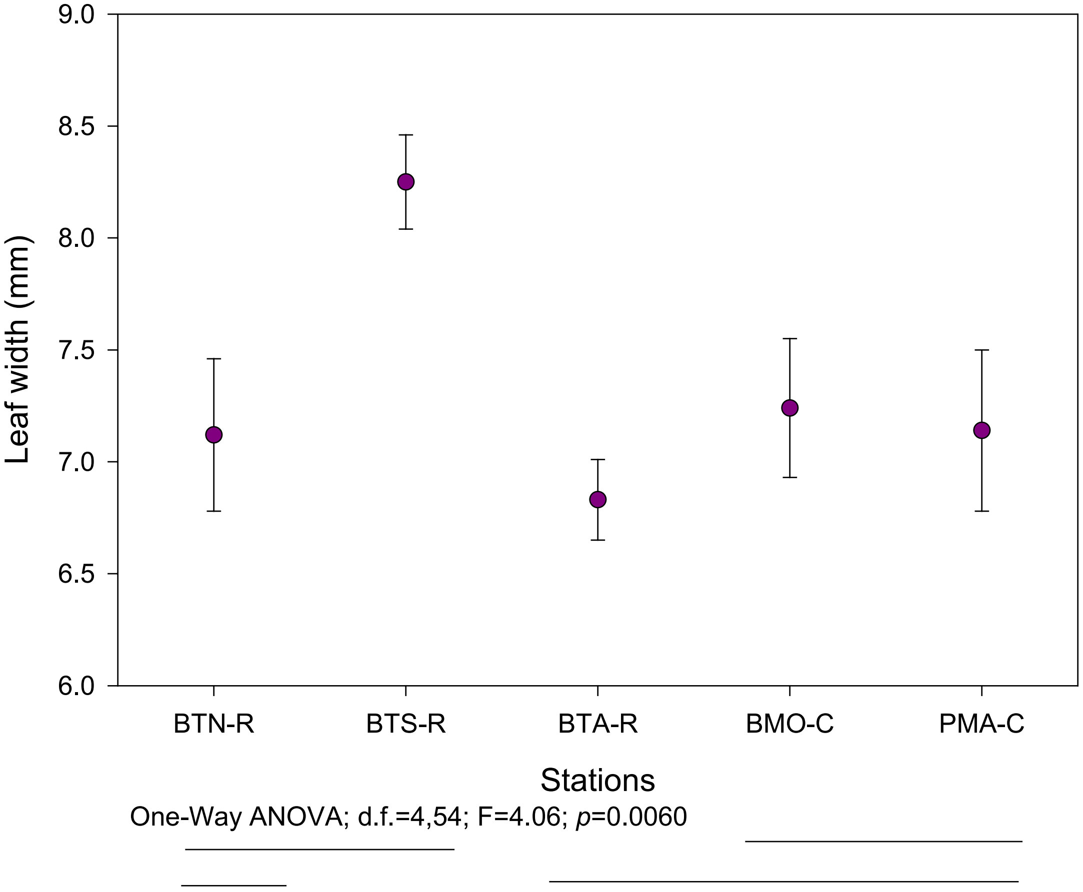
FIGURE 18. *Thalassia testudinum* size class distribution based on leaf length (mm).

classes.

Mean leaf width in *Thalassia testudinum* was significantly different (one-way ANOVA; *p*=0.0060) among sites (Table 5). Mean data was summarized in Table 6. BTMS-R showed the highest mean value (8.3 mm), followed by BMO-C (7.2 mm). The lowest mean value was observed at BTA-R (6.8 mm) (Figure 19). A Bonferonni comparison of means analysis (*p*=0.05) showed that BTMS-R was significantly different than BTA-R, but not from the remaining sites. The null hypothesis of no significant differences of *T. testudinum* leaf width was not rejected. The mean statistical power of this test was 96% (Appendix, Table 2). Leaf width size class distribution was summarized in Table 8. The most abundant size class category within the three LPCNR study sites was the 7-8 mm category (Figure 20). A similar pattern was observed at the control sites. These size classes were normally distributed.

Multivariate analysis of seagrass bed communities.

A cluster analysis was carried out based on a Bray-Curtis dissimilarity matrix on the proportion of major benthic components to characterize the structure of the seagrass bed communities at the five study sites (Figure 21). However, the classification of sites based on broad categorical data did not clearly differentiate sites by geographic distribution. However, geographically-based clusters were more clearly distinguished in the MDS ordination (Figure 22). The global ANOSIM-1 test (Table 9) revealed a highly significant difference (0%) of the seagrass bed community structure among sites. The pairwise ANOSIM-1 test (Table 10) revealed highly significant differences in the seagrass bed community structures within LPCNR sites, within control sites, and among LPCNR and control sites. The only exception was the lack of significant difference (9.3%) between BTMN-R and BMO-C. The results of the



One-Way ANOVA; d.f.=4,54; F=4.06; p=0.0060

FIGURE 19. *Thalassia testudinum* leaf width (mm). Mean±one standard error. Horizontal bars summarize post hoc analysis results.

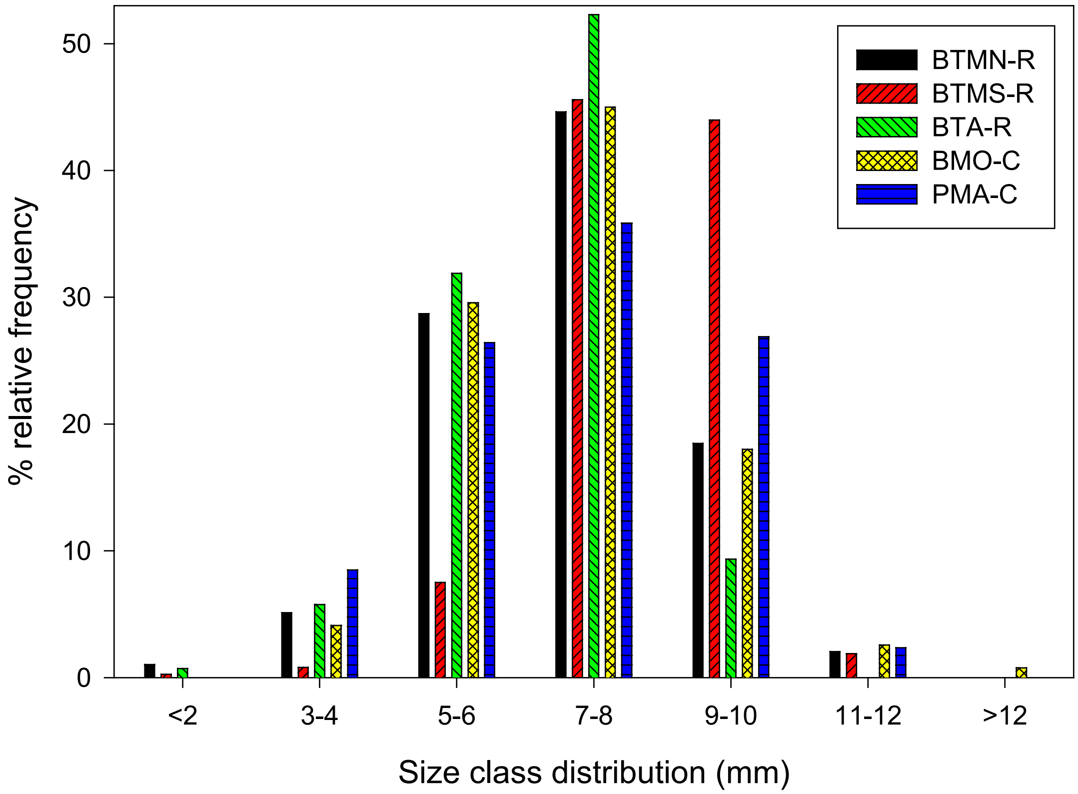


FIGURE 20. *Thalassia testudinum* size class distribution based on leaf width (mm).

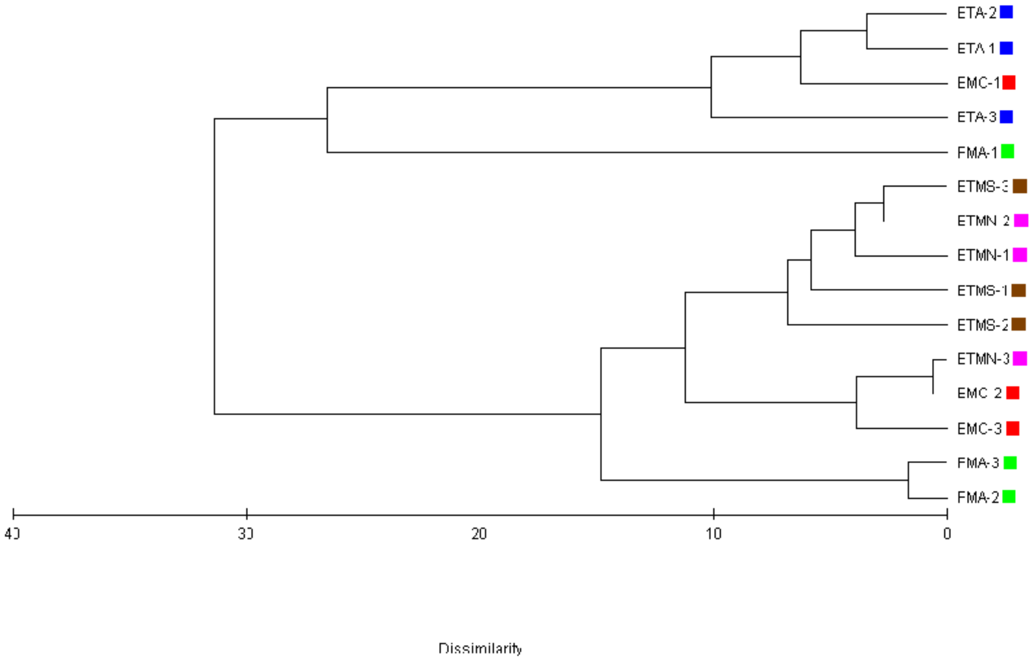


FIGURE 21. Cluster analysis of the community structure of seagrass beds based on the relative proportion of major benthic categories.

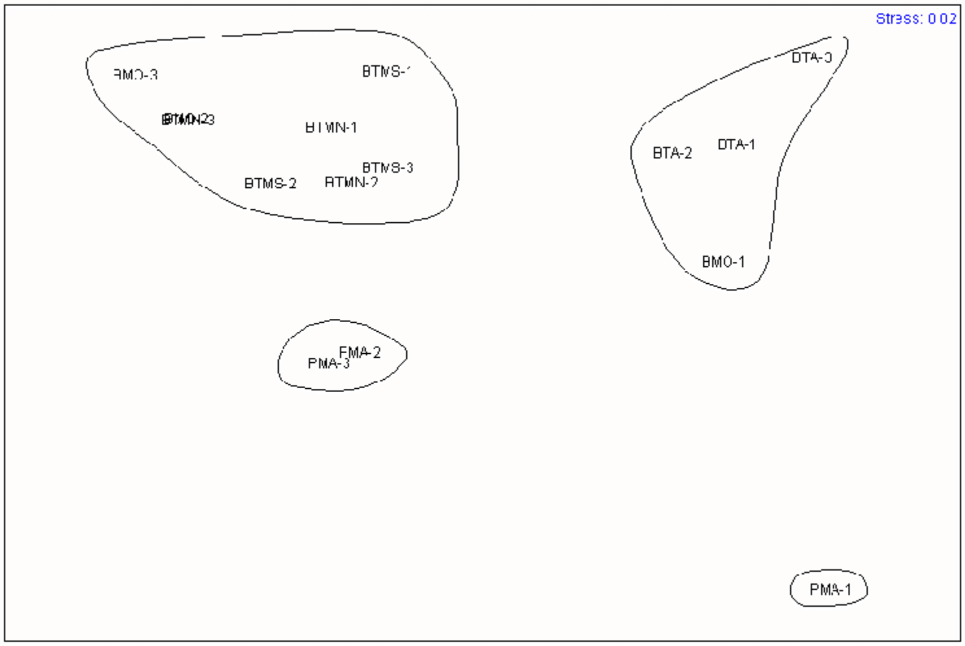


FIGURE 22. Multi-Dimensional Scaling (MDS) analysis plot of the community structure of seagrass beds based on the relative proportion of major benthic categories (Stress level =0.02).

TABLE 9. Results of the global ANOSIM-1 multivariable analysis of the seagrass community based on the proportion of major epibenthic categories*.

Factor	Global R value	Significance level
Sites	0.402	0%

*Square root-transformed data; based on 5,000 permutations.

TABLE 10. Results of the pairwise ANOSIM-1 multivariable analysis of the seagrass community based on the proportion of major epibenthic categories*.

Factor	**Global R value**	**Significance level**
BTMN-R vs. BTMS-R	0.121	2.4%
BTMN-R vs. BTA-R	0.979	0%
BTMN-R vs. BMO-C	0.048	9.3% (N.S.)**
BTMN-R vs. PMA-C	0.417	0%
BTMS-R vs BTA-R	0.847	0%
BTMS-R vs. BMO-C	0.083	4.2%
BTMS-R vs. PMA-C	0.255	0%
BTA-R vs. BMO-C	0.566	0%
BTA-R vs. PMA-C	0.642	0%
BMO-C vs. PMA-C	0.147	0.8%

*Square root-transformed data; based on 5,000 permutations.

**N.S.=Not Significant

TABLE 11. Results of the SIMPER multivariable analysis of the seagrass community based on the proportion of major epibenthic categories*.

Factors (Sites) 1 vs 2	**Benthic category**	**Percentage contribution**	**Abundance Site 1**	**Abundance Site2**	**Percentage dissimiliratiy**
BTMN-R vs. BTMS-R	SG**	46.07	0.97	0.93	6.20
	OTH	37.41	0.01	0.04	
	ALG	16.52	0.01	0.02	
BTMN-R vs. BTA-R	SG	49.22	0.97	0.57	40.74
	OTH	47.86	0.01	0.40	
	ALG	2.92	0.01	0.03	
BTMN-R vs. BMO-C	SG	55.04	0.97	0.85	14.64
	OTH	29.66	0.01	0.8	
	ALG	15.29	0.01	0.4	
BTMN-R vs. PMA-C	SG	49.51	0.97	0.66	32.34
	ALG	39.87	0.01	0.26	
	OTH	10.61	0.01	0.06	
BTMS-R vs BTA-R	SG	48.76	0.93	0.57	37.03
	OTH	48.22	0.04	0.40	
	ALG	3.01	0.02	0.03	
BTMS-R vs. BMO-C	SG	53.68	0.93	0.85	15.82
	OTH	32.11	0.04	0.08	
	ALG	14.20	0.02	0.04	
BTMS-R vs. PMA-C	SG	47.80	0.93	0.66	30.99
	ALG	38.81	0.02	0.26	
	OTH	13.39	0.04	0.06	
BTA-R vs. BMO-C	OTH	47.49	0.40	0.08	35.49
	SG	46.08	0.57	0.85	
	ALG	6.03	0.03	0.04	
BTA-R vs. PMA-C	OTH	37.85	0.40	0.06	45.84
	SG	36.43	0.57	0.66	
	ALG	25.72	0.03	0.26	
BMO-C vs. PMA-C	SG	49.22	0.85	0.66	34.73
	ALG	34.36	0.04	0.26	
	OTH	16.42	0.08	0.06	

*Square root-transformed data; based on 5,000 permutations.

**SG= Seagrasses; ALG= Calcareous algae; OTH= Other epibenthic components.

SIMPER analysis (Table 11) revealed that the % of relative seagrass cover was the most important benthic component for discriminating among all groups of sites in 8 out of the 10 comparisons of sites. The other components category was slightly more important than the seagrass beds in the remaining 2 cases.

Conch densities and size class distribution.

There were no significant differences in the status of Conch populations within and outside the LPCNR (Table 12). Only a total of 71 Conch were counted during the present study, which surveyed a total of 94 replicate 200 m^2 belt transects (Table 13). These figures included 58 individuals of *Strombus pugilis*, which included an aggregation of 50 individuals at BMO-C. The remaining Conch individuals included 12 *S. gigas*, 8 of them at PMA-C, and only a single individual of *S. costatus* at BMO-C. Kruskal-Wallis analysis suggested that there were significant differences in the density of *S. gigas* and in *S. pugilis* among sites. But a comparison of means analysis showed that there were no differences. Table 14 summarized the mean data for each site. The highest density of *S. gigas* was documented at BTMN-R (18.8/ha), followed by PMA-C (16.6/ha) (Figure 23). The highest density of *S. pugilis* was obtained from BMO-C (156/ha), followed by BTMS-R (14/ha) (Figure 24). *Strombus costatus* was only present at BMO-C at a density of 3.1/ha (Figure 25).

Maximum shell length (Table 12, Figure 26) and lip width (Table 12, Figure 27) was not significantly different among sites. Maximum shell length size class distribution (Figure 28) was skewed towards smaller size classes, (50-100 mm, 100-150 mm). Larger *Strombus gigas* individuals were only observed at PMA-C (250-300 mm). The highest mean maximum shell

TABLE 12. Comparison of Conch parameters among sites*.

Parameter	**D.F.**	**Kruskal-Wallis Statistic**	***p***	**Comparison of means (*p*=0.05)**
Density *Strombus gigas***	4,89	14.86	0.0050	15234 (N.S.)***
Density *Strombus pugilis*	4,89	27.88	<0.0001	42315 (N.S.)
Density *Strombus costatus*	4,89	4.88	0.3004	41235 (N.S.)
Max. length *Strombus gigas*	3,7	6.21	0.1018	5132 (N.S.)
Max. length *Strombus pugilis*	3,12	4.58	0.2056	3241 (N.S.)
Max. length *Strombus costatus*	-	-	-	-
Lip width *Strombus gigas*	3,7	1.74	0.6278	1523 (N.S.)
Lip width *Strombus pugilis*	3,12	6.51	0.0891	3412 (N.S.)
Lip width *Strombus costatus*	-	-	-	-

*Kruskal-Wallis non-parametric ANOVA.

** Log-transformed data. Only 1 individual of *S.costatus* prevented comparisons of conch size and lip width among sites. Density (#/ha); Max. length (mm); Lip width (mm).

***N.S.= Not Significant.

1=BTMN-R; 2=BTMS-R; 3=BTA-R; 4=BMO-C; 5=PMA-C.

TABLE 13. Conch abundance per site.

Parameter	**BTMN-R**	**BTMS-R**	**BTA-R**	**BMO-C**	**PMA-C**	**# Total**
Strombus gigas	1	1	2	0	8	*12*
Strombus pugilis	3	3	2	50	0	*58*
Strombus costatus	0	0	0	1	0	*14*
# Total	*4*	*4*	*4*	*51*	*8*	***71***

TABLE 14. Summary of Conch parameters.

Parameter	**BTMN-R**	**BTMS-R**	**BTA-R**	**BMO-C**	**PMA-C**
Density *Strombus gigas**	18.8±6.3	4.6±4.6	3.9±3.9	0±0	16.0±4.8
Density *Strombus pugilis*	0±0	13.6±7.0	3.9±2.7	156.3±62.9	0±0
Density *Strombus costatus*	0±0	0±0	0±0	3.1±3.1	0±0
Max. length *Strombus gigas*	145.0±0.0	66.0±0.0	75.5±3.5	0±0	239.4±6.9
Max. length *Strombus pugilis*	131.7±1.7	156.3±12.0	157.5±22.5	139.3±3.1	0±0
Max. length *Strombus costatus*	0±0	0±0	0±0	134.0±0.0	0±0
Lip width *Strombus gigas*	3.0±0.0	0.5±0.0	0.45±0.05	0±0	1.68±0.72
Lip width *Strombus pugilis*	0.47±0.03	0.37±0.09	3.65±2.25	0.48±0.04	0±0
Lip width *Strombus costatus*	0±0	0±0	0±0	0.55±0.00	0±0

*Density (#/ha); Max. length (mm); Lip width (mm); Mean±one standard error.

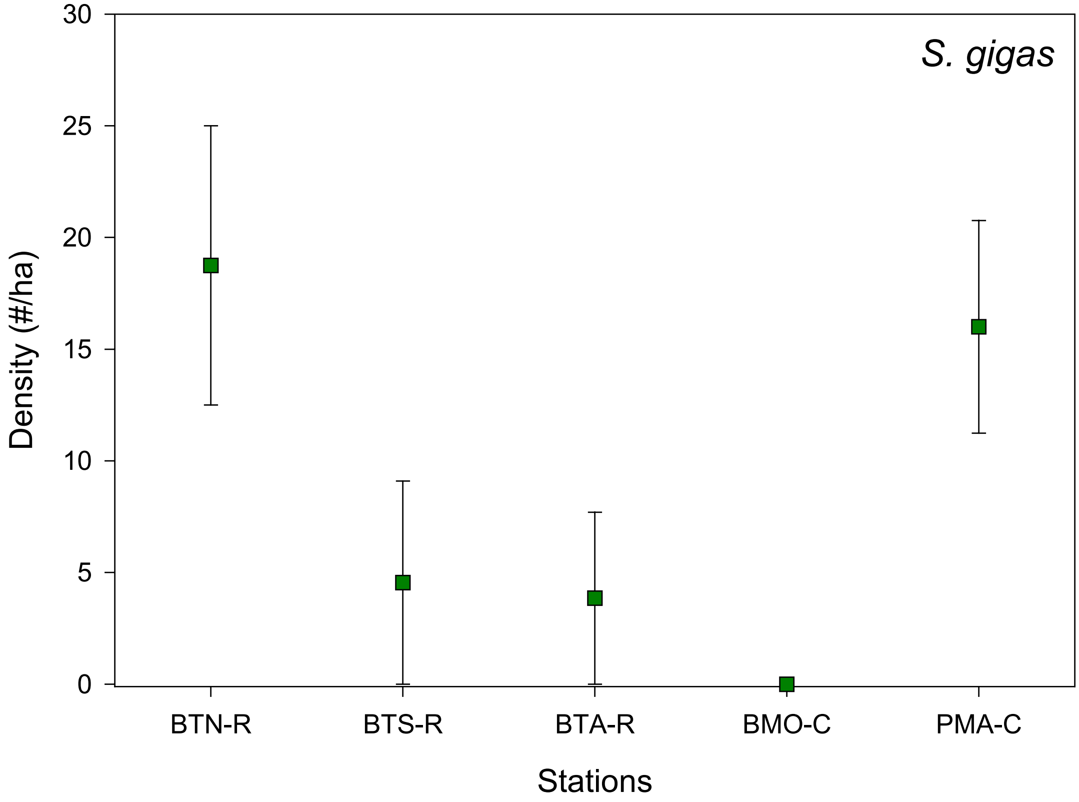
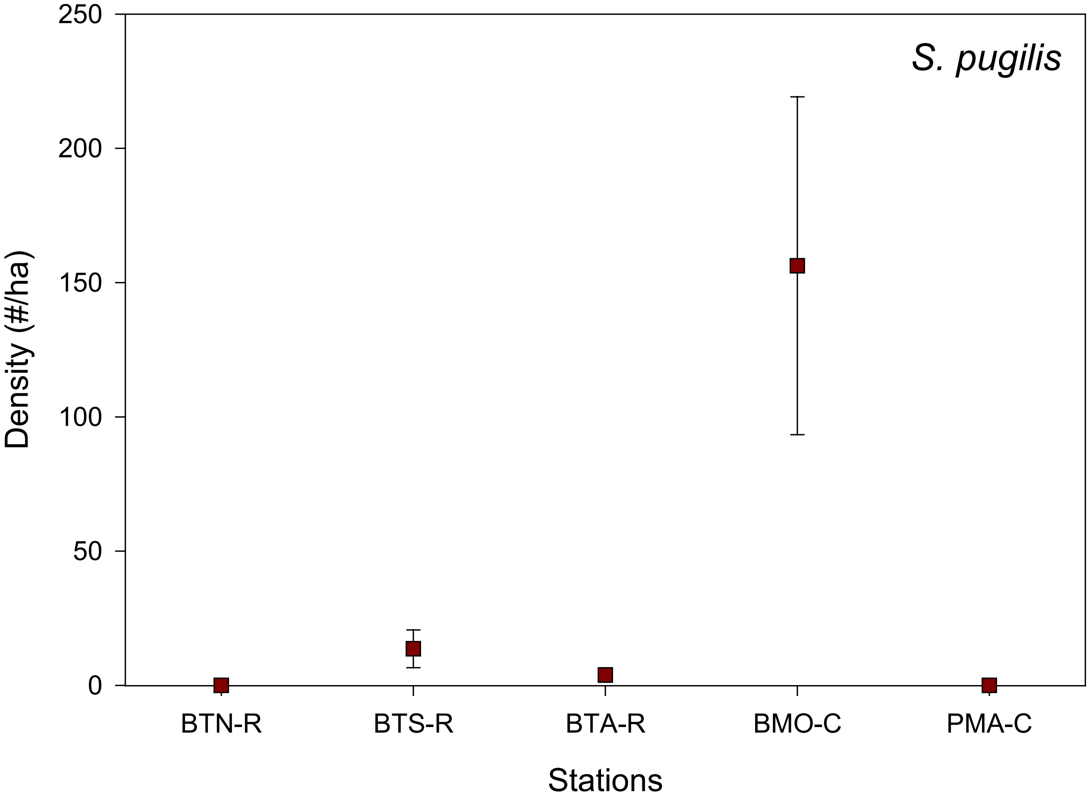


FIGURE 23. Density of *Strombus gigas* (#/ha). Mean±one standard error. Horizontal bars summarize post hoc analysis results.



Kruskal-Wallis Statistic= 27.88; <0.0001

FIGURE 24. Density of *Strombus pugilis* (#/ha). Mean±one standard error. Horizontal bars summarize post hoc analysis results.

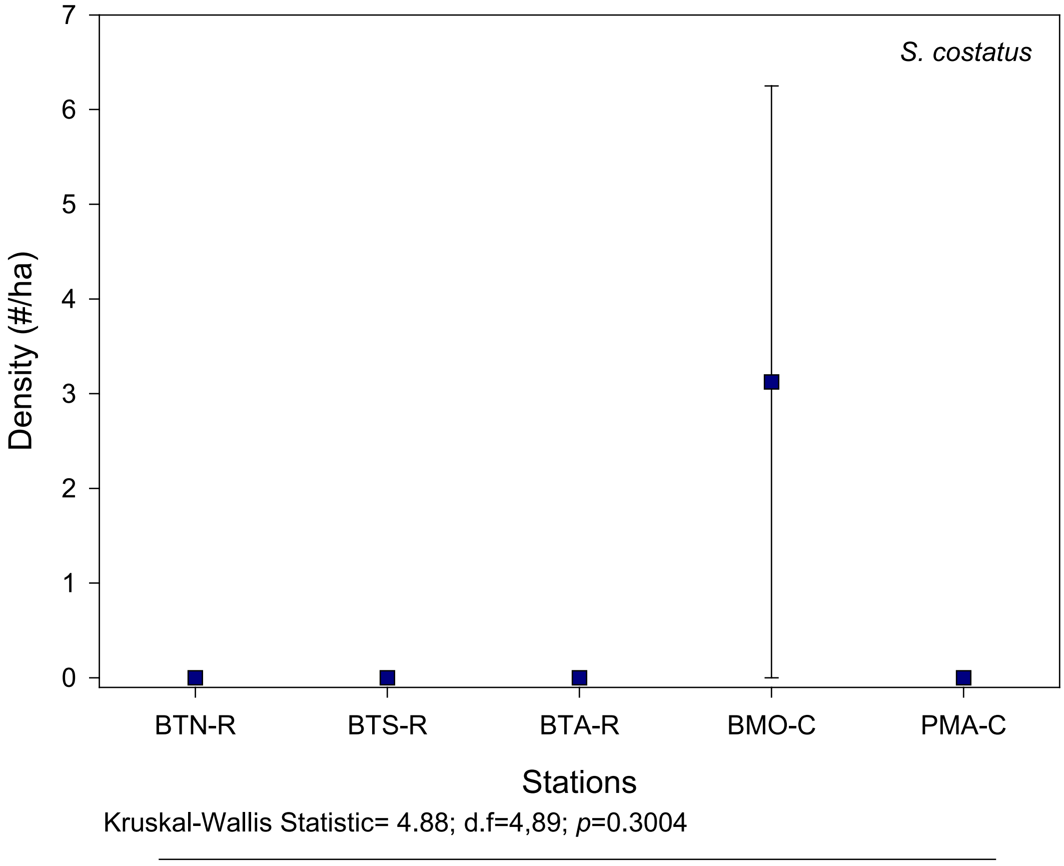
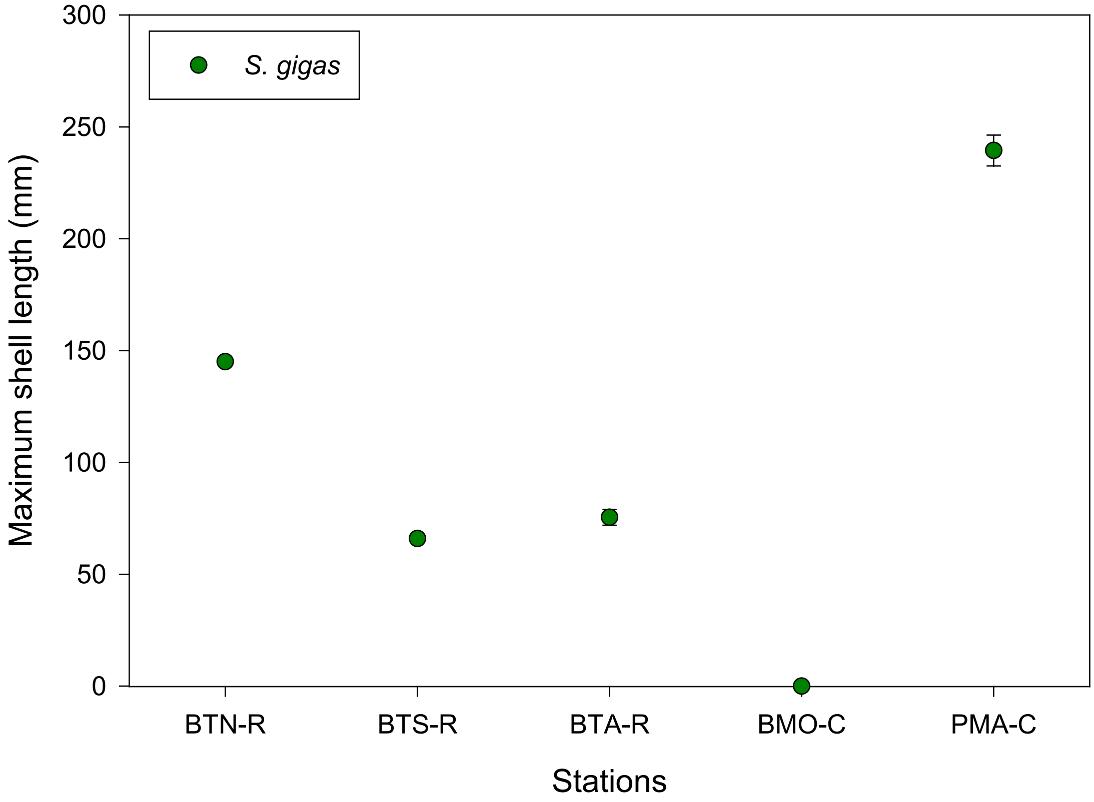


FIGURE 25. Density of *Strombus costatus* (#/ha). Mean±one standard error. Horizontal bars summarize post hoc analysis results.



Kruskal-Wallis Statistic= 6.21; d,f= 3,7; *p*=0.1018

FIGURE 26. Maximum shell length (mm) in *Strombus gigas* (Mean±one standard error). Horizontal bars summarize post hoc analysis results.

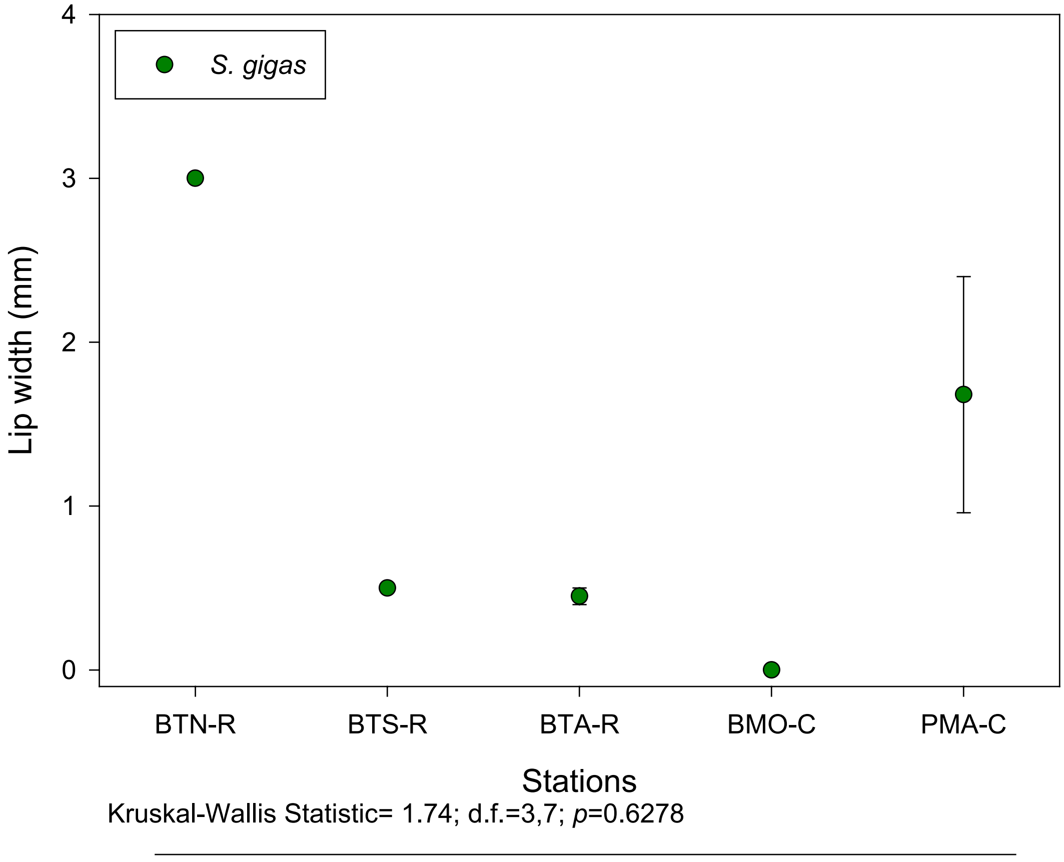


FIGURE 27. Lip width (mm) in *Strombus gigas* (Mean±one standard error). Horizontal bars summarize post hoc analysis results.

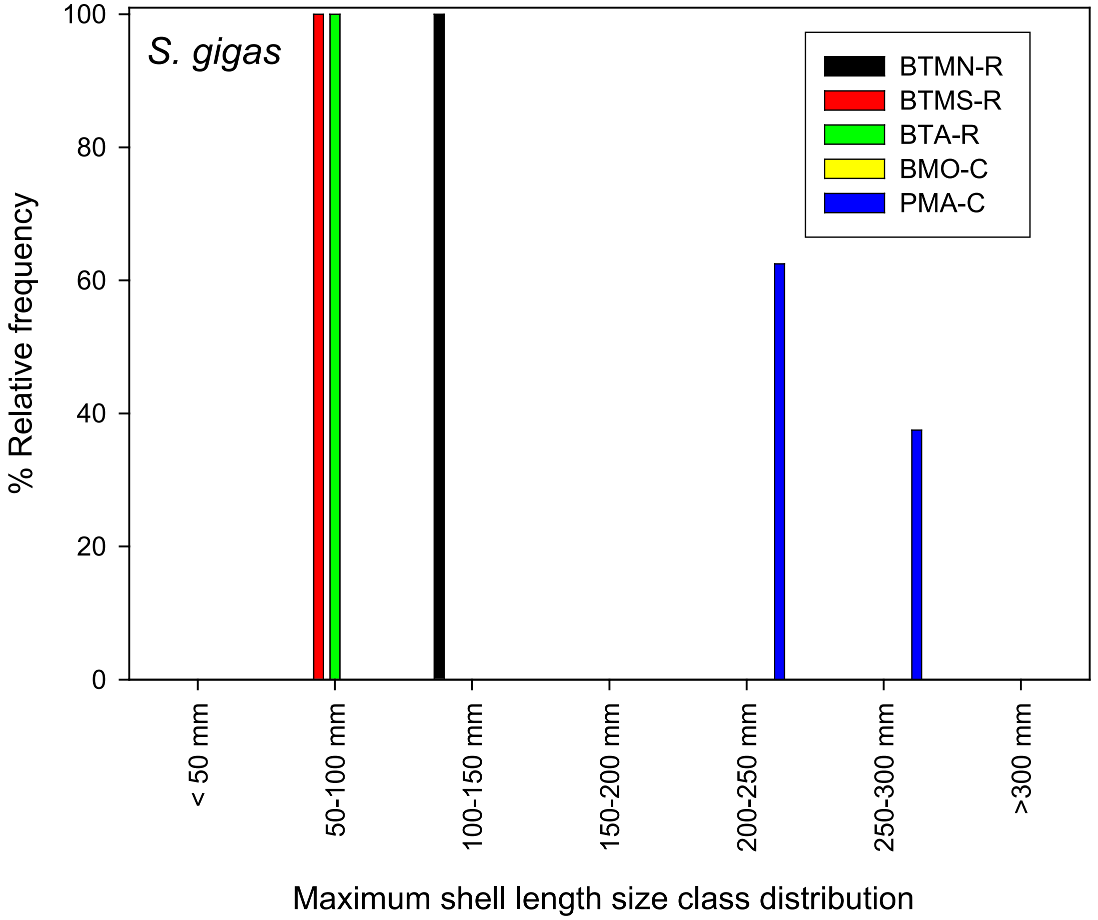


FIGURE 28. Size class distribution of maximum shell length (mm) in *Strombus gigas*.

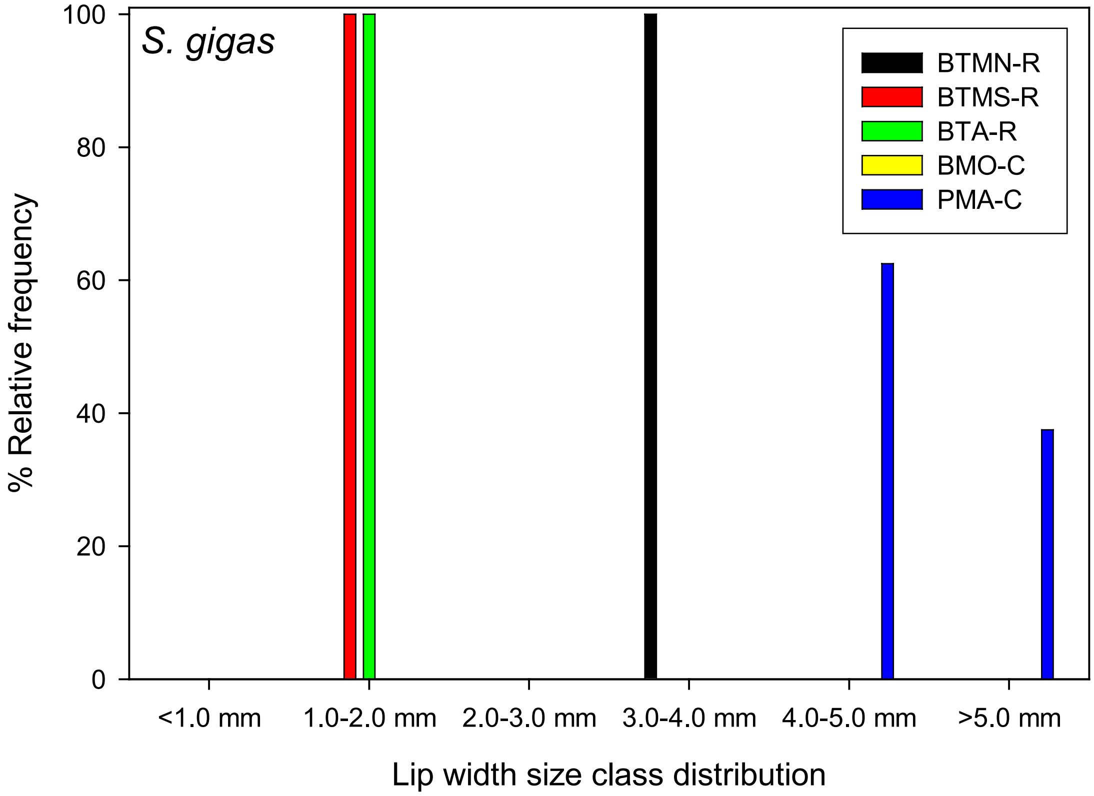
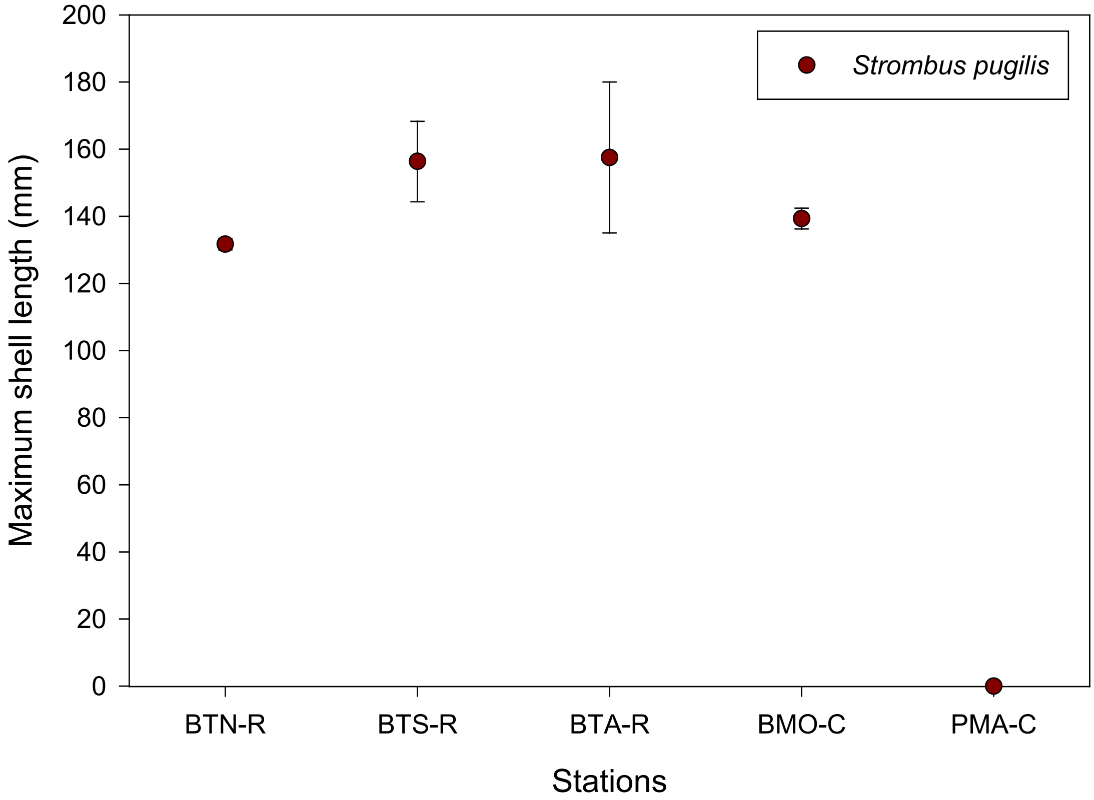


FIGURE 29. Size class distribution of lip width (mm) in *Strombus gigas*.

length was observed at PMA-C (239 mm), and the lowest at BTMS-R (66 mm) (Table 14). Lip width size class distribution (Figure 29) was similarly skewed towards smaller size classes, suggesting that most of the individuals were not sexually mature. The highest mean lip width was observed at BTM-N (3.0 mm), and the lowest at BTA-R (0.45 mm) (Table 14). Similar non-significant results were documented in *S. pugilis* (Table 12, Figures 30 and 31). The highest mean maximum shell length in *S. pugilis* was observed at BTA-R (158 mm), and the lowest at BTMN-R (132 mm). Most of the individuals were distributed in the 140-160 mm category for maximum shell length, with another group in the 120-140 mm and in the 100-120 mm categories (Figure 32). The highest mean lip width in *S. pugilis* was observed at BTA-R (3.65 mm), and the lowest at BTMS-R (0.37 mm). Most of the individuals were distributed in the 0.5-0.7 mm category for lip width (Figure 33). Only a single individual of *S. costatus* was observed at BMO-C (Figures 34-37).

A linear regression analysis ($y= y_o+ ax$) (Zar, 1984) was performed to determine the relationship between Conch maximum shell length and lip width for *S. gigas* and for *S. pugilis*. There were no relationships between both parameters in *S. gigas* ($n=12$; $y= -1.195 + 0.50x$; $r^2= 0.0633$). Similar results were observed for *S. pugilis* ($n= 58$; $y= -2.984 + 1.219x$; $r^2= 0.0809$).

The statistical power of Conch density data was low for the two more common species, *S. gigas* (23%) and *S. pugilis* (52%) (Appendix, Table A3). Maximum shell length data showed a consistent high statistical power for *S. gigas* (96%) and for *S. pugilis* (94%) (Appendix, Table A4). But, it was rather low for lip width data, with 73% and 51%, respectively (Appendix, Table A5).



Kruskal-Wallis Statistic= 4.58; d.f.=3,12; *p*=0.2056

FIGURE 30. Maximum shell length (mm) in *Strombus pugilis* (Mean±one standard error).

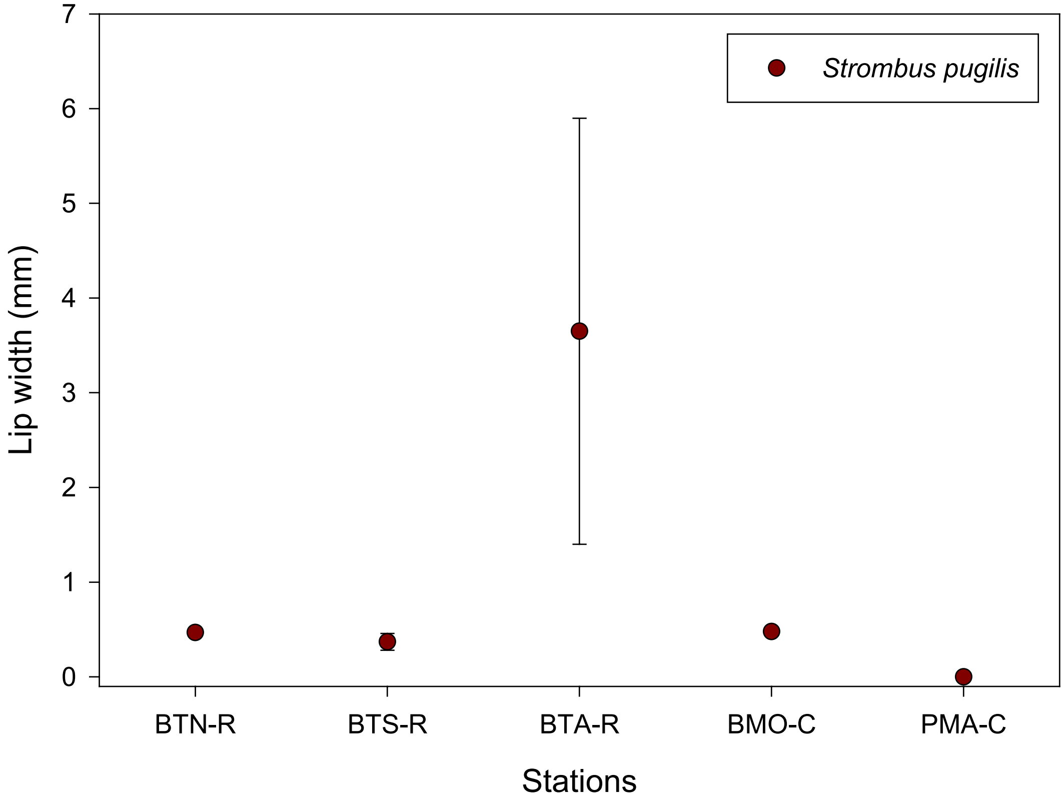


FIGURE 31. Lip width (mm) in *Strombus pugilis* (Mean±one standard error).

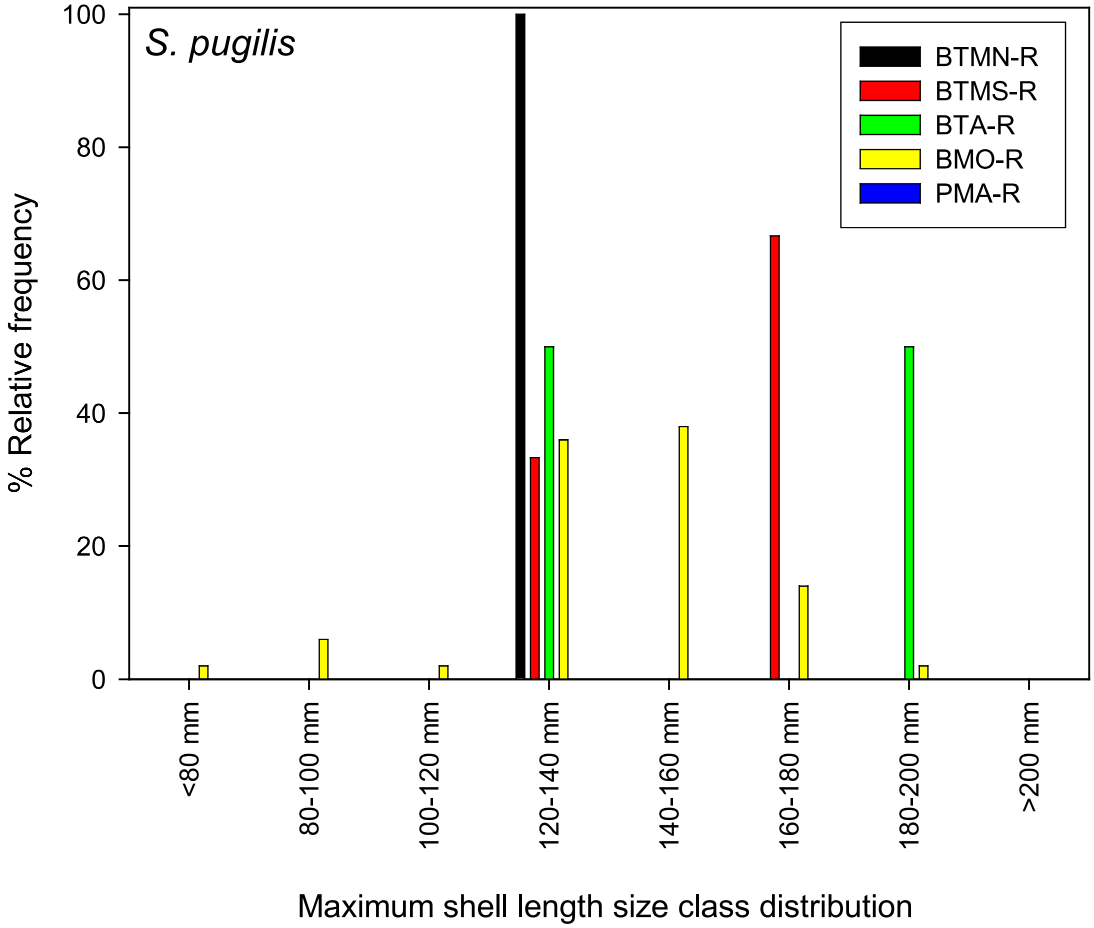


FIGURE 32. Size class distribution of maximum shell length (mm) in *Strombus pugilis*.

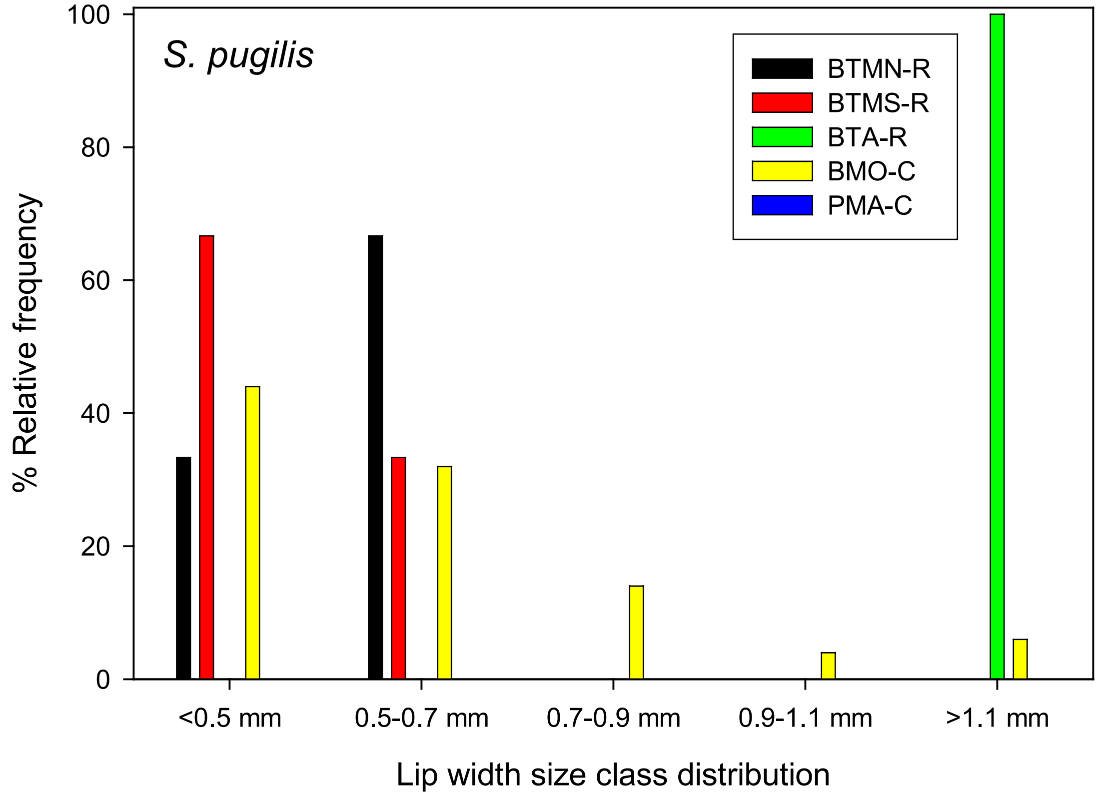


FIGURE 33. Size class distribution of lip width (mm) in *Strombus pugilis*.

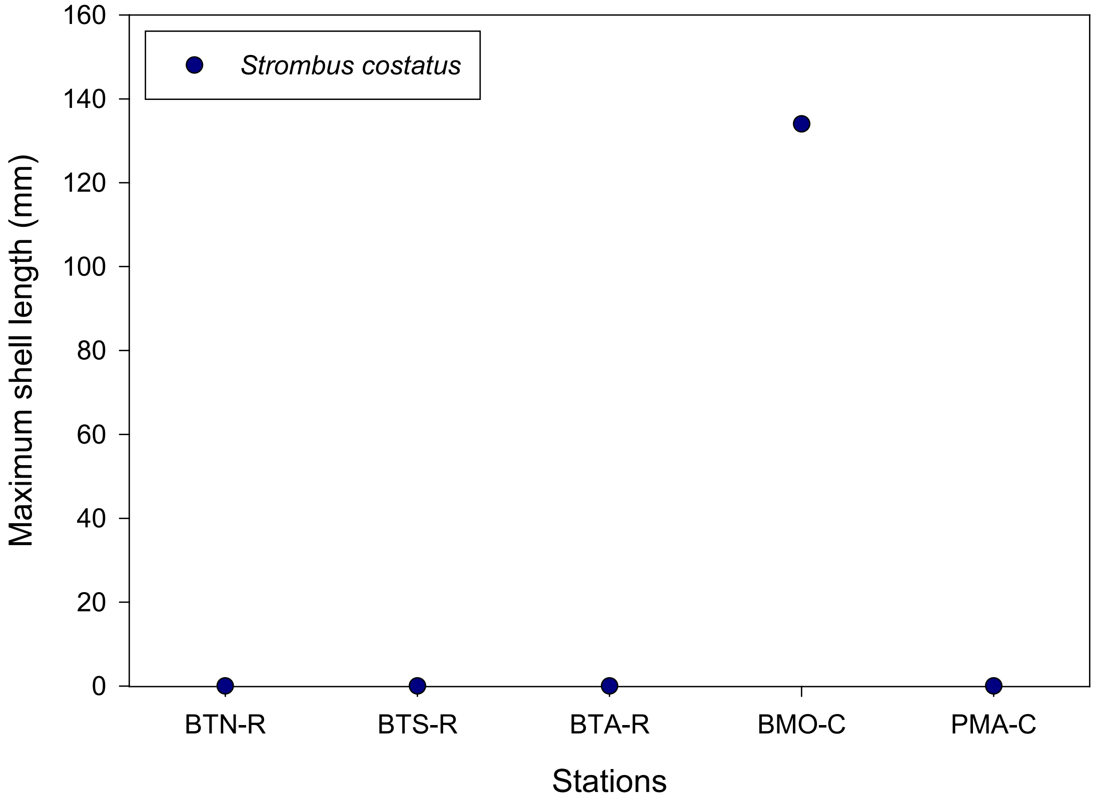


FIGURE 34. Maximum shell length (mm) in *Strombus costatus* (Mean±one standard error).

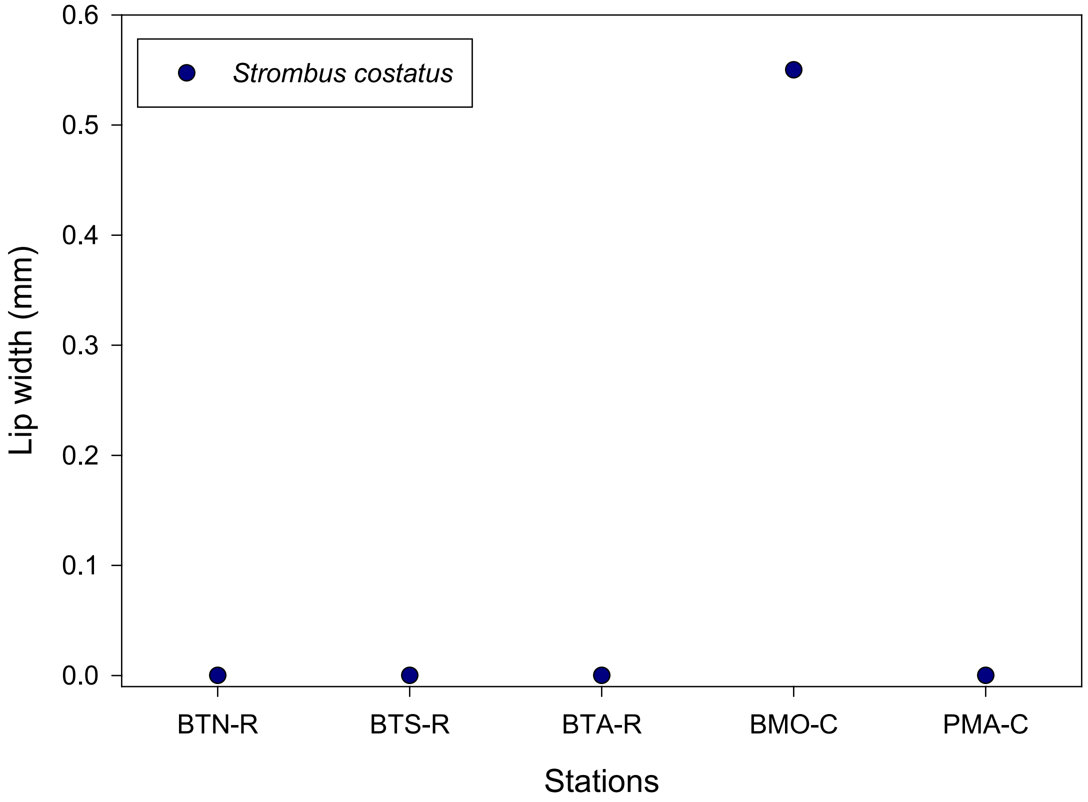


FIGURE 35. Lip width (mm) in *Strombus costatus* (Mean±one standard error).

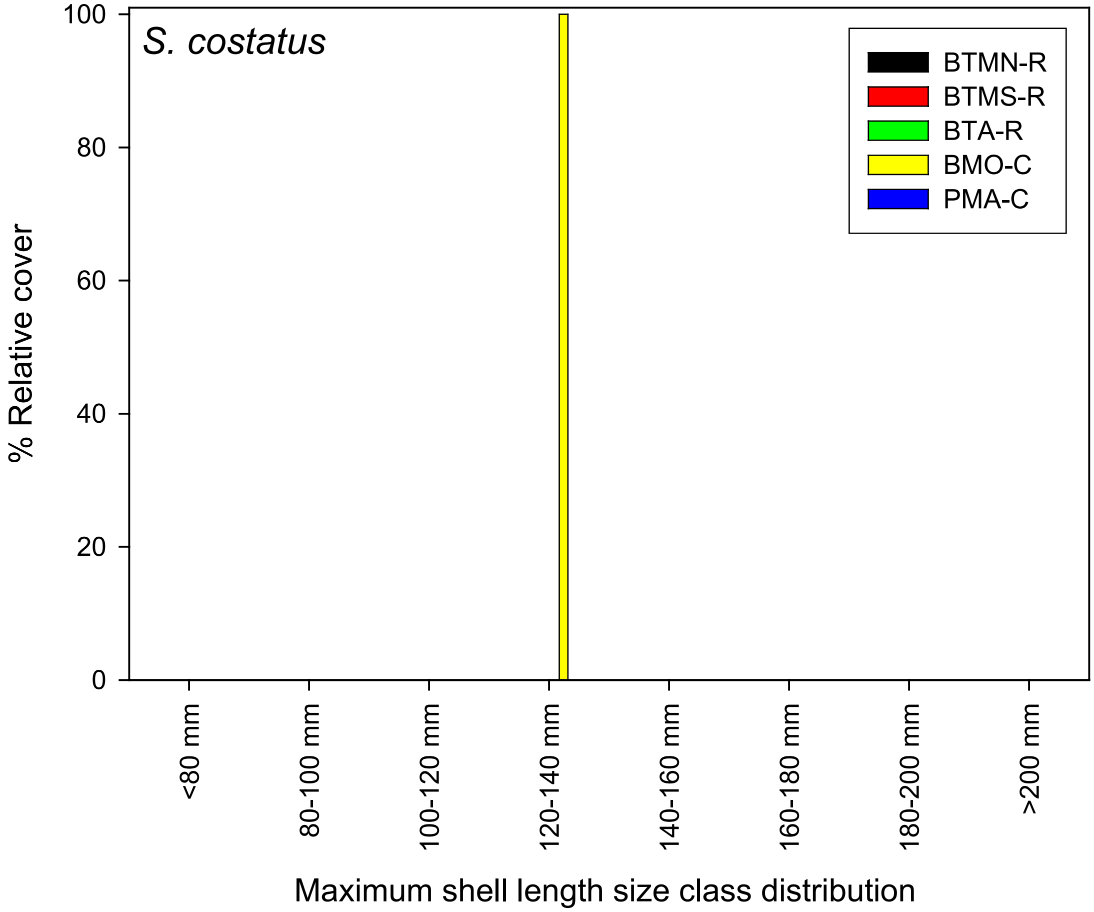


FIGURE 36. Size class distribution of maximum shell length (mm) in *Strombus costatus*.

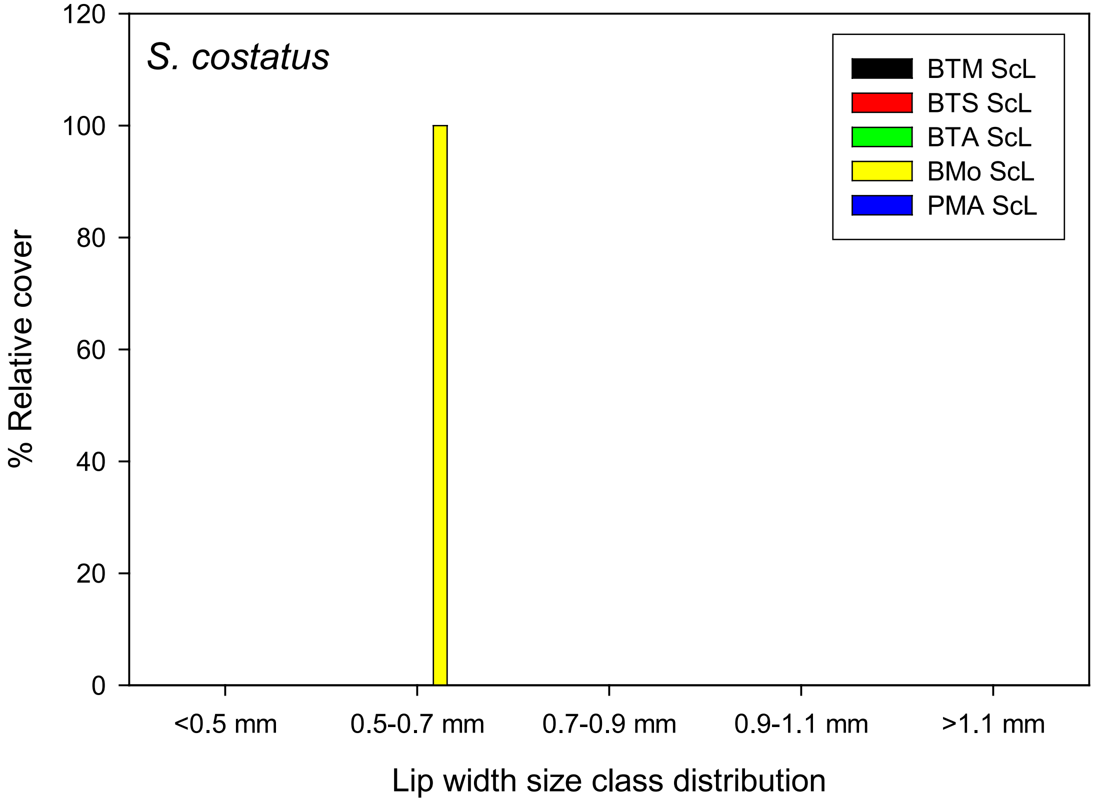


FIGURE 37. Size class distribution of lip width (mm) in *Strombus costatus*.

Status of Conch populations before and after the LPCNR designation.

Conch populations were assessed at BTMS-R (Table 15a) and at BTA-R (Table 15b) during the months of July and August, 1998 by Hernández-Delgado (unpublished data), just approximately a year prior to the LPCNR designation. In the case of BTMS-R (Table 16, Figure 38), there was a significant decline (85%) in time in the density of *Strombus gigas* (Kruskal-Wallis; p=0.0471). A similar pattern was documented in *S. pugilis*, with a 96% decline (Kruskal-Wallis, p<0.0001). But no differences were observed in *S. costatus*. Only a single individual of this species was observed in 1998 and none in 2002. In the case of BTA-R (Table 17, Figure 38), there was also significant decline (90%) in time in the density of *S. gigas* (Kruskal-Wallis; p=0.0013) and a similar pattern in *S. pugilis*, with a 99% decline (Kruskal-Wallis, p<0.0001). *Strombus costatus* was absent from this site during both surveys.

Based on the fact that the potential Conch habitat within the LPCNR has been estimated in 327.3 ha, and that the mean *S. gigas* density prior to the LPCNR designation (1998) was 35.7/ha, then, the potential *S. gigas* stock was estimated in 1998 to be approximately 10,244 individuals within the actual LPCNR. The mean 1998 *S. pugilis* density was 386.9/ha, and its potential stock was estimated to be approximately 126,632 individuals. The mean 1998 *S. costatus* density was 3.15/ha, and its potential stock was estimated to be approximately 1,031 individuals. However, a major decline has occurred in Conch stock densities during the last four years within the LPCNR. Mean densities of *S. gigas* within the LPCNR were estimated in 2002 to be 9.1/ha, with a resulting stock estimate of only 2,978 individuals. Mean densities of *S. pugilis* within the LPCNR were estimated in 2002 to be 5.83/ha, which results in a total stock estimate of only 1,909 individuals.

TABLE 15a. Comparison of Conch densities (#/ha) at BTMS-R before and after the designation of the LPCNR in 1999*.

Species	**1998**	**2002**	**% Change**
Strombus gigas	31.3±13.2	4.6±4.6	**-85%**
Strombus pugilis	343.8±53.8	13.6±7.0	**-96%**
Strombus costatus	6.3±6.3	0±0	**-100%**

Mean±one standard error.

TABLE 15b. Comparison of Conch densities (#/ha) at BTA-R before and after the designation of the LPCNR in 1999*.

Species	**1998**	**2002**	**% Change**
Strombus gigas	40.0±16.3	3.9±3.9	**-90%**
Strombus pugilis	430.0±55.9	3.9±2.7	**-99%**
Strombus costatus	0±0	0±0	**-**

Mean±one standard error.

TABLE 16. Comparison of Conch densities at BTMS-R between years 1998 and 2002*.

Parameter	**D.F.**	**Kruskal-Wallis Statistic**	***p***
Strombus gigas density** x Year	1,17	3.94	0.0471
Strombus pugilis density x Year	1,17	14.32	<0.0001
Strombus costatus density x Year	1,17	1.38	0.2520 (N.S.)***

*Kruskal-Wallis non-parametric ANOVA.
** Log (x+1)-transformed data.
***N.S.= Not Significant.

TABLE 17. Comparison of Conch densities at BTA-R between years 1998 and 2002*.

Parameter	**D.F.**	**Kruskal-Wallis Statistic**	***p***
Strombus gigas density** x Year	1,34	10.32	0.0013
Strombus pugilis density x Year	1,34	29.96	<0.0001
Strombus costatus density x Year	N.A.	N.A.	N.A.***

*Kruskal-Wallis non-parametric ANOVA.
** Log (x+1)-transformed data.
***N.A.= Not analyzed, no *S. costatus* individuals observed.

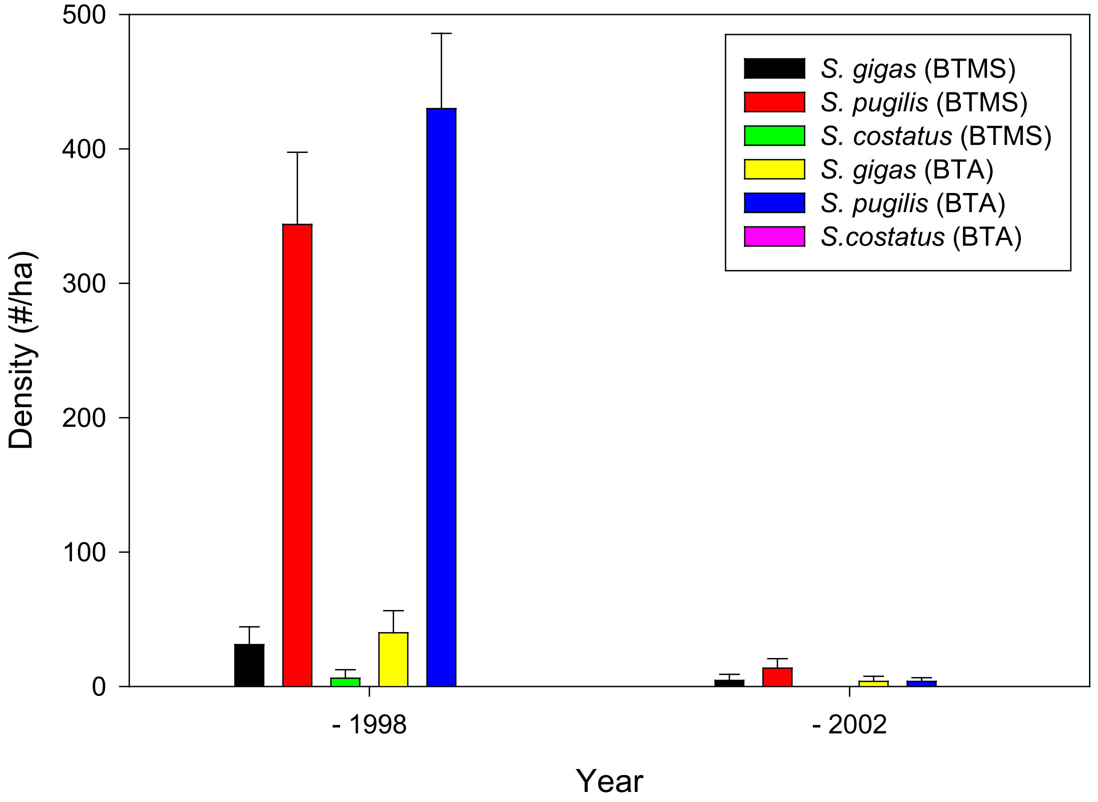


FIGURE 38. Major changes in conch densities (#/ha) between years 1998 and 2002 within BTMN-R and BTA-R, before and after the designation (1999) of the LPCNR.

The statistical power of the Conch density data was high only for the BTMS-R 1998 *Strombus pugilis* data (89%) (Appendix, Table A6) and for the BTA-R 1998 data of this species (87%) (Appendix, Table A7). During 2002, the statistical species for this species declined to 49% at BTMS-R and to 31% at BTA-R. Statistical power of the *S. gigas* density data was substantially low during both surveys, particularly, during 2002.

Water transparency.

The horizontal water transparency was measured only three times at each site (Table 18). Therefore, no statistical analyses were carried out to this parameter. But there was a consistent decline in the Secchi disk horizontal water transparency when moving from the core seagrass habitat within the LPCNR at BTMN-R (19.3 m) to the farthest control study site at PMA-C (8.7 m) (Figure 39). The mean statistical power of this data set was 97% (Appendix, Table A8).

Regression analyses.

A set of regression analyses were carried out to determine the relationships of seagrass bed, Conch and environmental parameters. There was a relatively strong linear relationship between the standing crop biomass of *T. testudinum* and the horizontal water transparency (r^2=0.7778) (Figure 40). Moreover, there was an even stronger linear relationship among the *T. testudinum* LAI and the horizontal water transparency (r^2=0.9653) (Figure 41). We documented also a strong quadratic relationship between the *Thalassia testudinum* standing crop biomass and LAI (r^2=0.9069) (Figure 42). In addition, we correlated the % of seagrass cover, with LAI, using the horizontal water transparency as a co-variate (Figure 43) and found a moderately strong relationship (r^2=0.6256). There was also a strong polynomial 2nd order inverse relationship

TABLE 18. Mean horizontal water transparency (m) at each site*.

Site	# samples	Secchi horizontal water transparency (m)
BTMN-R	3	19.3±0.7
BTMS-R	3	16.0±1.0
BTA-R	3	12.0±0.0
BMO-C	3	12.0±0.0
PMA-C	3	8.7±0.7

Mean±one standard error.

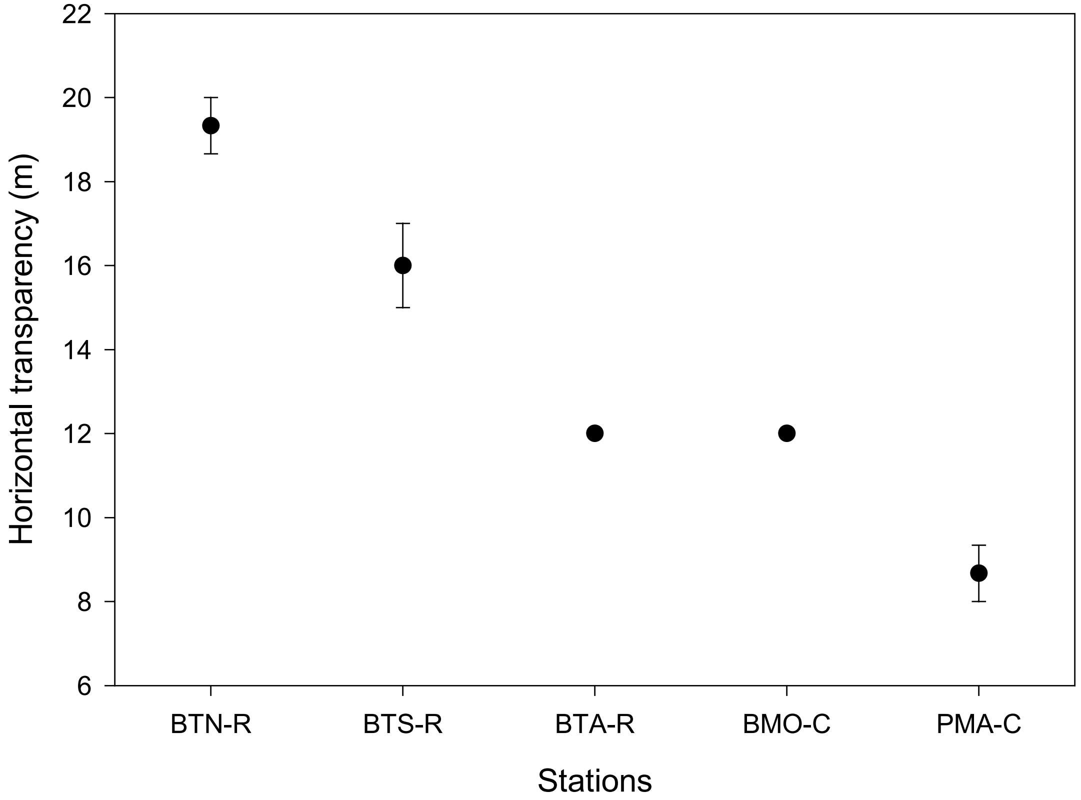


FIGURE 39. Horizontal Secchi disk water transparency (m). Mean±one standard error.

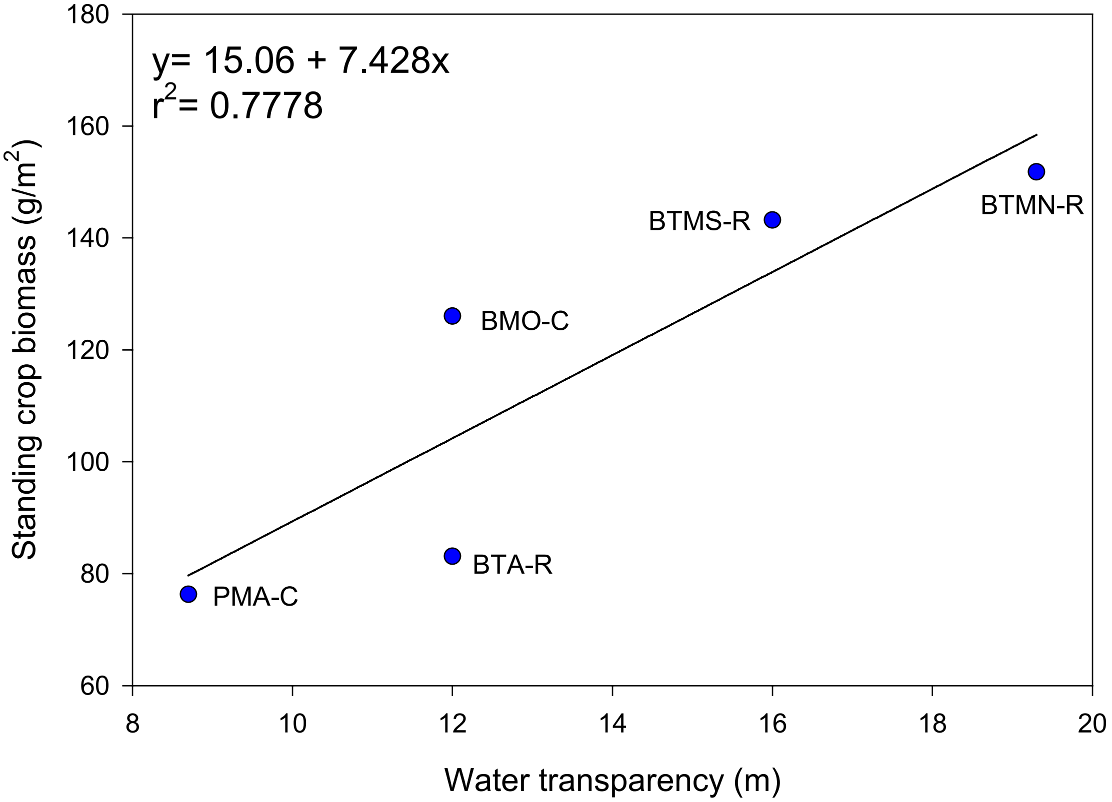


FIGURE 40. Linear regression plot of *Thalassia testudinum* standing crop biomass as a function of the horizontal water transparency.

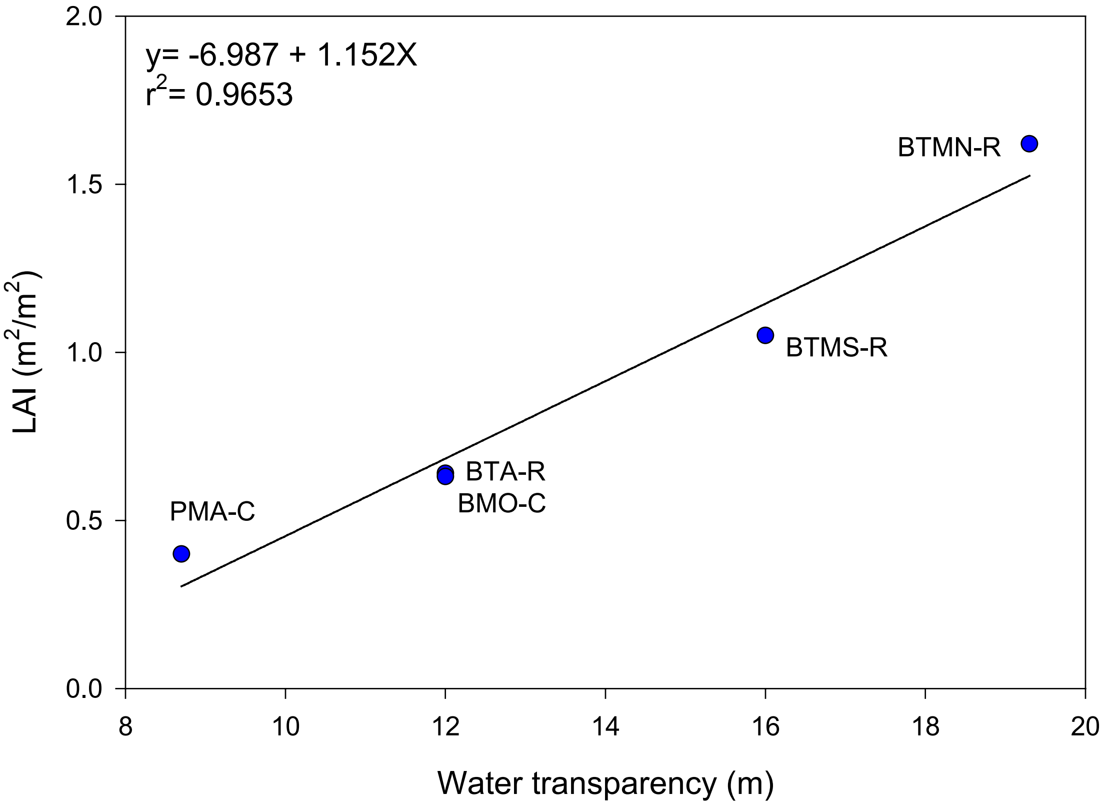


FIGURE 41. Linear regression plot of *Thalassia testudinum* leaf area index as a function of the horizontal water transparency.

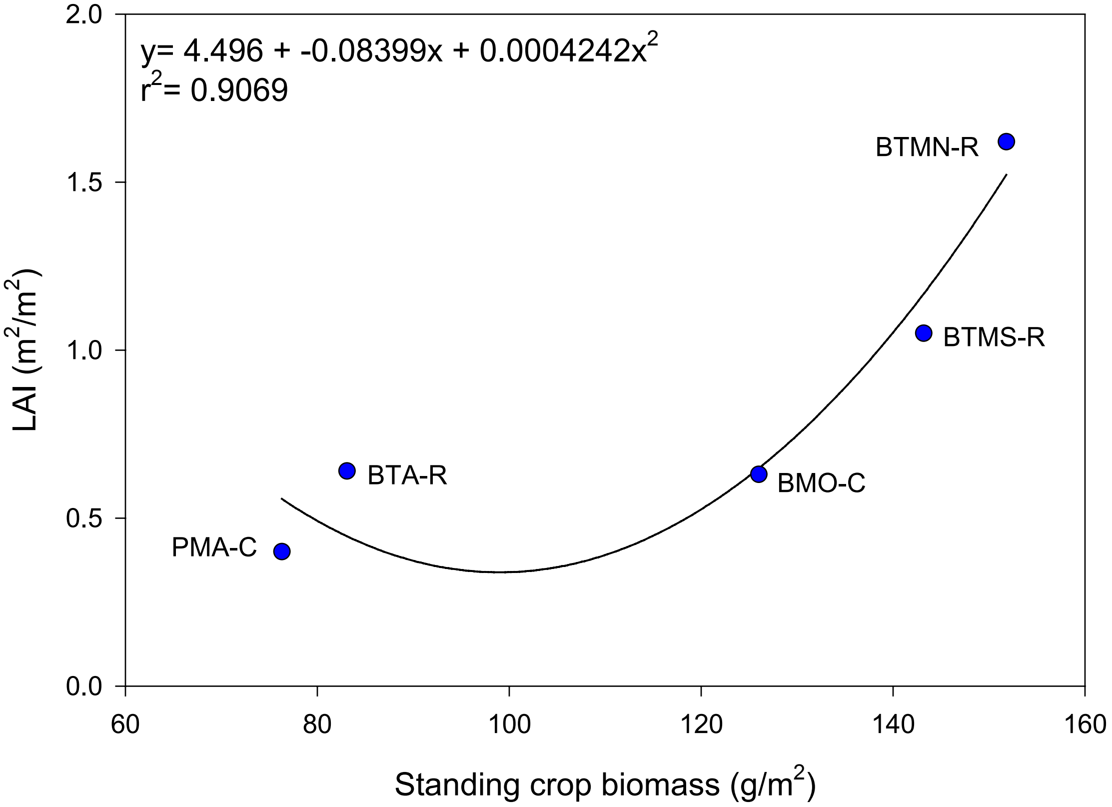


FIGURE 42. Quadratic polynomial regression plot of *Thalassia testudinum* leaf area index as a function of standing crop biomass.

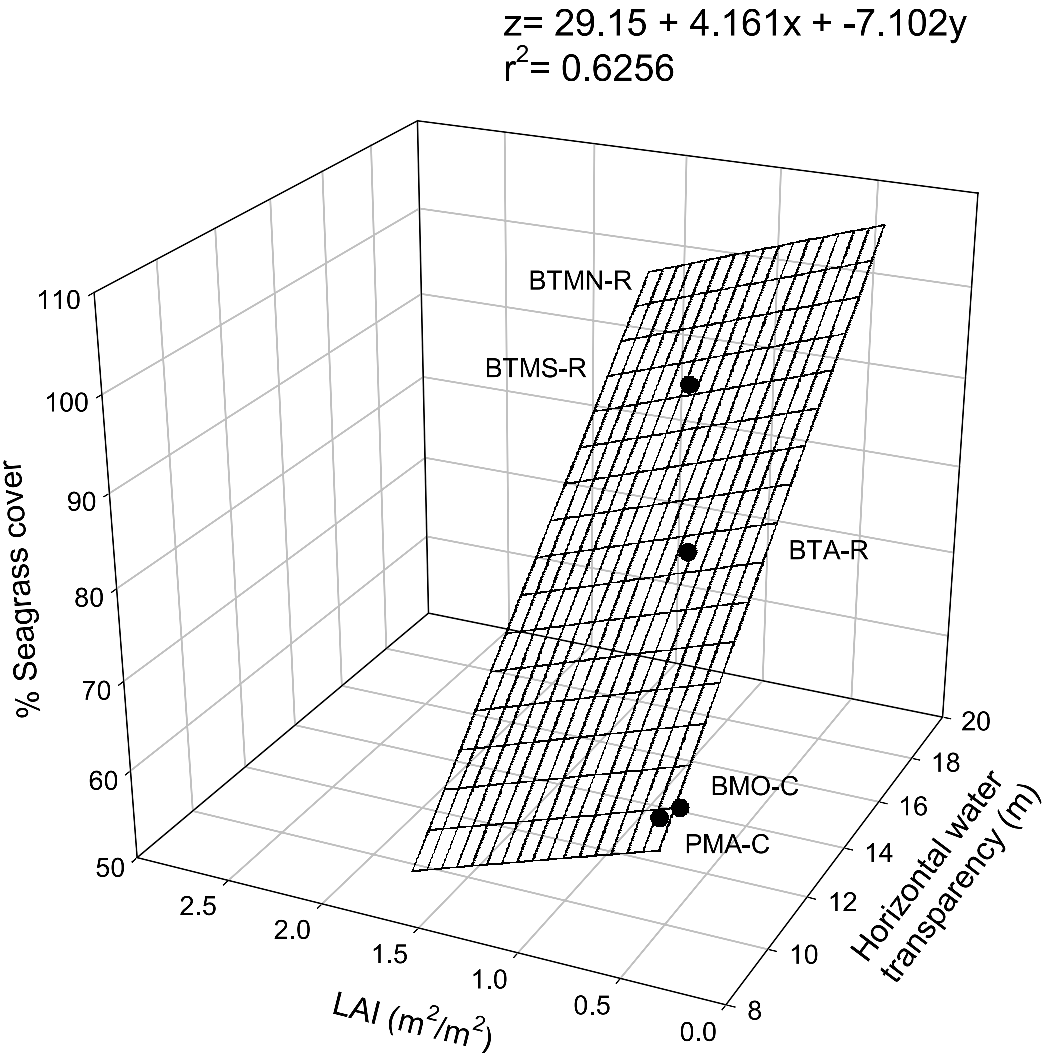


FIGURE 43. Tridimentional regression plot of the *Thalassia testudinum* % cover and leaf area index as a function of the horizontal water transparency.

between the % of algal cover and the horizontal water transparency (r^2=0.9845) (Figure 44). Finally, a strong relationship (r^2=0.9903) was observed between the % of algal cover and the total algal standing crop biomass, using the horizontal water transparency as a co-variate (Figure 45). However, Conch densities did not correlated with any of the seagrass community parameters and environmental variables (Table 19).

DISCUSION

Seagrass bed communities.

Seagrass bed communities in Culebra Island can be considered to be representative of the northeastern Caribbean region. Mean seagrass cover values as high as 57 to 97% were observed in study sites both within and outside the LPCNR boundaries. Similarly, relatively high shoot densities of *Thalassia testudinum* (660-1,820 shoots/m^2), *Syringodium filiforme* (200-653 shoots/m^2), and of *Halodule wrightii* (0-1,300 shoots/m^2) were documented in this study. In comparison, Zieman et al. (1997) reported densities of *T. testudinum* averaging approximately 1,000 shoots/m^2 at Belize, 300 shoots/m^2 at Colombia, 290 shoots/m^2 at La Parguera, PR, 260 shoots/m^2 at Cuba, 160 shoots/m^2 at Venezuela, and 70 shoots/m^2 at Curaçao. Cortés (1998) reported a density of 1,035 shoots/m^2 for *T. testudinum* and of 731 shoots/m^2 for *S. filiforme* at Parque Nacional Cahuita, Costa Rica. Gerace et al. (1998) observed densities of *T. testudinum* as high as 660 to 1,800 shoots/m^2, as high as 450 to 2000 shoots/m^2 for *S. filiforme*, and as high as 470 shoots/m^2 for *Halodule wrightii* at San Salvador, Bahamas. Hernández-Delgado (unpublished data) documented densities of *Thalassia testudinum* of 488 to 576 shoots/m^2 at Ensenada Yegua, Fajardo, PR, and of 1,074 shoots/m^2 at Bahía Demajagua, Fajardo. Williams

TABLE 19. Summary of linear regression analysis of Conch density and several seagrass bed community parameters*.

Parameter of regression analysis	Strombus gigas	Strombus pugilis
% Seagrass cover	$y= 18.55 + -0.1651x$ $r^2= 0.1590$	$y= -108.8 + 2.023x$ $r^2= 0.1990$
***T. testudinum* density**	$y = 11.71+ -0.002334x$ $r^2= 0.0212$	$y = -65.35 + 0.07653x$ $r^2= 0.3321$
***T. testudinum* standing crop biomass**	$y = 7.903 + 0.006524x$ $r^2= 0.1735$	$y = -8.727 + 0.3746x$ $r^2= 0.0361$
***T. testudinum* LAI**	$y = 2.478 + 7.123x$ $r^2= 0.1735$	$y = 68.13 + -38.45x$ $r^2= 0.0737$
% Total algal cover	$y = 6.397 + 0.3092x$ $r^2= 0.1563$	$y = 43.67 + -1.218x$ $r^2= 0.0353$
Total algal standing crop biomass	$y = 5.838 + 0.03203x$ $r^2= 0.2544$	$y = 43.77 + -0.1022x$ $r^2= 0.0377$
Horizontal water transparency	$y = 1.886 + 0.4981x$ $r^2= 0.0617$	$y = 79.91 + -3.320x$ $r^2= 0.0399$

*Analysis based on mean values for each site.

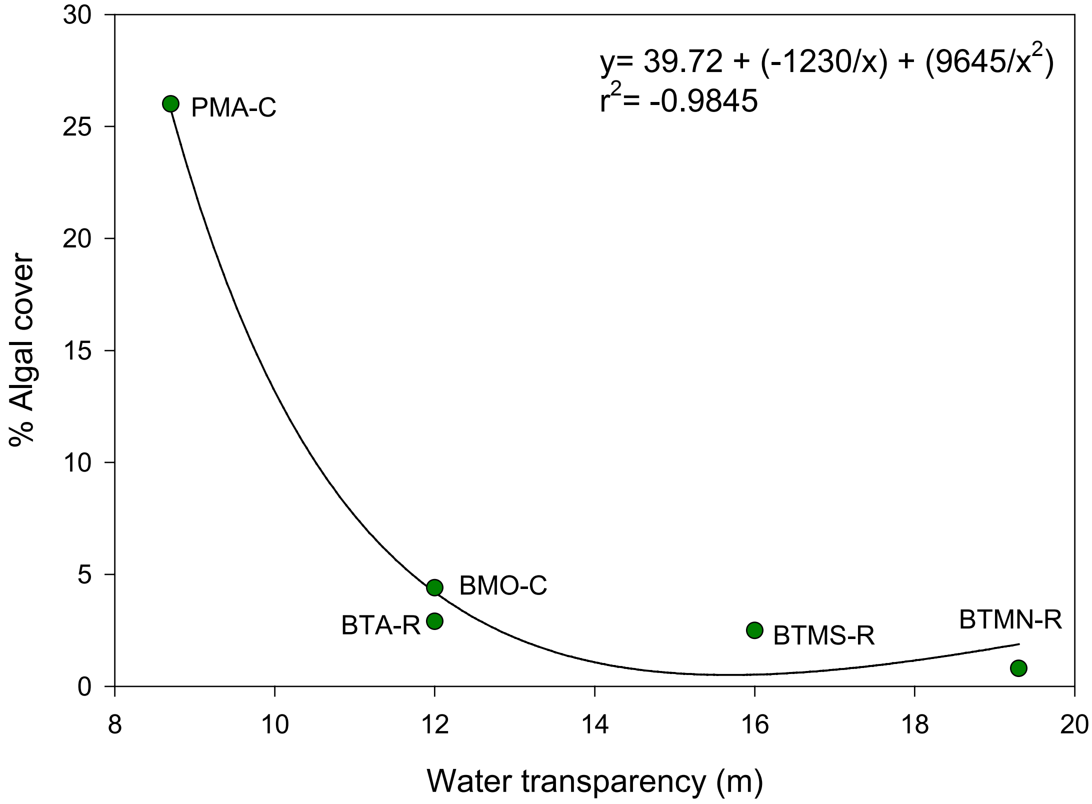


FIGURE 44. Polynomial 2nd order inverse regression plot of the % of total algal cover as a function of the horizontal water transparency.

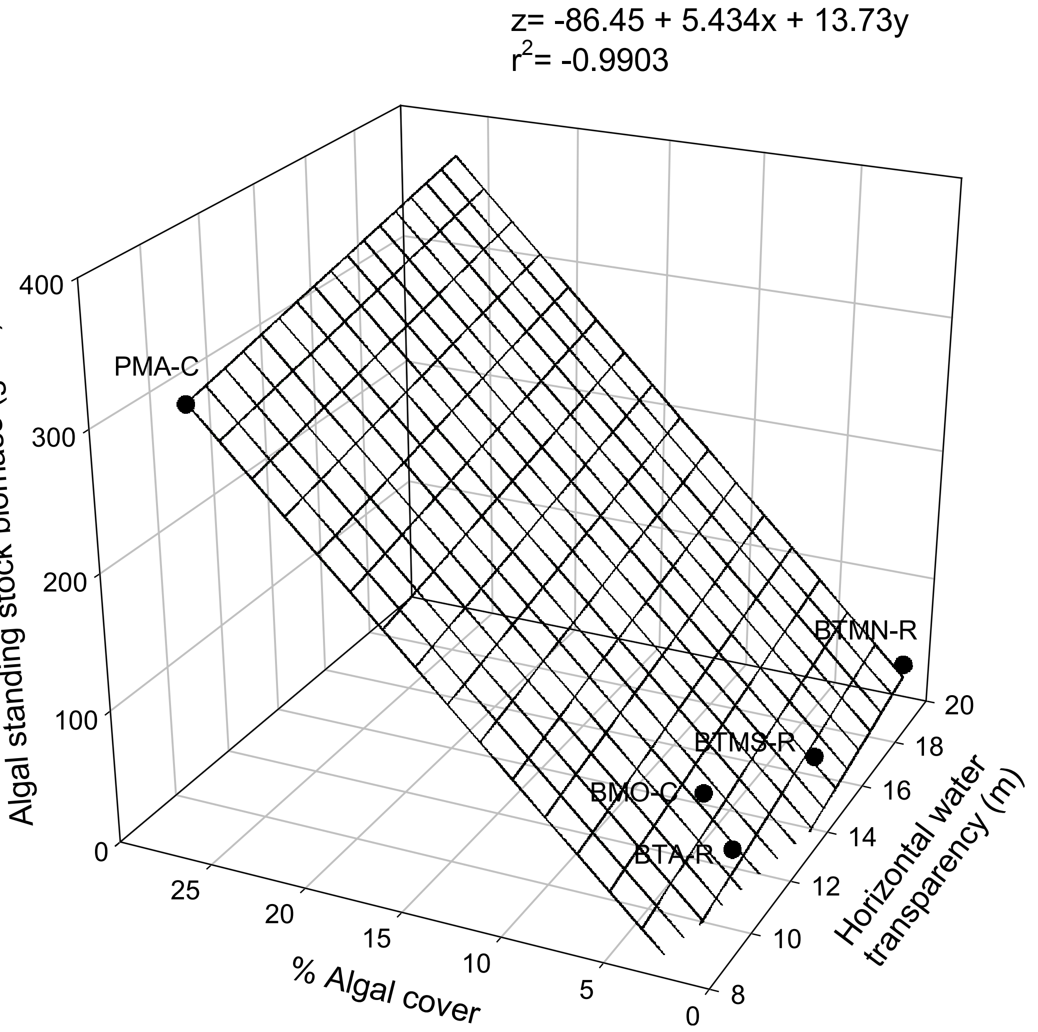


FIGURE 45. Tridimentional regression plot of the % of total algal cover and standing crop biomass as a function of the horizontal water transparency.

(1988a) informed densities of *T. testudinum* fluctuating between 80 and 200 shoots/m^2 in St. John, USVI. Densities of *S. filiforme* at St. John fluctuated from 0 to 154 shoots/m^2, and those of *H. wrightii* ranged from 84 to 290 shoots/m^2 (Williams, 1988a). Averza and Almodóvar (1985) reported densities of *H. wrightii* ranging from only 64 to 113 shoots/m^2 at Cayo Laurel, La Parguera.

Seagrass beds are highly productive ecosystems (Westlake, 1963; Patriquin, 1973). Sseagrass standing crop biomass, which is one of the major indicators of productivity, could be affected by a sort of natural and anthropogenic factors. Natural factors may include the exposure to wave action, nutrient availability, seasonal fluctuations, and herbivory (or lack of herbivory). The bucktooth parrotfish, *Sparisoma radians*, is one of the most common consumers of seagrasses (Ogden and Zieman, 1977), and was also occasionally observed during our study. But also, anthropogenic factors such as water quality degradation, eutrophication, sedimentation, power boating, anchoring, and coastal development, among others, can cause severe permanent damage to seagrass bed habitats. Seagrass standing crop biomass in Culebra Island was comparable to many other studies elsewhere through the region. Biomass of *T. testudinum* fluctuated from 76 to 152 g/m^2 in this study. In the case of *S. filiforme*, biomass ranged from 2.5 to 162 g/m^2, and from 0 to 10.3 g/m^2 in *H. wrightii*. Calcareous algal biomass also fluctuated from 15 to 307 g/m^2.

Burkholder et al. (1959) informed standing crop values for *Thalassia testudinum* at La Parguera, PR, as high as 99 to 1,353 g/m^2, and of 80 to 450 g/m^2 through the rest of PR. Delgado-Hyland (1978) registered values ranging from 96 to 480 g/m^2 at La Parguera.

González-Liboy (1979) documented seagrass standing crops fluctuating between 33 and 904 g/m^2 through PR. This author also informed values as low as 24 g/m^2 in Punta Guayacán, La Parguera, PR, and as 281 g/m^2 in Punta Medio Mundo, Ceiba, PR, for this species. Vicente (1992) documented total biomass values for *T. testudinum* fluctuating from 100 to 5,800 g/m^2 from Bahía de Jobos, Salinas, PR, and a mean total biomass value of 330 g/m^2 on Guayanilla, PR. Cintrón (unpubl. rep., as cited by Odum, 1974) documented a seagrass standing crop biomass of 132 g/m^2 at Fajardo, PR. Gayle and Woodley (1998) informed mean *T. testudinum* biomass of 1,045 g/m^2 and a mean biomass of 106 g/m^2 for calcareous algae. Ruíz-Rentería et al. (1998) informed mean values of 5 to 80 g/m^2 for *T. testudinum*, and of 2 to 160 g/m^2 for *S. filiforme* at Puerto Morelos, México. Koltes et al. (1998) documented a standing crop biomass of 731 to 883 g/m^2 for *T. testudinum* at Carrie Bow Key, Belize, while Garzón-Ferreira (1998) informed mean values of 49 to 221 g/m^2 for this species at Bahía de Chengue, Colombia. Pors and Nagelkerken (1998) observed values of 193 to 504 g/m^2 at Curaçao, and values ranging from 190 to 840 g/m^2 were reported by Varela et al. (1998) from Isla Margarita, Venezuela. *Thalassia testudinum* standing crop biomass at Cayo Coco, Cuba, ranged from 60 to 116 g/m^2 (Alcolado et al., 1998). Smith (1998) also observed a mean 790 g/m^2 at Bermuda. Hernández-Delgado (unpublished data) observed in 1993 standing crop values fluctuating between 314 and 332 g/m^2 for *T. testudinum*, an average of 96 g/m^2 for *S.filiforme*, and values fluctuating between 161 and 1,154 g/m^2 for macroalgae in Ensenada Yegua, Fajardo. But Williams (1988a) informed a standing stock biomass fluctuation of 1.8 to 4.6 g/m^2 for *T. testudinum* on anchor-disturbed habitats in St. John. Under severe coastal water degradation in Peñuelas (i.e., high turbidity, high sedimentation rates), Hernández-Delgado (1997) documented a seagrass standing crop fluctuating only between 1 and 99 g/m^2. Lower values were also documented in seagrass beds severely scoured by power boat navigation.

The *Thalassia testudinum* LAI at Culebra was considered to be rather low (0.40 to 1.62 m^2/m^2) in comparison to other studies in the region. Zieman et al. (1997) informed mean LAI values approximately of 8.5 m^2/m^2 at Colombia, 6.0 m^2/m^2 at Belize, 4.5 m^2/m^2 at Cuba, 3.5 m^2/m^2 at Venezuela, 2.5 m^2/m^2 at La Parguera, PR, and of 0.75 m^2/m^2 at Curaçao. But Dixon (2000) informed LAI values fluctuating only between 0.080 and 0.413 m^2/m^2 at Tampa Bay, Fl. To our knowledge, the highest LAI value ever published in the Caribbean was that of Gessner (1971) from Venezuela with 18.6 m^2/m^2, and a standing crop biomass of 608 g/m^2. This was followed by a high value of 16.8 m^2/m^2 and a standing crop biomass of 616 g/m^2 from southwestern PR.

Mean leaf length of *T. testudinum* fluctuated between 96 and 137 mm during this study, values which are comparable to other localities through the region. Vicente (1992) reported mean leaf length values of 60 mm from intertidal seagrasses up to 151 mm under low herbivory conditions from southwest PR. Hernández-Delgado (unpublished data) documented mean leaf length values of 181 mm at Ensenada Yegua, Fajardo, PR, and of 169 mm at Bahía Demajagua, Fajardo. Bush (1998) informed values as large as 450 mm at Central Swamp, Cayman Islands. Zieman et al. (1997) informed mean leaf length of approximately 220 mm from Colombia, 170 mm from Belize, 180 mm from Cuba, 155 mm from Curaçao, 130 mm from Venezuela, and 100 mm from La Parguera, PR. Ruíz-Rentería et al. (1998) documented mean leaf length values for Puerto Morelos, México, of 90-150 mm, while García and Holtermann (1998) informed values as high as 300 mm from Belize. Smith (1998) documented values of 50 to 110 mm at Bermuda.

Leaf lengths of 79-194 mm were informed at Tampa Bay, Fl (Dixon, 2000). González-Liboy (1979) informed values fluctuating from 95 mm at Medio Mundo, Ceiba, PR, to 204 mm at Bahía Salinas del Sur, Vieques Island, PR. González-Liboy (1979) documented a mean leaf length of 193 mm, with a range of 45 to 335 mm at PMA-C. However, in this study, mean leaf length at PMA-C was estimated only in 110 mm, with a range of 18 to 276 mm. This decline can be the result of natural fluctuations, a long-term change in the environmental conditions of PMA-C (see discussion below), or the indirect result of the long-term effects of serial overfishing. Artisannal and recreational fisheries have shifted from first class top-predators to second class predators and third class herbivores. Lack of herbivory has been associated to increasing macroalgal overgrowth (Hernández-Delgado, 2000). Drifting macroalgal mats could significantly limit seagrass development by several mechanisms that will be discussed below.

Mean leaf width of *T. testudinum* in this study fluctuated between 6.8 and 8.3 mm. Zieman et al. (1997) reported mean leaf lengths of approximately 16 mm from Venezuela, 15 mm from Colombia, 12 mm from Belize, 9 mm from Curaçao, and 8 mm from Cuba and PR, respectively. Hernández-Delgado (unpublished data) observed mean values of 8.2 mm at Ensenada Yegua, Fajardo, and of 9.4 at Bahía Demajagua. Vermaat et al. (1996) suggested that seagrasses can produce such shifts in morphology as a response to environmental stresses (i.e., increasing turbidity). It is possible that higher leaf width values at Fajardo, relative to Culebra, could be a phenotypic response of seagrasses as a result of long-term environmental fluctuations, higher background nutrient levels, lower herbivory, etc.

Many of the physically-disturbed zones at PMA-C, BTA-R and BTMS-R showed extensive growth of *Halodule wrightii* which is considered to be a pioneer seagrass species, with a high ability to recolonize disturbed habitats (Averza and Almodóvar, 1986; Fonseca et al., 1994). Averza and Almodóvar (1984) informed a standing crop biomass of *H. wrightii* of 0.86 g/m^2 at Bahía Fosforecente, La Parguera. Averza and Almodóvar (1985) informed mean values of *H. wrightii* standing crop biomass fluctuating between 0.98 and 1.33 g/m^2, and a density of 64 to 113 shoots/m^2 at Cayo Laurel, La Parguera. However, in this study mean standing crop biomass values for this species ranged from 0 to 10.3 g/m^2. Density of this species reached values of 400 shoots/m^2 at BMO-C and of 1,300 shoots/m^2 at PMA-C. There were two sites in our study where this species was absent. This difference could be the result of differences in the disturbance regimes. Also may be indicative that seagrass bed habitats could still be recovering from recent hurricane impacts and there is an active recolonization of scoured bottoms.

Several seagrass parameters ranked lower at PMA-C or at BTA-R during this study. Both sites are frequently exposed to strong currents and occasional surge under natural conditions. But also, are frequently receiving the impacts of recreational navigation (i.e., anchoring, sediment resuspension due to scouring by propellers). Hernández-Delgado (1994) observed that seagrass cover drop to mean values of 3 to 31% in the margins of propeller-scoured seagrass beds at Mata de la Gata Island, La Parguera, PR. Cover dropped to 0% within the scour channel. However, control sites sustained a seagrass bed with a 60-80% cover, suggesting that propeller scouring, as well as jet skis turbine scouring, could inflict severe physical damage to the bottom. Power boating could also result in a major alteration of the seagrass epibenthic faunal community structure (Holmquist et al., 1989; Hernández-Delgado, 1994).

In the particular case of PMA-C, there is also a major problem of frequent highly sedimented runoff coming from poorly managed and totally land-cleared private properties (Hernández-Delgado and Lucking, personal observations). Increasing runoff could probably result in a long-term decline in water transparency. PMA-C showed the lowest mean horizontal water transparency of all of the study sites, but also the highest cover and biomass of algae. There was a strong negative correlation between declining horizontal water transparency, and delining seagrass standing crop biomass and LAI. But, there was a strong positive relationship between declining horizontal water transparency and increasing total algal % of cover. This suggests that some of our study sites, particularly PMA-C, are already showing signs of environmental degradation, most probably related to inadequate land use and water quality.

PMA-C frequently harbors dozens to hundreds of recreational boats at the same time, particularly during weekends. Thus, the combination of natural sediment resuspension due to wave action, sediment resuspension due to propeller scouring, pollution from recreational vessels, runoff and other non-point sources of pollution (i.e., cesspools, septic tanks) could be contributing to a possible eutrophication problem at PMA-C which needs to be assessed by means of a water quality characterization study. It has been already shown that water quality degradation (i.e., eutrophication, high turbidity, high concentration of solid suspended material, low water transparency and high sedimentation rates) can cause a major decline in seagrass cover (Vicente et al., 1980; Vicente and Rivera, 1982; Vicente, 1992; Vermaat et al., 1996; Hernández-Delgado, 1997).

Early eutrophication is also highly suspected at PMA-C due to the lowest water transparency and high algal cover, abundance and biomass, in relation to other surveyed sites. This site also showed the only documented aggregation of drifting macroalgal mats during the study. Orth and Van Montfrans (1984) demonstrated that such mats could obstruct sunlight penetration, could create thermal stratification of the bottom, could increase photosynthetic rates on local scales to such high levels that pH values in the bottom could rise to levels above 9.0, causing a limitation of bicarbonate ions to the seagrasses that could lead to a cease of all photosynthetic activity. Algal mats could also increase the biological oxygen demand on local scales due to increasing respiration rates during night time, thus contributing to local increasing mortality of seagrasses. Neely (2000) also demonstrated through an elegant experiment that at least a 30% light attenuation level can result in a significant decline in the density, biomass and productivity of seagrasses. Beal and Schmit (2000) also documented a significant decline in the % of cover and density of seagrass shoots exposed to low irradiance levels. It has been found that once *T. testudinum* disappears, its recovery is extremely difficult (Williams, 1988b, 1990; Thayer et al., 1994). Thus, it is suggested that a possible combination of natural and anthropogenic factors could explain the observed differences in seagrass cover and other parameters. A particular concern is the rapid development trend on the lands surrounding PMA-C and that the highest density of Queen Conch, *Strombus gigas* was observed at this site.

Another potential sign of disturbance at PMA-C is the high variation documented in most of the parameters, which resulted in a very poor statistical power of the analysis. This is significant because high variation at the community level could be an unequivocal sign of disturbance. The multivariate analysis approach similarly pointed out at a major geographic separation of PMA-C and, into some extent BTA-R and one of the sampling stations at BMO-C. Special attention should be also given to the fact that BTA-R and BMO-C could be starting to show up similar signs of degradation such as PMA-C. Both sites receive the indirect effects of water quality degradation within Ensenada Honda, which receive raw sewage effluents. During low tide, water from Ensenada Honda moves to BMO-C through the main bay entrance, and to BTA-R through the highly polluted Lobina Channel. This channel has been pointed out as a potential source of nutrients causing a rapid increase in macroalgal cover within the LPCNR coral reefs (Hernández-Delgado, 2001). It is important to mention that, in spite of the fact that traditional statistical methods failed to address geographic differences among the seagrass bed communities at the individual parameter level, multivariate analysis proved to be a highly precise and reliable tool to determine geographic variation at the community level on seagrass bed habitats.

The usually calmer oceanographic conditions of shallow seagrass bed environments attracts many recreational activities. Aquatic recreational activities in the coastal waters of Puerto Rico have significantly increased in recent years (R. Chaparro, U.P.R., Sea Grant College Program, personal communication). This has been associated to a recent increase in the construction of private marinas, increased interest in water-based recreation, and to the introduction of a wide variety of jet skis and power boats to the local market. Anthropogenic threats to coastal natural communities such as coral reefs and hard grounds in Puerto Rico, have been thoroughly reviewed by Goenaga and Cintrón (1979), Velazco-Domínguez et al. (1985), Goenaga (1986, 1991), Goenaga and Boulon (1992), Hernández-Delgado (1992, 2000). However, besides González-Liboy (1979), the impact of anthropogenic activities on seagrass beds in Puerto Rico has been poorly documented. Thus, our first general recommendation is to coordinate with the PRDNER a major effort to carry out a general assessment of the status of segrass bed essential fish habitats throughout the Puerto Rican archipelago. Such an assessment should be based in actual standard methods (i.e., CARICOMP) and should have a hierarchical organization (i.e., geographical provinces, anthropogenic stress gradients, depth gradients).

A second major recommendation, specific to Culebra Island, is to carefully assess the direct and indirect impacts of land clearing activities on seagrass bed habitats. Preliminary evidence obtained in this study suggests that major community changes could be related to long-term chronic water quality degradation associated to land development and, possibly, to the lack of sewage treatment in Culebra and to increasing recreational boating activities. A hierarchical design is also recommended that should include: runoff, turbidity and sedimentation gradients, as well as Culebra's landfill effects. A third recommendation specific to Culebra Island, is to document the direct and indirect impacts of recreational navigation on seagrass bed habitats. Preliminary evidence also suggests that seagrass bed community declines could be related to recreational boating activities and anchoring. A hierarchical design is also recommended that should include: anchoring gradients, zones with propeller scars, and control zones with mooring buoys.

The most significant threats to seagrass habitats observed in Culebra Island associated to recreational activities and coastal development include: 1) uncontrolled recreational navigation of power boats and jet skis over shallow seagrass bottoms (<3 m); 2) anchoring; 3) seagrass tear-off, bottom blow outs, and sediment resuspension caused by propellers, turbines and anchors; 4) garbage dumping; 5) illegal point-source and non-point source sewage discharges, including from vessels; 6) damage caused by tramplers; 7) spearfishing and gill netting; 8) collection of organisms for the aquarium trade; 9) construction on the maritime zone; 10) increased water turbidity and concentration of suspended solid material within Ensenada Honda and its surroundings; 11) sewage- and runoff-associated eutrophication; and 12) other non-point source pollution (i.e., oil and fuel spills). In spite of that, there is a lack of quantitative data regarding the impacts of such factors, particularly, water quality issues. Thus, our fourth recommendation is to implement a comprehensive hierarchically-designed question-driven long-term water quality monitoring program parallel to a long-term seagrass bed ecological change monitoring program. Such programs should be implemented in partnerships with NOAA, PRDNER, the University of Puerto Rico, the Culebra Island Fishermen Association, the Municipal government, and any other interested stakeholder. The final product of such efforts should be the development of coral reef-seagrass bed co-management plan.

A specific fifth recommendation regarding the seagrass long-term monitoring program should be to include an epiphyte photo-index (EPI) for aasessing epiphytic algal loadings on seagrasses (Miller-Myers and Virnstein, 2000). Epyphyte loadings were significant in some seagrass bed communities during our study, particularly at PMA-C, and at BTMS-R, the closest seagrass bed to the Culebra Island's municipal landfill, which showed a particularly high density of cyanobacteria. EPI is a rapid non-destructive approach to visually determine epiphyte loadings of seagrasses. Seagrass epiphytism is one of the major factors associated to mortality (Orth anb van Montfrans, 1984; Miller-Myers and Virnstein, 2000) and should be incorporated into regular monitoring programs.

Hernández-Delgado (1994) documented that boating and anchoring activities in Cayo Mata de la Gata affected: 1) seagrass canopy by significantly reducing or eliminating seagrass cover; 2) reducing seagrass shoot density; 3) reducing seagrass leaf length; 4) affecting epibenthic faunal community structure (declining abundances of some taxa); and 5) affecting epibenthic faunal species distribution. The most obvious effects were the near-elimination of vegetation due to the scouring caused by propellers or by the formation of gaps or blowouts caused by anchoring. *Thalassia testudinum* is not well adapted for rapid recolonization following disturbance, particularly if rhizomes have been severely damaged. It has a slow rate of rhizome elongation, a low population growth rate, produces few seeds, has no seed reserve, has poor seedling success and low productivity (Williams, 1988b; 1990). Our fifth recommendation is that the CFMC should recommend the PRDNER that boat traffic should be regulated around the Culebra Island shallow bottoms, particularly the maximum boat speed on shallow seagrass and coral reef habitats. Collisions signs are evident from both types of habitats around the Island, therefore a system of marker buoys with speed limits and identification/informative markers should be established on both types of communities.

Jet skiing is most probably the most threatening anthropogenic factor affecting seagrass bottoms at Cabezas de San Juan Natural Reserve, Fajardo, PR, and at La Cordillera Natural Reserve, Fajardo. There are already major signs of damages to these habitats. However, jet skis are not still a major threat in Culebra, mostly due to the large distance from Fajardo. But, although at a low scale, the problem is present. Early signs of damage include higher water turbidity, small sporadic seagrass blowouts and severe resuspension of sediments (Hernández-

Delgado, personal observations). However, there are no quantitative accounts of impacts. Moreover, there are no specific restrictions to jet skiing since there are no management plans for the LPCNR. Since slow recovery following disturbance is typical for seagrasses such as *T. testudinum*, and since boating and jet skiing activities are expected to increase in Culebra Island, including the LPCNR, our sixth recommendation is to recommend the PRDNER to establish sound management measures for the LPCNR to regulate boating and jet skiing activities within the Reserve's boundaries.

Conch populations.

Queen Conch, *Strombus gigas*, fishery has been overexploited in Puerto Rico. It is particularly vulnerable in its shallow-water aggregations during the reproductive season, which extends from May to November each year (CFMC, 1996). The reproductive season of the Fighting Conch, *S. pugilis* extends from March to November (Reed, 1992) and are being similarly harvested all year-round. Conch posses internal fertilization and fertilized egg masses are laid on clean, low-organic content sands, on *Thalassia testudinum*, other seagrass species and near coral reefs. Female Conchs cover eggs with sands to protect them from predators. Each female *Strombus gigas* can reproduce from 6 to 8 times per season. However, heavy surge and cold temperatures could be detrimental to their reproductive success. Conchs are known to migrate from deep open sandy bottoms or hard grounds to shallow seagrass habitats during their reproductive season. Shallow habitats are also one of the most important nursery grounds for juvenile stages before their ontogenic shift towards deeper habitats during their adult stages. However, Conch populations are extremely vulnerable to overfishing on such shallow habitats. This has forced the Puerto Rican government to establish of a seasonal closure for *S. gigas* between July 1 and September 30 of each year. During the rest of the year, Conch size restrictions are implemented, which include a 9" (25 cm) minimum shell length and a 3/8" (9.375 mm) minimum lip width size limit. Also, there is a maximum daily catch limit of 3 Conch shells per person and/or 12 Conch shells per boat for recreational/incidental fishermen, and a maximum of 150 conch/boat during the first year of the restrictions, 100 during the second year, and 75 during the third year, for commercial fishermen. The use of Hookah in the EEZ is also prohibited.

Conch species represent one of the most important fishery species in Puerto Rico. During the period of 1983 to 2000, Queen Conch (*Strombus gigas*) ranked third among the most important fishery target species, with a total of 7.25% of all captures (Valle-Esquivel, 2002). Only the Caribbean Spiny Lobster (7.79%) and the Silk Snapper (9.28%) ranked higher than Conch in total landings. Most of the Conch catches are being done using SCUBA (82.04%), during spear fishing (8.42%), and during skin diving (5%) (Valle-Esquivel, 2002). Therefore, it is expected that the most severely impacted fishery stocks are those from shallow-water habitats. During the 1983-2000 time span, there was a nearly 30% decline in Puerto Rican Conch landings, which includes a simultaneously continuous 42% increase in mean price of Queen Conch meat by year (Valle-Esquivel, 2002). Culebra Island ranked lowest in terms of the Catch per unit effort (CPUE) of all species in the fishery within the east coast of Puerto Rico, with 38.84 CPUE (Valle-Esquivel, 2002). Vieques Island ranked highest in the region (92.64), followed by Fajardo (85.13) and Naguabo (71.23). This could be the result of the use of different fishing methods/gears, to the productivity of different fishing grounds, different fishery targets, etc. But the fact is that Queen Conch fisheries have been declining through the region at an alarming rate and there is still a lack of long-term data about the status of Conch populations, not only of *Strombus gigas*, but of *S. pugilis*, and *S. costatus*, which have been found to be important fishery targets, at least in eastern Puerto Rico (Hernández-Delgado, personal observations).

Most of the data concerning the Queen Conch population declines in Puerto Rico has come from minimum size and landings data obtained only for *S. gigas* (Appeldoorn, 1994), and keeping the information gap for *S. pugilis* and *S. costatus*. According to Rivera (1999), 24% of the total Queen Conch landed in Puerto Rico were undersized for shell length, and 19% for lip width. But 80% of the Conch harvested in St. Croix were juveniles (Tobias, 1987) and 91% of the catch from St. Thomas/St. John was under the 9" size limit (Rivera, 1999). Rodríguez and Posada (1994) observed that 87% of the Conch landings at Los Roques, Venezuela, were juveniles. However, field surveys are still scarce and just until recently populations assessments at an island wide level have been carried out. Our data from Culebra showed density fluctuations for *S. gigas* from 0 to 18.8 conchs/ha, with a mean of 9.1 conchs/ha within the LPCNR and a mean of 8.0 conchs/ha outside the Reserve's boundaries. That difference was not significant. A fluctuation of 0 to 156 conchs/ha was found for *S. pugilis*, with a mean of 5.83 conchs/ha within the LPCNR and a mean of 78.15 conchs/ha outside the Reserve's boundaries. The density of *Strombus costatus* ranged from 0 to 3.1 conchs/ha, with a mean of 1.55 conchs/ha outside the LPCNR's boundaries. This species was absent within the Reserve. Besides the mean 156 conchs/ha for *S.* pugilis at BMO-C, these figures are considered to be significantly low in comparison to other figures from the regional literature. Mateo-Rabelo (1997) informed densities of 9.20 conchs/ha for *S. gigas* at the Puerto Rican west coast, and of 7.46 conch/ha at the east coast. Previously, Torres Rosado (1987) informed a density of 8.11 from La Parguera, PR. In comparison, data from Culebra Island is not different from the remaining of Puerto Rico. Even the LCPMFR has not been effective in supporting high densities of Conch. But Appeldoorn (1995a, as cited by Mateo-Rabelo, 1997), informed densities of *S. gigas* at Pedro Bank, St. Croix, USVI, ranging from 89 to 277 conchs/ha. Elsewhere, Wood and Olsen (1983) informed densities of 7.60 conchs/ha at St. Croix, and of 9.70 at St. Thomas/St. John. Friedlander et al. (1994) informed densities of 12.25 conchs/ha at St. Thomas, and of 38-75 conhs/ha at St. John, even after 7 years of moratorium and new management regulations. Florida Keys showed very low densities (0-1.55 conchs/ha) (Glazer and Berg, 1994). Berg et al. (1992a,b) reported densitites of 0.52 to 2.94 conchs/ha in Bermuda.

An early indication of the severe Conch overfishing problem could also be the fact that 68% of the western shelf and 62% of the eastern shelf Queen Conch population visually surveyed in Puerto Rico by Mateo-Rabelo (1997) were juvenile. In this study, 75% of the Queen Conch shells surveyed in Culebra were juveniles. Maximum conch shell length data fluctuated from 0 to 239 mm for *S. gigas*, with a mean value of 105.2 mm. Shell length ranged from 0 to 158 mm for *S. pugilis*, with a mean value of 117 mm. The only conch of *S. costatus* found during the study measured 134 mm. Since this study was limited to waters less than 7 m in depth, surveyed habitats were mostly dominated by juvenile individuals. According to Appeldoorn (1991), nearby deeper habitats (21 m) in Culebra and eastern Vieques (35 m) support large-sized Queen Conch, with mean shell lengths of 243 mm and 228 mm, respectively. Moreover, Appeldoorn (1991) documented the existence of very old, highly eroded Queen Conch individuals in Culebra (21 m) locally known as “rafta conch” because of the algal flags on top of the shell. These conchs showed a mean shell length of 201 mm. Appeldoorn (1991) also informed mean shell lengths for *S. gigas* from Caja de Muertos, Ponce, PR (219 mm), Bajo El Sico, Mayagüez (244 mm), Tourmaline Reef, Mayagüez (235 mm), Buoy 6, Abrir la Sierra, Cabo Rojo (234 mm), La Parguera (247 mm), and from northern Vieques (231 mm).

Lip width in our study ranged from 0 to 3.0 mm in *S. gigas*, with a mean value of 1.13 mm. Mean lip width in *S. pugilis* was 0.99 mm, with a range of 0 to 3.65 mm. Lip width measured 0.55 mm in the only individual of *S. costatus* observed during this study. Lip width for adult *S. gigas* at Culebra and eastern Vieques averaged 26.4 and 21.5 mm, respectively (Appeldoorn, 1991). Mean lip width from Caja de Muerto was 9.4 mm, 14.9 mm at La Parguera, 15.9 mm at El Sico, 16.6 mm at Abrir la Sierra, and 21.5 at northern Vieques (Appeldoorn, 1991). It is evident that most of the *S. gigas* individuals sampled during our study were juveniles. Thus, most of the shallow-water fishery stock of conch of the three species is composed of juvenile individuals. Conchs are characterized by having deterministic growth, which means that shell length is basically fixed at maturation. Conch shell grow initially without lip thickening until maturation. Then lip thickens and shell ceases growth. This is significant because, according to Appeldoorn (1991), conch that mature at a size smaller than the minimum legal size would never be legal for harvest, but conch that mature at a size greater than the minimum would be legally harvestable as juveniles.

The characterization study of the Conch populations within the LPCNR showed an astonishing result: conch populations in shallow-water seagrass habitats have collapsed. Our data from 1998 (1 year prior to the LPCNR designation) showed a density of *S. gigas* of 31.3

conchs/ha, which dropped in 2002 (3 years after designation) to 4.6 conchs/ha at BTMS-R. A similar trend was observed at BTA-R, with 40.0 conchs/ha in 1998 and only 3.9 conchs/ha in 2002. This collapse was more striking for the Fighting Conch, *S. pugiliss*, with densities in 1998 of 343.8 and 430 conchs/ha at BTMS-R and BTA-R, respectively. These plunged down in 2002 to 13.6 and 3.9 conchs/ha, respectively. These variations may be the result of natural fluctuations in conch recruitment patterns, possible shifts in the 2002 reproductive aggregation behavior, and/or severe overfishing. The LPCNR should have functioned as a control to exclude fishing effects, but it did not. Our data suggests that, at least in Culebra, the Queen Conch seasonal closure in effect since year 1997 has not been effective in helping restoring the population densities of this species. In spite of the fact that within 3 years finfish communities have shown a major recovery within the LPCNR (Hernández-Delgado and Sabat, in preparation), it has failed to recover Conch populations.

According to Rodríguez and Posada (1994), the failure to implement on time seasonal closures through an adequate management plan caused a Conch fishery collapse in Venezuela. Laughlin et al. (1985; as cited by Rodríguez and Posada, 1994), reported densities of 4,600 conchs/ha within a no-take marine protected area in Los Roques, but of 800 conchs/ha outside the protected area. Besides, this is actually more than 1000 orders of magnitude higher densities within the Venezuela no take zone than what we found on the LPCNR's shallow habitats. Rathier and Battaglya (1994) informed that conch is depleted from areas shallower than 8m in Martinique. But, Twefik and Benes (2000) informed a 355-700% increase in the Queen Conch densities within a no take zone in Turks and Caicos. Therefore, carefully managed no take zones could actually be a successful tool during Conch fishery conservation and restoration efforts.

It is evident that lack of law enforcement by the PRDNER and lack of preventive patrolling and enforcement are allowing Conch pouchers to overfish Conch in Culebra, even within the LPCNR waters. MFRs elsewhere have been shown to be highly successful in recovering Conch populations, but only where there have been a serious compromise and political will by the local government. Also, another major concern in Culebra Island that needs to be addressed in detail is the effect of recreational overfishing. Hernández-Delgado (2000) documented the negative effects of recreational spearfishing in the structure of Culebra Island's coral reef fish and epibenthic communities. Recreational fishing is major factor contributing to the depletion of shallow-water conch stocks in Culebra Island and elsewhere in Puerto Rico. Therefore, it of paramount importance that final rules are established in Puerto Rico to implement the new Fisheries Law Number 278. This will establish new regulations, including recreational fishing licenses.

One aspect that deserves particular attention by the CFMC and the PRDNER is that Queen Conch, as well as the Fighting Conch have been observed in reproductive aggregations sometimes as early as the month of March. Rodríguez and Posada (1994) recommended a seasonal closure of Conch fisheries in Venezuela to be effective between the months of February and September of each year. This support our observation of aggregation behavior earlier than the closure season. This means that larger individuals are captured in shallower habitats, that part of the reproductive populations will not be able to mate or that part of it can only mate once or twice before capture. Thus, a second recommendation would be to include in future stock assessments a careful evaluation of the aggregation behavior of conch species. It is important to identify conch aggregation sites and prepare a geo-referenced map. This will be a powerful tool to designate potential Habitat Areas of Particular Concern (HAPCs) in the future. Also, such a product should be used by the PRDNER to enforce existing regulations. But definitely, the CFMC and the PRDNER should consider the alternative of expanding the Queen Conch seasonal closure.

The preliminary data from this study also points out that the densities of the other two conch species, *Strombus pugilis* and *S. costatus* are significantly low. Size class distribution of all of the conch species are skewed towards smaller categories (juveniles). Thus, we recommend the CFMC and the PRDNER to carry out a major assessment of the status of Milk and Fighting Conch populations in Puerto Rico, with the objective of evaluating the possibility of establishing seasonal closures for both species, simultaneously with the Queen Conch. This would be a precautionary measure to protect the Queen Conch from "accidental" harvest during the seasonal closure. We have observed that the three conch species are indiscriminately overfished in eastern Puerto Rico, even during the seasonal closure. Since, the regulation which requires that each landed conch should have the shell is not being enforced by the DNER, then, it is extremely difficult to determine which conch species meat is being landed. Therefore, a total conch seasonal closure will help to avoid that problem.

Another recommendation would be to designate as a HAPCs for Conch the entire LPCNR, Culebra Island, but also seagrass bed communities around Cayo de Luis Peña (portions not included within the LPCNR), Ensenada Malena, Cayo Dákity, Ensenada Honda, Bahía Mosquito, Playa Manzanillo, Puerto de Manglar, Playa Larga, Playa Zoní, Culebrita Island and

Cayo Norte. These seagrass beds have historically supported important Conch fisheries in Culebra and some of them are being threatened by land development and degraded water quality. Thus, the above recommended seagrass bed monitoring program should also include a conch population assessment. Any future assessment should include dusk and night diving to facilitate estimation of conch densities. Also, deepwater habitats should be equally assessed for conch in order to evaluate the status of adult populations.

CONCLUSIONS

This study constitutes the first quantitative assessment of the seagrass bed communities of the LPCNR. In general, seagrass beds were in better status within the LPCNR than outside. Traditional statistical methods were not completely able to geographically separate study sites, but multivariate analysis proved to be a powerful method to separate each site based on the seagrass community structure. Major differences in the seagrass bed community structure could be the combined result of natural fluctuations and the long-term effects of anthropogenic factors such as recreational boating activities and water quality degradation as a result of increasing land clearing activities and eutrophication associated to raw sewage dumping. There is a need to initiate a long-term monitoring program of ecological change of seagrass bed communities to address both, natural and anthropogenic changes. This program should also include a strong water quality assessment and monitoring. Land clearing activities in Culebra are a major cause of concern and will require a radical change in the land use patterns, planning and decision-making in order to prevent future chronic damages to seagrass and coral reef communities. Our final concern is water pollution resulting from raw sewage being emptied at Ensenada Honda, and from potential landfill leacheates that could be impacting seagrass and reef habitats within the LPCNR. A water quality study is urgently needed within the LPCNR to establish a baseline data bank.

The other major finding of this study was that conch shallow-water fishery stocks in Culebra Island have collapsed during the last four years. There are low densities or total absence of conchs in most seagrass habitats, and most of the individuals are juveniles. There was conch fishing, even within the LPCNR, during the Queen Conch seasonal closure. This suggests that the PRDNER has totally failed to implement fishery regulations in Culebra Island. A major change in the PRDNER attitude towards enforcing the Queen Conch fishery regulations and the fishing prohibition within the LPCNR is required. There is a need to establish a long-term monitoring program for conch populations in Culebra Island. Moreover, the CFMC and the PRDNER should evaluate the possibility of expanding the closure season to include the months of March to September, and to expand the seasonal closure to the other two conch species in the local fishery, the Milk Conch and the Fighting Conch. This would have the precautionary objective of avoiding “accidental” capture of Queen Conch during the closure. Finally, the PRDNER should have the political will of producing the final new fishery regulations in order to have more control over licensing, particularly of recreational fisheries which has been poorly assessed. Only through a radical change in the attitude of the PRDNER in Culebra Island we might be able to see a population recovery of shallow-water conch populations.

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APPENDIX

TABLE A1. Statistical power of the seagrass parameter data.

Parameters	BTMN-R	BTMS-R	BTA-R	BMO-R	PMA-R	Mean
% seagrass cover	**99.12%**	**98.3%**	**94.3%**	**93.4%**	**86.4%**	***94.3%***
% algal cover	69.6%	77.7%	**83.4%**	64.4%	76.4%	*74.3%*
% other cover	46.9%	66.0%	**92.0%**	60.9%	60.1%	*65.2%*
Density T. testudinum	**91.7%**	**91.4%**	**93.3%**	**92.8%**	**83.0%**	***90.4%***
Density S. filiforme	70.9%	**83.5%**	73.1%	47.0%	70.8%	*69.1%*
Density H. wrightii	-	-	-	39.3%	62.7%	*51.0%*
Density Calc. algae	57.6	69.0%	65.9%	61.1%	59.8%	*62.7%*
Biomass T. testudinum	**89.5%**	**88.9%**	**89.8%**	**87.2%**	76.7%	***85.5%***
Biomass S. filiforme	64.9%	**80.8%**	77.4%	62.6%	64.9%	*70.1%*
Biomass H. wrightii	-	41.3%	-	36.9%	16.1%	*31.4%*
Biomass Calc. Algae	45.3%	51.5%	47.2%	66.3%	63.1%	*54.7%*
Biomass Macroalgae	-	-	-	-	39.1%	*39.1%*

TABLE A2. Statistical power of *Thalassia testudinum* data.

Site	Leaf length	Leaf width	LAI
BTMN-R	**90.8%**	**95.2%**	72.3%
BTMS-R	**96.1%**	**97.5%**	**83.9%**
BTA-R	**94.1%**	**97.3%**	**81.1%**
BMO-C	**94.0%**	**95.7%**	**82.2%**
PMA-C	**89.4%**	**95.0%**	73.7%
Mean	***92.9%***	***96.1%***	*78.6%*

TABLE A3. Statistical power of Conch density data.

Site	***Strombus gigas***	***Strombus pugilis***	***Strombus costatus***
BTMN-R	-	67.7%	-
BTMS-R	0%	48.4%	-
BTA-R	0%	30.7%	-
BMO-C	-	59.7%	-
PMA-C	70.2%	-	-
Mean	*23.4%*	*51.6%*	-

TABLE A4. Statistical power of Conch maximum shell length data.

Site	***Strombus gigas***	***Strombus pugilis***	***Strombus costatus***
BTMN-R	-	**98.7%**	-
BTMS-R	-	**92.3%**	-
BTA-R	**95.4%**	**85.7%**	-
BMO-C	-	**97.8%**	-
PMA-C	**97.1%**	-	-
Mean	***96.3%***	***93.6%***	-

TABLE A5. Statistical power of Conch lip width data.

Site	***Strombus gigas***	***Strombus pugilis***	***Strombus costatus***
BTMN-R	-	66.7%	-
BTMS-R	-	48.4%	-
BTA-R	**88.9%**	30.7%	-
BMO-C	-	59.7%	-
PMA-C	56.8%	-	-
Mean	*72.9%*	*51.4%*	-

TABLE A6. Statistical power of the 1998 and 2002 Conch density data at BTMS-R.

Species	**1998**	**2002**
Strombus gigas	57.9%	0%
Strombus pugilis	**84.3%**	49.4%
Strombus costatus	0%	-

TABLE A7. Statistical power of the 1998 and 2002 Conch density data at BTA-R.

Species	**1998**	**2002**
Strombus gigas	59.2%	0%
Strombus pugilis	**87.0%**	30.7%
Strombus costatus	-	-

TABLE A8. Statistical power of the horizontal water transparency data.

Site	Power
BTMN-R	96.6%
BTMS-R	93.8%
BTA-R	100%
BMO-C	100%
PMA-C	92.3%
Mean	***96.5%***