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A NEW BASELINE FOR *DIADEMA ANTILLARUM*, *ECHINOMETRA VIRIDIS*, *E. LUCUNTER*, AND *EUCIDARIS TRIBULOIDES* POPULATIONS WITHIN THE CAYOS COCHINOS MPA, HONDURAS

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ABSTRACT: We investigated the density of 4 urchin species from 5 shallow reefs in the Cayos Cochinos Marine Protected Area in Honduras. Individual species density varied among reefs with total urchin density ranging from 3.2–7.9 individuals/m². *Echinometra viridis* (Agassiz, 1863) was the numerically dominant species (2.29/m²) followed by *E. lucunter* (Linnaeus, 1758) (1.76/m²) with *Eucidaris tribuloides* (Lamarck 1816) representing the fewest individuals (0.42/m²). Our results indicated that density of *Diadema antillarum* (Philippi, 1845) differed significantly among reefs (0.4–1.3/m², mean = 0.63/m²), but are three times greater than data collected 12 years after the mass mortality event of the 1980's. Size frequency analysis of *D. antillarum* indicated that test diameter also differed significantly among reef sites while correlation analysis showed that *D. antillarum* density was negatively related to both coral and algal cover. *Echinometra viridis* density, however, was positively related to coral cover ($p < 0.05$). When all urchins were combined, they showed a negative correlation with algal cover and a positive correlation with coral cover indicating a potential top–down effect within the reefs. While urchin densities remain relatively low compared to other sites in the Caribbean, the *D. antillarum* population appears to be recovering in this Honduran MPA. If overall grazer abundance continues to increase then they may limit macroalgae on reefs, thereby encouraging coral settlement.

KEY WORDS: sea urchin, coral reef, Caribbean, top–down

INTRODUCTION

Urchins are important herbivores in tropical reef systems. The devastating mass mortality of *Diadema antillarum* in the Caribbean during the 1980's (Lessios et al. 1984b) contributed to substantial shifts in reef community structure (Bruno et al. 2009, Mumby 2009). Substantial declines in coral cover have been observed throughout the region (Gardner et al. 2003), surmised to be the result of epidemic disease presence in corals, elevated algal growth due to the loss of herbivores, and natural and anthropogenic disturbances; thus leading to the potential of community phase shifts and loss of resilience (Norström et al. 2009, Hughes et al. 2010). While some areas have seen recovery in *D. antillarum* populations (Edmunds and Carpenter 2001, Ruiz–Ramos et al. 2011), others are still plagued with substantially reduced population sizes and high algal cover (Harborne et al. 2009). Although the loss of *D. antillarum* is certainly not the only cause of the decline in coral cover, the system shift discussed by Hughes (1994) and Bruno et al. (2009) demonstrates the necessity of understanding population dynamics and inventory of urchin populations throughout the Caribbean as part of a comprehensive approach for reef management. If recovery of *D. antillarum* populations leads to reduced algal cover, then perhaps some coral recovery is possible by opening up settlement sites (Carpenter and Edmunds 2006, Mumby et al.

2007a). However, *D. antillarum* is not the only reef echinoid. Lessios et al. (1984a) demonstrated that after the mass mortality, *Echinometra viridis* and *E. lucunter* showed significant increases in their density on Caribbean reef flats, potentially filling the grazing niche voided by the die–off, and currently they are a numerically dominant echinoid grazer on some reefs (Brown–Saracino et al. 2007).

One interesting observation regarding the recovery and distribution of *D. antillarum* after the mass mortality event relates to system protection. Harborne et al. (2009) reported that in regions that were open to fishing, *D. antillarum* were present, while in the preserve, they were absent. They surmised that it was the increase in larger predatory fish present in the protected area that led to predation pressure on *D. antillarum* and may have limited their recovery. Similarly, Brown–Saracino et al. (2007) reported their lowest urchin density in the Marine Protected Areas (MPA). However, Blanco et al. (2010) observed the opposite trend with significantly greater densities of *D. antillarum* within the marine reserve compared to non–reserve sites in Cuban reefs. As such, in MPAs the recovery of coral reefs may be impeded by low urchin densities unless alternate herbivores are present to eliminate substantial algal growth (McClanahan et al. 1996, Mumby et al. 2006, 2007a, Mumby and Harborne 2010).

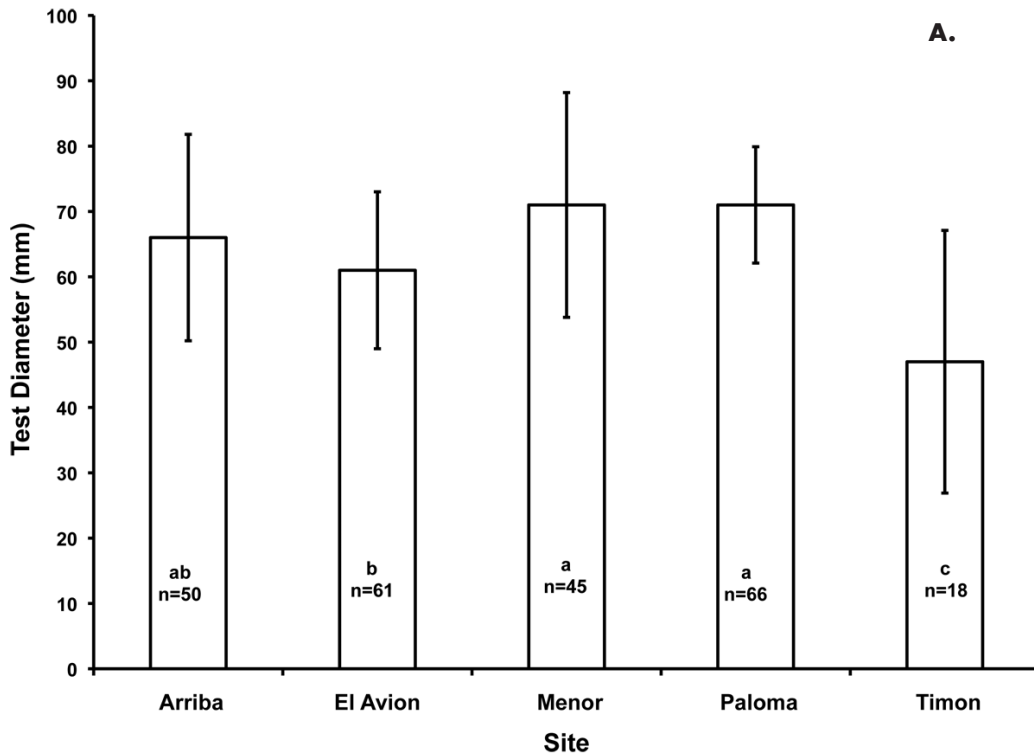
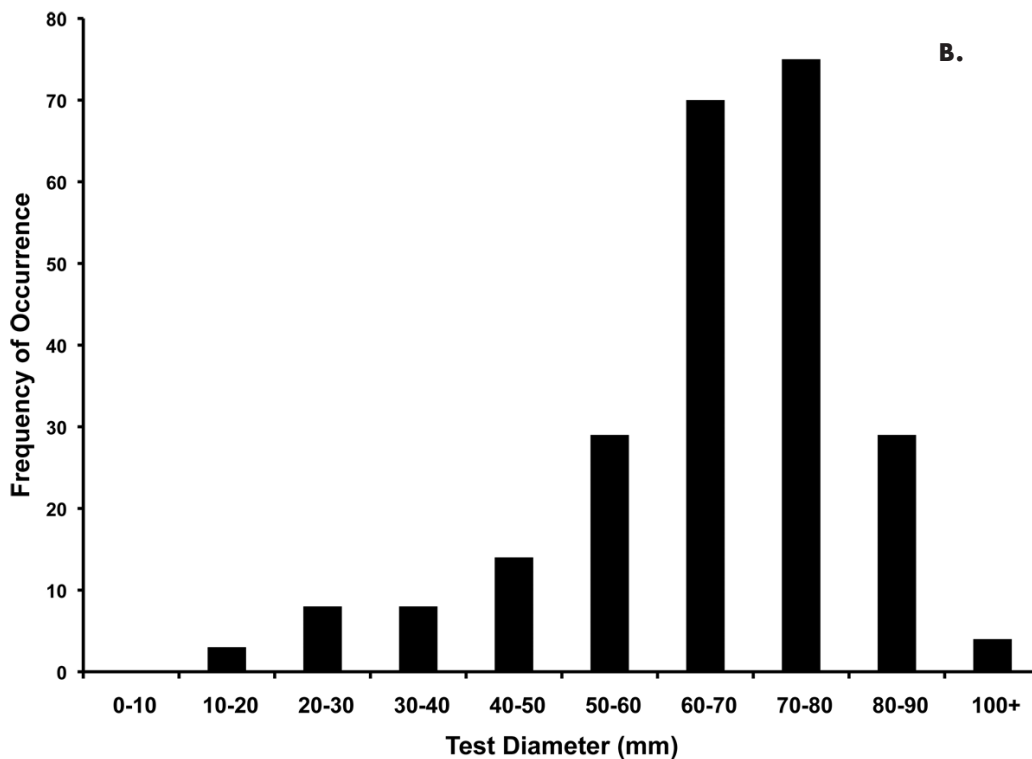


Figure 1. Urchin population demographics among the reefs sampled. **A.** Mean test diameter (\pm SD) among reef sites. Significant differences designated by unique letters. **B.** Combined *Diadema antillarum* population size frequency distribution. $n = 240$.



In Honduras, the Cayos Cochinos archipelago was designated a Marine National Monument in 1993 by the Honduran government. The aim of the designation is conservation and management of the regions' natural resources as well as economic stability. In particular, tight fishing regulations were enacted to protect and enhance fish populations. The islands comprise the very southern tip of the Mesoamerican

Barrier Reef System and were recognized as an important site of biodiversity in the area by the establishment of the MPA and designation as a Marine National Monument. The MPA has a no fishing core zone and a buffer zone with differing levels of fishing effort for select species at certain times of the year. The reefs surveyed in this study were all contained within the core zone. One emerging area of research in this

TABLE 1. Reef survey results among sites. Values represent the mean urchin density ($\#/m^2 \pm SD$) or the mean spatial coverage (% cover $\pm SD$). Differing letters next to means indicate significant differences among sites.

Site	<i>Echinometra viridis</i>	<i>Echinometra lucunter</i>	<i>Diadema antillarum</i>	<i>Eucidaris tribuloides</i>	Coral Cover	Algal Cover
Arriba	2.27 \pm 1.28 ^{ab}	1.71 \pm 1.59	0.06 \pm 0.09 ^b	0.13 \pm 0.07 ^b	29.0 \pm 15.5	30.1 \pm 9.3
Menor	0.97 \pm 0.55 ^b	2.53 \pm 1.43	0.18 \pm 0.19 ^b	0.37 \pm 0.12 ^b	18.6 \pm 8.1	39.9 \pm 9.1
Paloma	0.97 \pm 1.20 ^b	0.97 \pm 0.74	1.01 \pm 0.71 ^{ab}	0.29 \pm 0.15 ^b	10.8 \pm 5.0	33.5 \pm 23.1
El Evion	4.55 \pm 3.43 ^a	1.02 \pm 0.72	1.38 \pm 1.08 ^a	0.96 \pm 0.57 ^a	19.0 \pm 9.0	40.8 \pm 14.2
Timon	2.68 \pm 2.40 ^{ab}	2.59 \pm 2.85	0.54 \pm 0.53 ^{ab}	0.38 \pm 0.23 ^b	25.1 \pm 16.3	25.2 \pm 5.6
p-value	0.03	0.28	0.006	0.0007	0.11	0.26

MPA is urchin dynamics and the recovery of *D. antillarum* to this reef system. Lessios (1998) conducted the only estimate of urchin populations within the MPA by sampling a variety of habitats. In this study, we focus on reporting a new baseline for echinoid populations on shallow reefs.

MATERIALS AND METHODS

Research was conducted during the summer of 2009 on five reefs associated with the Cayos Cochinos MPA in Honduras: Cayos Arriba (15°57.5'N, 86°27.9'W), El Avion (15°57.2'N, 86°29.0'W), Menor (15°57.4'N, 86°30.4'W), Paloma (15°56.4'N, 86°30.0'W) and Timon (15°55.8'N, 86°32.6'W). Cayos Timon and Paloma represent shallow reef sites that were examined previously by Lessios (1998, his sites 1 and 5, respectively) for echinoid abundance. Snorkelers surveyed reef flats (1–3 m) for urchins, coral cover and algal cover using haphazardly selected 15 x 2 m transects laid out on the reefs (n = 6 surveys/reef site) during daylight hours. Transects were a minimum of 15 m apart. Abundance of the four most common urchins (*D. antillarum*, *E. lucunter*, *E. viridis*, and *Eucidaris tribuloides*) was counted along each transect and standardized to number/m². Prior to analysis, data were square root transformed to eliminate heteroscedacity and then compared among reefs using a one-way ANOVA. Coral and algal coverage were estimated visually by pooling five 1m² gridded quadrat samples (10 cm x 10 cm sub-grids) collected along the length of the transect. Specific sampling locations were selected *a priori* at distance intervals of 0–1 m, 3–4 m, 6–7 m, 9–10 m and 12–13 m along the transect. Initial samples were collected randomly (right or left), then each subsequent sample was collected on the alternate side. We recognize the limitations of visual estimates, but all samples were collected in the same manner and provide a context of spatial coverage. While species were identified to lowest reasonable taxa in the field, individual coral and algal taxa varied highly among reefs and transects and subsequently, these data were pooled into broad categories of coral and algal spatial coverage. Correlation analyses were then carried out on urchin species density, total urchin abundance, coral cover, and algal cover to assess potential relationships. Additionally, *D. antillarum* test diameter was measured in

the field using 40 cm stainless steel tree calipers on as many urchins that were present and measurable during a 50 min sampling period (n = 240) for each of the 5 reefs. Measurements of *D. antillarum* reflect a sequential survey measuring any individual encountered during the sampling period. We recognize that the smallest *D. antillarum* are highly cryptic, so care was taken to investigate potential refugia for small individuals. However, it is possible that the smallest individuals were not measured due to their cryptic nature. Urchin test diameter was then compared among reefs using a one-way ANOVA and discriminated using REGWQ pairwise comparisons (SAS®).

RESULTS

Densities of *D. antillarum* ($F_{4,25} = 4.62$, $p < 0.007$), *E. viridis* ($F_{4,25} = 3.13$, $p < 0.03$) and *E. tribuloides* ($F_{4,25} = 6.95$, $p < 0.001$) varied significantly among reefs investigated, while those of *E. lucunter* did not (Table 1). Additionally, our results showed that coral cover was relatively low and macroalgal cover high, but neither differed among reefs (Table 1). Our correlation analyses indicated a significant negative relationship between *D. antillarum* density and coral cover (Pearson $r = -0.43$, $p < 0.02$), but positive correlation between *E. viridis* and coral cover ($r = 0.36$, $p < 0.05$). No other factors showed significant relationships, but total urchin abundance was negatively correlated with algal cover ($r = -0.32$, $p > 0.08$) and positively correlated with coral cover ($r = 0.13$, $p > 0.4$). While *D. antillarum* density was low at all sites, test diameter was large and differed significantly among reefs ($F_{4,235} = 13.06$, $p < 0.0001$), with urchins from Menor and Paloma being larger than urchins collected at the other three sites (Figure 1A). The overall size frequency distribution demonstrates that for the Cayos Cochinos region, one large population peak occurs in the 60–80 mm size range, but the presence of small individuals (12 mm test diameter) indicates that recruitment is occurring (Figure 1B).

DISCUSSION

The mass die-off of *D. antillarum* left many Caribbean coral reef systems devoid of a primary herbivore (Lessios 1988), but in some cases reef communities showed increases of other echinoids possibly due to the abatement of competi-

tion (Lessios et al. 1984a). Regardless, the subsequent shift from a coral dominated system to one that reflects high algal abundance occurred in many regions of the Caribbean (Carpenter 1990). In 1995, Lessios (1998) assessed the reefs within the Cayos Cochinos MPA for the presence of echinoids. Based on his results, reef echinoid density was about $4.7/\text{m}^2$ and was dominated by *E. lucunter* ($3.8/\text{m}^2$) with lesser amounts attributed to *E. tribuloides* and *E. viridis* (0.47 and $0.24/\text{m}^2$, respectively), and *D. antillarum* ($0.19/\text{m}^2$). Our results showed similar densities of *E. tribuloides*, but 40% fewer *E. lucunter*. The biggest differences observed relate to a tripling of *D. antillarum* density and an order of magnitude increase in *E. viridis*.

Our results indicate that urchin populations are potentially recovering, but *D. antillarum* density is still far below the densities of other Caribbean reefs (Carpenter and Edmunds 2006) and total urchin density is far below those observed elsewhere (Brown–Saracino et al. 2007). For *D. antillarum*, Carpenter (1990) describes densities for St. Croix, USVI ranging from 5.8–13 individuals/ m^2 prior to the mass mortality, but recovery there has seen densities range as high as $5/\text{m}^2$ and this is similar to densities observed on Jamaican reefs (Edmunds and Carpenter 2001). This increase in density, with subsequent reduction in macroalgal cover, has allowed elevated coral recruitment (Carpenter and Edmunds 2006). While the loss of urchins is surmised to have been a proximal mechanism used to describe the phase shift between coral and macroalgal dominated communities, recent assessments demonstrate the complexity of coral reef structuring mechanisms (Bruno et al. 2009, Mumby 2009, Hughes et al. 2010). One challenge that we face is the limited historical data regarding the structure of this coral reef system. Our data suggest that coral cover is low and macroalgal cover relatively high (Table 1), but greater efforts in assessing all critical parts of the reef community are needed before generalizations are possible for Cayos Cochinos. Correlation analyses showed a positive correlation between *E. viridis* and coral cover, but a negative one for *D. antillarum*. While this may appear contrary to the concept that *D. antillarum* abun-

dance has a positive impact on coral recruitment, the lack of response may merely be an artifact of a system in recovery and further research should clarify this issue. However, total urchin abundance was negatively correlated with algal cover and positively with coral cover, so the combined grazing pressure by echinoids may be setting the stage for greater recovery. In fact, our density of *E. tribuloides* is similar to that reported in the Florida Keys by Chiappone et al. (2002), but density of *E. viridis* was 4–10x higher than theirs, and an order of magnitude greater for *D. antillarum*. Consequently, the lack of recovery seen in many reefs in the Florida Keys may relate to substantially lower combined grazer abundance, and not just the lack of *D. antillarum*, coupled with natural and anthropogenic disturbance (Hughes 1994, Mumby et al. 2007b).

While *D. antillarum* density was low at all sites, the collection of small individuals suggests that recruits are coming into the system. As such, recovery of *D. antillarum* on these reefs is possible. However, with no pre–mortality population density and size data, it may be impossible to infer whether *D. antillarum* populations have ‘recovered’ (*sensu* Lessios 2005). Additionally, the restrictions on fishing may change trophic structure of these reefs leading to changes in predation pressure on juvenile and new recruits, which may differentially impact the density and size distribution of *D. antillarum* (*sensu* Clemente et al. 2009). However, if this site is recruitment limited due to regional populations (*sensu* Miller et al. 2009), then population increases may not occur above some limited threshold until large scale recovery of *D. antillarum* occurs in the Caribbean. Carpenter and Edmunds (2006) demonstrate that when *D. antillarum* populations increase, coral recruitment can increase as well. Consequently, the regional recovery of *D. antillarum* may signal a starting recovery for Caribbean reefs (Mumby 2009, Mumby and Harborne 2010). As such, continued assessment of urchins, and specifically *D. antillarum*, coupled with algal and coral coverage in the shallow reefs is necessary to establish baseline conditions upon which future management and reef assessments can be made.

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