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NOTE

M. Chiappone · D.W. Swanson · S.L. Miller
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Large-scale surveys on the Florida Reef Tract indicate poor recovery of the long-spined sea urchin *Diadema antillarum*

Introduction

The 1983–1984 Caribbean-wide mass mortality of the long-spined urchin (*Diadema antillarum* Philippi) (Lessios et al. 1984) had severe consequences for many coral reefs (Hughes et al. 1985). Increases in macroalgal cover, declines in crustose coralline algae and reef corals, lower recruitment by corals, and greater sediment trapping by filamentous algae were well-documented changes resulting from the urchin mortality (Hughes et al. 1985; Liddell and Ohlhorst 1986; Carpenter 1990). The intensity of the effects has been especially great in those areas subjected to intense fishing (Hay 1981, 1984; Carpenter 1990). However, dramatic changes in coral reef community structure have also been documented in areas such as the Florida Keys (Porter and Meier 1992; Forcucci 1994) where Scaridae and Acanthuridae are not intensively fished (Bohnsack et al. 1994). Current patterns on many Florida Keys reefs include a dominance by algae, low coral cover, and low coral recruitment (Miller et al. 2002).

Historical population densities of *Diadema antillarum* in the Florida Keys were similar to Caribbean reefs (Bauer 1980), but declined after the 1983–1984 mortality event, even suffering a second mass mortality in 1991 (Forcucci 1994). In contrast to recent observations of *D. antillarum* population recovery on the north coast of Jamaica (Edmunds and Carpenter 2001; Moses and Bonem 2001), urchin densities remain very low in the Florida Keys. The consequences of the urchin mortality may be manifested in current widespread algal abundance patterns and illustrate that top-down controls on benthic community structure need to be considered (Chiappone and Sullivan 1997; Hughes et al. 1999). While there is consensus that Florida Keys reefs have changed in recent decades (Dustan and Halas 1987; Porter and Meier 1992), there is substantial debate concerning the mechanisms. Water quality concerns in Florida are often used to explain increased coverage of algae on reefs (Lapointe 1997), but other factors may be important (Hughes et al. 1999). This short communication summarizes density and size structure estimates of *D. antillarum* quantified at multiple spatial scales on the Florida Reef Tract during 1999 (Fig. 1). The synoptic surveys focused on four offshore coral reef and hard-bottom types from 4–12 m depth within and adjacent to no-fishing zones established in the Florida Keys National Marine Sanctuary in 1997. The surveys were part of a broader assessment and monitoring program to evaluate large-scale ecological patterns in reef benthos and the responses of small reef areas to protection from fishing pressure in the Sanctuary (Miller et al. 2002).

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Materials and methods

A two-stage, stratified random sampling design was employed following similar procedures outlined in Cochran (1977) to sample and obtain urchin density estimates in the Florida Keys. Surveys for *D. antillarum* were conducted during August to December 1999. Eighty sites representing four habitat types from 4–12 m depth were sampled over a 205-km distance (Fig. 1): (1) aggregate offshore (O) patch reefs; (2) high-relief spur and groove and low-relief hard-bottom on the shallow (S) fore reef (4–7 m depth); (3) high-relief spur and groove on an inner (I) reef tract near Key Largo; and (4) low-relief spur and groove and low-relief

Fig. 1. Location and names of no-fishing zones sampled and distribution of 80 offshore reef sites surveyed for *Diadema antillarum* in the Florida Keys National Marine Sanctuary during 1999
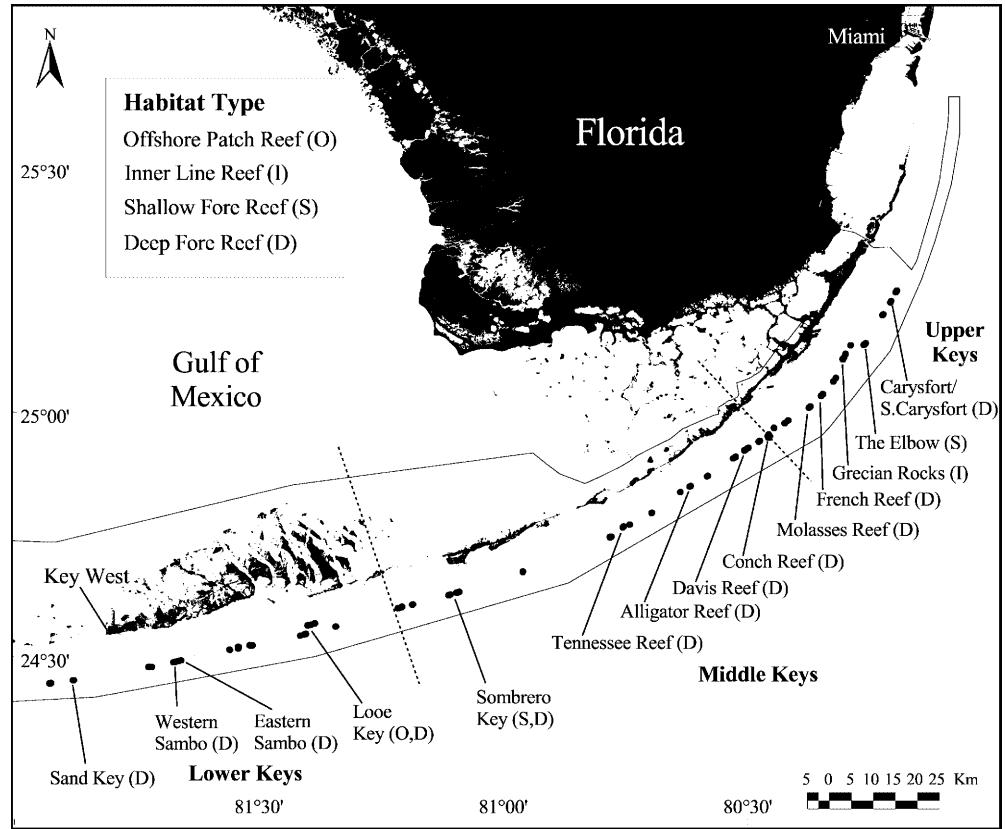
hard-bottom on the deeper (D) fore reef (8–12 m) (Florida Department of Environmental Protection 1998). Sites on the shallow (4–7 m) and deeper (8–12 m) fore reef were further classified into three regional sectors (upper, middle and lower Keys), because reef development and distribution in the Florida Keys are patterned after exchange processes between Florida Bay and the Atlantic Ocean, the position of the Florida Current relative to the platform margin, and the underlying Pleistocene topography (Chiappone and Sullivan 1997). Based on funding and logistical constraints, 16 of the 23 no-fishing zones in the Sanctuary were included. Spatial areas comprising each of the sampling strata were constructed in a geographical information system (GIS) using geo-referenced data on benthic habitat type (Florida Department of Environmental Protection 1998). Calculations of stratum areas and random allocations of sampling sites within strata were performed with the GIS. Two study sites, each with four paired transects, were allocated to each no-fishing zone by randomly selecting 200×200 m sites within each habitat stratum. Reference sites were randomly assigned by habitat type [according to Florida Department of Environmental Protection (1998) data] and regional sector. Samples were allocated among strata based on stratum size; that is, the area coverage of the habitat types targeted for surveys (Florida Department of Environmental Protection 1998).

At each site, four pre-generated, random sampling points using differential GPS were located. Two paired, 25-m transects were deployed at each sampling point per site, typically from inshore to offshore. A 0.4-m swath or band along each transect (each 10 m^2 in area) was surveyed for the number and size (test diameter) of *D. antillarum* (total area of 80 m^2). The area surveyed was delimited by using a 0.4-m scale bar. The same observer for the duration of the study completed the surveys. Mean density was computed for each pair of transects combined to derive a site-level density estimate for comparisons at three spatial scales: among habitat types, among geographic regions for shallow and deeper fore reef habitats, and between no-fishing zones and reference areas by habitat type and geographic region. Statistical comparisons of mean urchin density at these spatial scales were performed

by calculating confidence intervals (CI) based on the equation: $CI = mean \pm t_{[\alpha,\ df]} \times standard\ error$. Standard errors were estimated by the two-stage, stratified random sampling design (Cochran 1977) and confidence intervals were adjusted for multiple comparisons using the Bonferroni procedure (Miller 1981). The experiment-wise error rate was held at $\alpha = 0.05$ and the comparison-wise error rate was adjusted based on the number of multiple comparisons [comparison-wise error rate $= \alpha/c$, where $c = k\ (k–1)/2$]. Goodness of fit procedures using chi-square analyses were used to test the null hypothesis that the distribution of *D. antillarum* was independent of habitat type (Zar 1996), by comparing the number of individuals observed by habitat type with the expected frequency (i.e. product of total abundance and percent site allocation).

Results and discussion

Surveys of 80 locations spanning 200 km of the Florida Keys offshore reef tract yielded just 16 individuals of *D. antillarum*. Site-level densities were low (Table 1), with no individuals recorded from 69 of the 80 sites (86%) (Fig. 2, above). Eight of the 80 locations (10%) yielded only one individual per site. Although mean density was significantly greater on the deeper fore reef compared to the inner reef line habitat, this was due to the absence of urchins from four sites surveyed in the latter habitat type. Otherwise, no significant differences in mean density of *D. antillarum* were detected among habitat types, geographic regions, or between no-fishing zones and reference areas. The distribution of *D. antillarum* among the surveyed habitat types closely followed the allocation of sites in the stratified sampling design (Fig. 2, middle); that is, no significant habitat

associations were detected ($\chi^2 = 3.31$, df = 3, $0.50 > P > 0.25$). Thirteen of the 16 individuals (81.3% versus 76.3% of total effort) were recorded from the deeper fore reef, with the remaining distributed among offshore patch reef and shallow fore reef habitats. Test diameters (TD) ranged from 1.2–5.0 cm, with a modal size class (75% of all individuals) of 1.1–2.0 cm TD (Fig. 2, below). The mean (1 SD) test size was 2.1 (0.98) cm, and no individuals smaller than 1.0 cm or larger than 5.0 cm were found.

Table 1. Mean (± 1 standard error) densities of *Diadema antillarum* in the Florida Keys, based on surveys of eight 10-m^2 transects per site during 1999. Sites are arranged from SW to NE by benthic habitat type. Only those areas where urchins were encountered are listed from the 80 total sites sampled

Habitat type (no. of sites)	Regional sector classification	Management classification	Mean urchin density (no. per m^2)
Offshore patch reef			
Looe Key RO (2)	Lower Keys	No-fishing zone	0.013 ± 0.008
Shallow fore reef (4–7 m)			
American Shoal (2)	Lower Keys	Reference	0.006 ± 0.006
Deeper fore reef (8–12 m)			
Maryland Shoal (1)	Lower Keys	Reference	0.013 ± 0.013
West of Sombrero Reef (2)	Middle Keys	Reference	0.006 ± 0.006
Tennessee Light (2)	Middle Keys	Reference	0.025 ± 0.026
West of Alligator Reef (2)	Middle Keys	Reference	0.006 ± 0.006
Crocker Reef (2)	Middle Keys	Reference	0.006 ± 0.006
West of Davis Reef (2)	Middle Keys	Reference	0.013 ± 0.008
Conch Reef SPA (2)	Middle Keys	No-fishing zone	0.006 ± 0.006
Dixie Shoal (2)	Upper Keys	Reference	0.006 ± 0.006
North of Carysfort Reef (2)	Upper Keys	Reference	0.006 ± 0.006

Fig. 2. Frequency distribution in mean density (no. individuals per m^2) (*above*), habitat distribution (*middle*), and size (test diameter) distribution (*below*) of *Diadema antillarum* in the Florida Keys National Marine Sanctuary
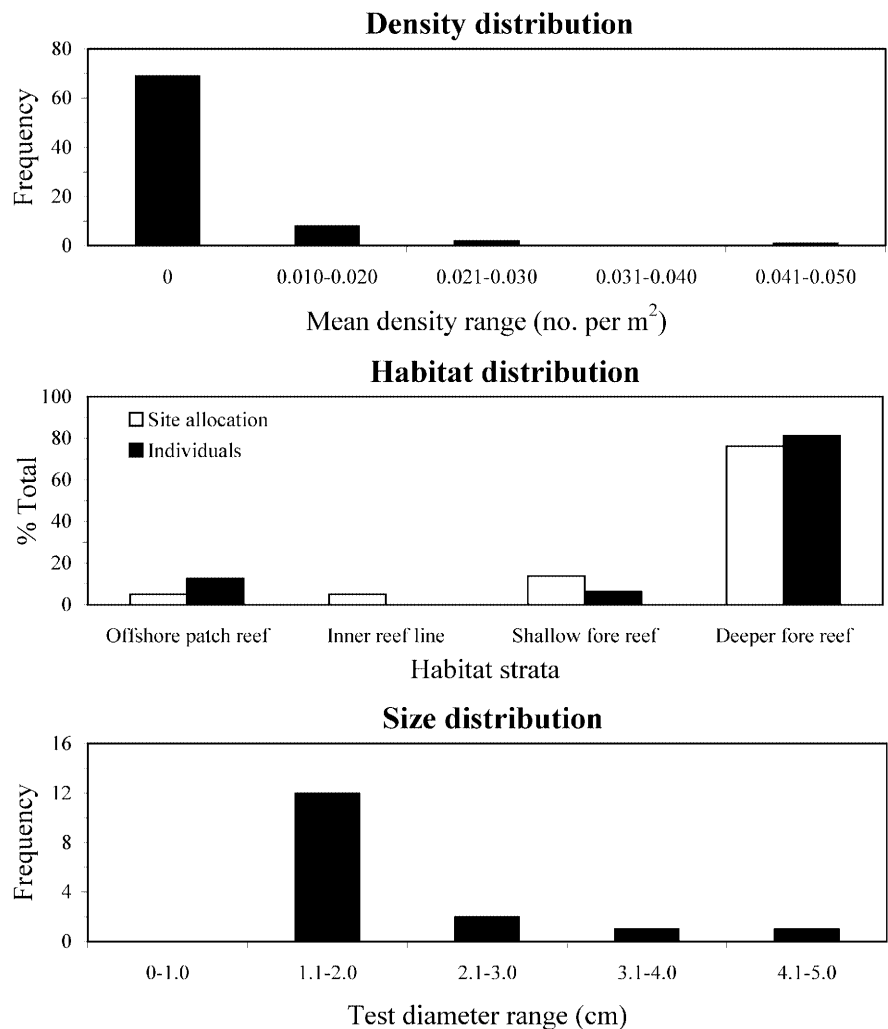
Despite nearly 15 years since the mass mortality of *D. antillarum*, populations have yet to recover to pre-1983 densities in the Florida Keys. Although our surveys were restricted to higher relief habitat types along the platform margin, *D. antillarum* densities were historically one to two orders of magnitude greater offshore in many of the same general reef areas that we surveyed during 1999 (Bauer 1980). Forcucci (1994) examined densities and recruitment in the Florida Keys 7 years after the 1983–1984 mortality and documented a second mass mortality during 1991. Prior to the 1991 mortality, *D. antillarum* densities were as high as 0.56 individuals/m^2 (range of 0.30–0.58 individuals/m^2 among seven shallow spur and groove reefs), but subsequently declined to <0.01 individuals/m^2. Surveys conducted by one of us during 1994 along the upper Keys platform margin in three reef types yielded low densities (<0.004 individuals/m^2) for four of the five locations where *D. antillarum* was found (M. Chiappone, unpublished data). Size distribution data before the 1991 event indicated a bimodal distribution, with a cohort in the 3.1- to 4.0-cm class and large individuals in the 6.1- to 7.0-cm TD class, with the larger size classes more frequent (Forcucci 1994). In contrast, no *D. antillarum* larger than 5.0 cm TD were recorded offshore during 1999, with 75% of all urchins 1.1–2.0 cm TD, indicating a summer recruitment event (Lewis 1966).

The prospects for recovery of *D. antillarum* in the Florida Keys are uncertain. Inadequate larval supply due to low population densities and hence lower fertilization success, poor recruitment surfaces due to increased algal cover, post-settlement mortality, or interspecific competition are important factors that affect recovery (Bak 1986; Lessios 1988; Levitan 1988). However, the size distribution data from 1999, similar to results from an earlier study (Forcucci 1994), indicate that recruitment pulses still occur in the Florida Keys. Anecdotal information from marine aquarium collectors indicates that juvenile *D. antillarum* are found in back reef rubble habitats, but are subsequently killed during storm events when rubble is moved about (K. Nedimyer, personal communication). Thus, larval supply may not be the primary bottleneck to recovery in the Florida Keys, but rather post-settlement survivorship, since few individuals appear to survive beyond 2 years of age, at least for rubble zones. Adult *D. antillarum* populations were recently discovered up-current of the Florida Keys in the Dry Tortugas (Chiappone et al. 2001) and in the western Caribbean (Edmunds and Carpenter 2001; Moses and Bonem 2001). Despite urchin density increases in some up-current locations, densities in the Florida Keys remain dramatically lower than historical numbers.

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