

The effectiveness of macroalgal reduction and *Diadema antillarum* addition in limiting macroalgal growth and facilitating coral recovery

D. R. Burdick

The Graduate School of the College of Charleston, 66 George St., Randolph Hall, Charleston, SC 29424

Abstract. The effectiveness of macroalgal reduction and *Diadema antillarum* addition in limiting macroalgal growth and facilitating coral recovery was investigated between June 2002 and July 2003. Two treatments, one involving the manual reduction of macroalgae, and another involving macroalgal reduction followed by the addition of *D. antillarum*, were each performed on a set of three replicate patch reefs off the island of Eleuthera, The Bahamas. Mean macroalgal cover was reduced to <2% immediately after macroalgal reduction, but neither treatment was effective at preventing macroalgal re-growth after 12 months. Mean juvenile coral density increased by more than 150% immediately after macroalgal reduction. No significant change in mean coral cover was detected for either treatment or the control between 0 mo and 12 mo and an increase in mean juvenile coral density observed on some reefs during this period cannot be attributed to the effectiveness of either treatment. The failure of the treatments in significantly limiting macroalgal growth and facilitating coral recovery was likely due to poor *D. antillarum* survivorship. The results suggest that the effectiveness of reef restoration efforts involving translocated or lab-reared *D. antillarum* may be limited on some reefs and may require multiple re-stocking efforts.

Keywords: Reef restoration, Macroalgal removal, *Diadema antillarum*, Bahamas, Patch reefs

Introduction

The vitality of coral reefs worldwide continues to decline despite heightened concern and the substantial efforts committed by hundreds of government agencies and nongovernmental organizations (Birkeland 2004). It may be necessary to supplement efforts to address climate change, overharvesting, coastal eutrophication, sedimentation and other threats with direct intervention at a local scale to help spur reef recovery in areas where these threats have been minimized, but where reef recovery has not been evident. The aim of this investigation was to assess the combined effectiveness of macroalgal reduction and the addition of *Diadema antillarum* Philippi on patch reef communities in the Bahamas. Macroalgal reduction, combined with the enhancement of *D. antillarum* population densities, may help facilitate localized phase-shift reversal on Caribbean/Western Atlantic coral reefs.

Materials and Methods

This study was conducted between June 2002 and July 2003 on a series of small patch reefs (4.9-8.3 m²) found in 10 to 11 m of water off the northwestern coast of Cape Eleuthera, Eleuthera, The Bahamas. Nine experimental reefs were selected at random from a set of 20 patch reefs. Separate sets (= three

replicates) of reefs received one of two treatments: macroalgal reduction (Treatment A) or macroalgal reduction combined with the addition of *D. antillarum* individuals (Treatment B). A third set of reefs did not receive any treatment and served as the control. Macroalgae were removed from the experimental reefs by hand. *Diadema antillarum* were collected from a nearby fringing reef and translocated to Treatment B reefs. The urchins were individually placed on the reefs with a target population density of approximately 4 m⁻². All Treatment B reefs were restocked with urchins at t = 5 mo, as no urchins survived from the initial translocation.

Surveys designed to monitor changes in coral and macroalgal cover, algae biomass, juvenile coral density, and coral health were conducted on the experimental reefs during the course of the 12-month investigation. Macroalgal and coral percent cover was quantified by filming non-overlapping transects placed across the length of each reef with a digital video camera at t = 0 (pre-treatment), t = 0^{post} (immediately post-treatment), t = 5, and t = 12 mo. The number (2-3) and length (2-4 m) of the transects varied depending on the size of reef. Percent cover estimates were obtained by analyzing consecutive, non-overlapping video frames using PointCount99[®]. Twenty-five randomly-generated points were used for

each frame; a minimum of thirty frames were analyzed for each reef. The percent cover value for each video frame was considered a sample; all samples from a single reef were pooled to obtain mean macroalgal and coral cover values for each reef. For comparison between treatments, mean macroalgae and coral cover values for each treatment were obtained by pooling all samples from all replicate reefs.

Algae were collected from four random sample quadrats (0.0625 m²) for each reef at t = 0 and t = 12 mo. All macroalgae and most turf algae were removed from each quadrat and placed into individual sample bags. The samples were rinsed and separated into algal groups: fleshy green, fleshy brown, and calcareous green. The samples were dried and then weighed using an Acculab v1-200 balance (200g capacity, 0.01 g readability). The mean algae dry-weight value for each reef was obtained by averaging the algae dry-weight values for all samples taken from each reef. Mean algae dry-weight values for each treatment were obtained by pooling all samples for all reefs that received the same treatment.

Attempts to conduct a visual census of all juvenile corals on Treatment A and Treatment B reefs were made at t = 0 mo, t = 0^{post} mo, and at t = 12 mo, while the control reefs were surveyed only at t = 0 mo and at t = 12 mo. Juvenile corals were located by closely inspecting the substratum; they were identified to species when possible. Measurements (mm) were taken at the longest axis of the polyp or colony. Only corals with 20 polyps or fewer and a diameter ≤ 4cm were counted. Mean juvenile coral count values for treatments were obtained by averaging the total counts for reefs receiving the same treatment at each sampling period. The total number of juvenile coral individuals was normalized by reef area and reported as density (colonies m⁻²). A visual census for adult and juvenile *D. antillarum* was conducted in conjunction with the juvenile coral survey.

Three 0.0625 m² permanent quadrats were established on each reef prior to treatment and filmed at t = 0 mo, t = 0^{post} mo, t = 5 mo, and at t = 12 mo, to examine fine-scale changes in the benthic community, focusing particularly on coral/algal interactions and sources of coral mortality and tissue stress.

An equal variance *t*-test was used to compare pooled treatment data between various time intervals. The equivalent non-parametric test (Mann-Whitney *U* test) was used when data did not satisfy the assumptions of normality and homoscedasticity or if the sample size was very small (n < 5). A one-way ANOVA or the equivalent non-parametric test (Kruskal-Wallis one-way ANOVA on ranks) was used to compare treatments and reefs at each time period; Tukey-Kramer Multiple Comparison test or

the equivalent non-parametric test (Kruskal-Wallis Multiple-Comparison Z-value Test) was used to determine the sources of significance.

Results

Macroalgal reduction lowered mean macroalgal cover to approximately 2% on both Treatment A and Treatment B reefs between t = 0 and t = 0^{post} mo (Mann-Whitney *U*, p < 0.001)(Fig. 1A). Treatment A reefs and the control reefs exhibited increases in mean macroalgal cover of 31.4% and 19.3%, respectively, between t = 0 and t = 12 mo (Mann-Whitney *U*, p < 0.001; equal variance *t*-test, p < 0.001). No change in mean macroalgal cover was detected on Treatment B reefs between t = 0 and t = 12 mo. Mean scleractinian coral cover on Treatment A and Treatment B reefs appeared to increase by 66.7% and 50% respectively, immediately following treatment, but these change were not significant (Mann-Whitney *U*, p = 0.078 and 0.091, respectively). No significant changes in mean scleractinian coral cover were observed for either treatment or the control between any other sampling periods (Fig. 1B).

In contrast to the decrease in algae biomass expected for Treatment A and Treatment B reefs, both treatments and the control exhibited a significant increase in mean algae dry weight after 12 mo (Fig. 1C). Mean dry weight values on Treatment B reefs had increased the most (72.3%), while increases of 41.6% and 60.4% were observed on Treatment A and the control reefs, respectively (Mann-Whitney *U*, p = 0.001, 0.002, and 0.009, respectively). Fleshy, green macroalgae dominated the algae biomass samples, with a mean percent composition across all samples of 93.1 ± 9.4%. Visual estimates suggest that *Microdictyon marinum* comprised the majority of the dry-weight for all the samples.

Mean juvenile coral density increased 286% on Treatment B reefs immediately after treatment (Mann-Whitney *U*, p = 0.05) (Fig. 1D). Treatment A reefs exhibited a 165% increase in mean juvenile coral density during this period, but the change was not significant at the 0.05 level (Mann-Whitney *U*, p = 0.092). Mean juvenile coral density on Treatment A reefs decreased 49.3% between t = 0^{post} and t = 12 mo (Mann-Whitney *U*, p = 0.05). Mean juvenile coral density on Treatment B reefs appeared to decrease by 55% during this period, but the change was not significant (Mann-Whitney *U*, p = 0.1). Mean juvenile coral density had increased by 95.7% on Treatment B reefs between t = 0 and t = 12 mo (Mann-Whitney *U*, p = 0.038). An apparent increase of 174% was observed on the control reefs during this period, but the change was not significant (Mann-Whitney *U*, p = 0.1). No significant increase was

seen on Treatment A reefs between $t = 0$ and $t = 12$ mo (Mann-Whitney U , $p = 0.329$).

No adult urchins were found on any of the experimental reefs at $t = 0$, $t = 5$, and $t = 12$ mo. No juvenile *D. antillarum* were found on any of the reefs prior to treatment; one juvenile was found on one reef undergoing Treatment A, immediately after macroalgal removal. Three juveniles, one on each of two reefs undergoing Treatment B and one reef undergoing Treatment A, were found at $t = 5$ mo, while none were found on any reefs at $t = 12$ mo.

Examination of the permanent quadrat video frames revealed evidence of the partial and full mortality of several coral colonies. Partial mortality and tissue discoloration as a result of macroalgal overgrowth or abrasion was evident on several colonies. White Plague appeared responsible for the full mortality of a colony of *Dichocoenia stokesii* on a Treatment A reef, and Red Band Type II was likely responsible for the partial mortality of a *Montastrea faveolata* colony on another Treatment A reef. Infestation by the boring sponge, *Cliona* sp., was also evident on several colonies.

Discussion

Mean macroalgal cover decreased markedly on both Treatment A and Treatment B reefs immediately after macroalgal reduction, but the low macroalgal cover and biomass did not persist on either set of reefs. Even though Treatment B appeared to be more effective than Treatment A in limiting an increase in macroalgal cover, Treatment B was not able to prohibit the re-growth of macroalgae. Even more striking than the increase in mean macroalgal cover exhibited by Treatments A and the control reefs between $t = 0$ mo and $t = 12$ mo was the increase in mean algae biomass exhibited by all treatments during this same time interval. This increase suggests that the treatments were ineffective at reducing algal biomass and even at preventing algal biomass from increasing during this period. The observation that algae biomass samples were dominated by fleshy green algae, and, in particular, by *Microdictyon marinum*, is in concordance with reports of extensive *M. marinum* blooms on several Bahamian islands (Woodley et al. 2000; Kramer et al. 2003; Peckol et al. 2003). Coral tissue stress and mortality caused by *M. marinum* has been observed elsewhere in the Bahamas (Dustan 1987; Kramer et al. 2003; Peckol et al. 2003). The likely increase in mean scleractinian coral cover on Treatment A and Treatment B reefs between $t = 0$ and $t = 0^{\text{post}}$ can be attributed to the detection of live coral that had been hidden beneath the thick macroalgal canopy prior to macroalgal reduction.

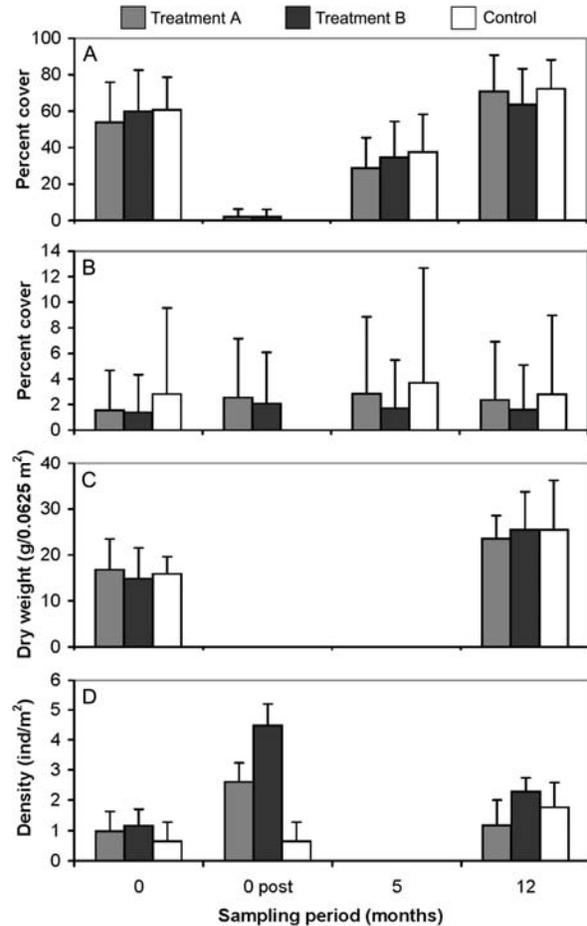


Figure 1: Mean macroalgal cover (A), mean scleractinian coral cover (B), mean algae dry weight (C), and mean juvenile coral density (D) at each sampling period. Algae dry weight data was not collected at $t = 0^{\text{post}}$ and $t = 5$ mo; juvenile coral surveys were not conducted at $t = 5$ mo. Error bars represent +SD.

The significant increase in mean juvenile coral density on Treatment B reefs, and the likely increase observed on Treatment A reefs immediately after treatment indicates that the thick macroalgal canopy covered a substantial number (approximately two-thirds) of juvenile recruits. The presence of such a large number of juvenile corals beneath the algal canopy strengthens the case for macroalgal reduction, which would increase their access to sunlight and food – if only temporarily. Some corals evidently recruited successfully despite high macroalgal cover, but their growth and survival was likely limited (Miller and Hay 1996; Edmunds and Carpenter 2001). The absence of any significant increase in juvenile coral density on Treatment A reefs and the apparent increase on both Treatment B and the control reefs between $t = 0$ and $t = 12$ mo suggest that the treatments were not responsible for any increase in mean juvenile coral density and that natural variability is a more likely explanation.

The decline in coral cover and vitality in Treatment A quadrats observed during this investigation can be attributed to active tissue necrosis resulting from at least two coral diseases/disease states, and to a lesser degree to macroalgal-related stress (e.g., abrasion, shading, and sediment stress). The active tissue destruction of the few remaining medium-to-large coral colonies by White Plague and Red Band Disease, combined with macroalgal overgrowth-associated tissue mortality, suggests continued decline in the vitality of the experimental site coral community.

Examination of the permanent quadrat video frames as well as observations made throughout the study also provide evidence of potentially significant infestation of coral heads by the boring sponge, *Cliona* sp. If, in fact, the incidence and severity of clionid sponge infestation on the experimental reefs were anomalously high, it could have been an indication of excess nutrient input (Rose and Risk 1985; Ward-Paige et al. 2005) and/or the result of the opportunistic infestation of colonies weakened or freshly killed by bleaching, disease, or other sources of stress (Schonberg and Wilkinson 2001).

The effectiveness of the combined treatment of macroalgal reduction and *D. antillarum* addition was likely limited by poor *D. antillarum* survivorship and may have been obscured by the onset of environmental factors more favorable to increased macroalgal growth. Despite restocking the experimental reefs with urchins five months after the initial treatment, none remained at the termination of the experiment. Based on the limited distribution and foraging of *D. antillarum* observed in other studies as a result of high levels of predation pressure (Carpenter 1984; Levitan and Genovese 1989), the urchins were expected to remain on the experimental reefs. Even when food supply is limited, *D. antillarum* tend to avoid the risks associated with migration and instead have the ability to reduce test growth rates and test size in more severe food-limited situations (Levitan 1988).

The low *D. antillarum* survivorship rates observed in this study are similar to those observed in a recent study that involved enclosing aggregates of wild *D. antillarum* around individual coral colonies in an attempt to improve the substrate quality for coral reseeded efforts and supplementing local *D. antillarum* populations with lab-reared individuals and monitoring their survivorship (The Nature Conservancy 2004). *Diadema antillarum* mortality was high in both components of the study, with approximately two-thirds of the enclosed wild urchins lost within the first 3 to 6 months of the project and 100% of the lab-reared urchins lost between 24 hours and several weeks after their release. Predation was

cited as the major source of the high mortality levels in both cases. Despite the low urchin survivorship observed in the Nature Conservancy (2004) study, high levels of algae biomass were removed by the wild urchins and coral spat were observed several months after collected coral spawn were seeded on the primed substrate.

In contrast to the low *D. antillarum* survivorship observed in this investigation and in the Nature Conservancy (2004) study, Nedimyer and Moe (2003) observed considerably greater survivorship (30% remained after 11 mo) in a similar effort that involved translocating juvenile *D. antillarum* from high settlement/high mortality reef crest rubble zones to deeper patch reefs. The significantly greater *D. antillarum* survivorship in the Nedimyer and Moe (2003) study was likely one of the major factors that contributed to the increased scleractinian coral cover, decreased macroalgal cover, and the increased juvenile coral density observed on their experimental reefs a year after the urchin additions.

The low *D. antillarum* survivorship observed in this study may have been controlled by several factors, such as the potential lack of suitable refugia, the presence of more robust populations of *D. antillarum* predators, or a combination of factors. The patch reefs used by Nedimyer and Moe (2003) were between 44 and 96 m², and may have provided more suitable urchin refugia.

In addition to the substantially greater *D. antillarum* survivorship observed in the Nedimyer and Moe (2003) study, the difference in benthic community compositions between the two sets of reefs prior to treatment may also have influenced the differing outcomes of *D. antillarum* additions of the Nedimyer and Moe (2003) study and this investigation. For example, scleractinian coral cover on the Nedimyer and Moe (2003) reefs was significantly higher (ranging from 5.5-14%, mean of 9.8%), while coral cover on the reefs examined in this investigation ranged from 0.7 to 5.6%, with a mean of 1.9%. Mean algae cover on the Nedimyer and Moe (2003) reefs was 54.3%, with well over half of the total algae cover on all reefs consisting of turf algae. Brown foliose algae cover ranged from 9-11%, while green foliose algae was less than 1% on all of the Nedimyer and Moe (2003) reefs. In contrast, macroalgal cover alone (not including turf algae) on the reefs in this study ranged from 37.2-71.0%, with a mean of 58.2%.

Also, *Microdictyon marinum* was not mentioned in the Nedimyer and Moe (2003) study. If it was present at all, it was only a minor component of the benthic community. This is in stark contrast to the dominance of *M. marinum* on the reefs of this study, which may have had a strong influence on the effect of the

experimental treatments. The rapid growth of *M. marinum* and the high canopy height achieved by the species may have a more profound affect on coral survivorship than other algae species. Its rapid growth, in combination with the poor survivorship of *D. antillarum*, may also have prevented the maintenance of an algal turf by overwhelming the urchins and the herbivorous fish. It is also interesting to note that although Nedimyer and Moe (2003) did not utilize macroalgal reduction to “prep” the substrate for the addition of *D. antillarum*, significant decreases in macroalgae and increases in coral cover were still observed on the experimental reefs.

Conclusions and Recommendations

The results of this study suggest that the proposed restoration technique in its present form was not effective at significantly reducing macroalgal regrowth or enhancing juvenile coral recruitment. The unexpectedly high flight and/or mortality rates for the translocated *D. antillarum* individuals are the most likely explanations for the less-than-expected performance of the restoration technique. One would expect a persistent *D. antillarum* population to be able to maintain an algal turf and prevent macroalgal regrowth, but the number of urchins needed to sustain such a population in the face of high flight/mortality rates would require a considerable supply of urchins and considerable human effort. Survivorship may be greater at other sites, as Nedimyer and Moe (2003) observed, and thus the success of the treatment may vary from site to site as a function of urchin predation levels, availability of suitable refugia, or some other factor.

Future restoration efforts using *D. antillarum* may benefit from abandoning patch reefs of such small sizes in favor of larger patch reefs – perhaps those with an area of 25 to 50 m². The larger size may enhance *D. antillarum* survivorship by providing more refugia. Macroalgal reduction on patch reefs of this size would still be feasible with a relatively small number of divers.

The development of a restoration technique involving macroalgal reduction and the addition of *D. antillarum* that was more successful at reducing macroalgae and increasing adult coral cover and juvenile coral density could be used concomitantly with efforts to manage local anthropogenic threats. Efforts to reduce anthropogenic impacts to coral reefs on a local level, and the implementation of localized restoration efforts should be continued and can be considered attempts at restoring to the reef-building corals the resilience required to acclimate or adapt to rapidly changing environmental conditions.

Acknowledgements

I would like to thank my thesis advisor, Dr. Phil Dustan, and my thesis committee members, Dr. James Carew, Dr. George Sedberry, and Dr. D. Reid Wiseman, for their advice, support, and patience. I also owe a debt of gratitude to my father, Roger Burdick, and Bill Davis, for their support with the fieldwork. I also wish to thank Chris Maxey, Director of the Cape Eleuthera Island School, and the Island School faculty and staff for assisting me with this project. Funding for this study was provided in part by a grant from SeaSpace 2002, a grant from the Explorer’s Club, the MES Program’s Faculty and Student Collaborative Research Grant, and the MES Program’s Travel Funding Grant

References

- Birkeland C (2004) Ratcheting down the coral reefs. *Bioscience* 54:1021-1027
- Carpenter RC (1984) Predator and population density control of homing behavior in the Caribbean echinoid *Diadema antillarum*. *Mar Biol* 82:101-108
- Dustan P (1987) Preliminary observations on the vitality of reef corals in San Salvador, Bahamas. in Curran H (ed) Proceedings of the Third Symposium on the Geology of the Bahamas: Fort Lauderdale, Florida, CCFL Bahamian Field Station, pp 57-65
- Edmunds PJ, Carpenter RC (2001) Recovery of *Diadema antillarum* reduces macroalgal cover and increases abundance of juvenile corals on a Caribbean reef. *Proc Natl Acad Sci U S A* 98: 5067-5071
- Kramer PA, Kramer PR, Ginsburg RN (2003) Assessment of the Andros island reef system, Bahamas (Part I: stony corals and algae). in: Lang JC (ed) Status of coral reefs in the western Atlantic: Results of initial surveys, Atlantic and Gulf Rapid Reef Assessment (AGRRA) Program. *Atoll Res Bull* 496:76-99
- Levitan DR (1988) Density-dependent size regulation and negative growth in the sea urchin *Diadema antillarum* Philippi. *Oecologia* 76:627-629
- Levitan DR, Genovese SJ (1989) Substratum-dependent predator-prey dynamics: Patch reefs as refuges from gastropod predation. *J Exp Mar Biol Ecol* 130:111-118
- Miller MW, Hay ME (1996) Coral-seaweed-grazer-nutrient interactions on temperate reefs. *Ecol Monogr* 66: 323-344
- Nedimyer K, Moe MA (2003) Techniques development for the re-establishment of the long-spined sea urchin, *Diadema antillarum*, on two small patch reefs in the upper Florida Keys. Year-end report provided to the Florida Keys National Marine Sanctuary
- Peckol PM, Curran HA, Greenstein BJ, Floyd EY, Robbart ML (2003) Assessment of coral reefs off San Salvador island, Bahamas (stony corals, algae and fish populations). in: Lang JC (ed) Status of coral reefs in the western Atlantic: Results of initial surveys, Atlantic and Gulf Rapid Reef Assessment (AGRRA) Program. *Atoll Res Bull* 496:76-99
- Rose CS, Risk MJ (1985) Increase in *Cliona delitrix* infestation of *Montastrea cavernosa* heads on an organically polluted portion of the Grand Cayman fringing reef. *Mar Ecol* 6:345-363
- Schonberg CHL, Wilkinson CR (2001) Induced colonization of corals by a clionid bioeroding sponge. *Coral Reefs* 20:69-76
- The Nature Conservancy (2004) Report: The Diadema Workshop. March 19-20
- Ward-Paige CA, Risk MJ, Sherwood OA, Jaap WC (2005) Clionid sponge surveys on the Florida Reef Tract suggest land-based nutrient inputs. *Mar Poll Bull* 51:570-579
- Woodley JD, Alcolado P, Austin T, Barnes J, Claro-Madruga R, Ebanks-Petrie G, Estrada R, Gerald F, Glasspool A, Homer F, Luckhurst B, Phillips E, Shim D, Smith R, Sullivan-Sealy K, Vega M, Ward J, Wiener J (2000) Status of coral reefs in the northern Caribbean and western Atlantic. in Wilkinson C (ed) Status of Coral Reefs of the World: 2000. Australian Institute of Marine Science, Cape Ferguson, Queensland and Dampier, Western Australia, pp 261-28