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Coral mortality, recovery and reef degradation at Mexico Rocks Patch Reef Complex, Northern Belize, Central America: 1995–1997

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Abstract

The 1995 coral bleaching event in the western Caribbean was the first reported episode that significantly affected the Belize barrier and lagoonal patch reefs. Bleaching was attributed to a 2 mo period of warm water temperatures above 30 °C. Near Ambergris Caye, barrier and patch reefs experienced up to 50% bleaching. At Mexico Rocks patch reef complex, the bleaching resulted in changes in reef health, community, and physical structure. Prior to the hyperthermal episode, patch reef surface area consisted of 47% healthy framework coral coverage, 12% secondarily colonized biotic coverage, 35% dead coral surfaces that were degraded by biological activity and physical erosion, and 6% cavities. six months after bleaching, most corals had regained their color, but, owing to coral mortality, areas of surface degradation had increased to an average 49% (p = 0.029 based on Kruskal–Wallis analyses). Eighteen months after bleaching, degraded surface areas expanded to 53% (p = 0.0366). Although re-coloring indicates rapid recovery for surviving corals, the persistence in dead coral surfaces suggests that reef skeletal structure recovery lags behind that of individual corals. Initial results of framework measurements indicate that bleaching events may result in an 'imbalance' in the carbonate production rate of coral reefs and produce mass wasting of the skeletal structure. Remapping of reef skeletal structure should establish quantitative measures for the long-term effects of bleaching on patch reef frameworks.

Introduction

Coral bleaching as a result of higher than average water temperatures commonly has been associated with El Nino/Southern Oscillation events as reported in the Pacific in 1982–1983, 1987, and 1998 (Glynn, 1988a, 1993; Viets, 1998). Elevated sea surface temperatures in the western Atlantic from August through October 1995 also produced widespread bleaching of corals from the Belize barrier and lagoonal patch reefs (Holden, 1995; CARICOMP, 1997). This event marked the first documented coral bleaching in this area (Stout, 1995). The Mexico Rocks patch reef complex, located 0.3 km seaward of Ambergris Caye in northern Belize, was severely affected by this thermal episode, when surface water temperatures in the shallow back-reef area in northern Belize increased to 32–34 °C (Sprowls, 1995). Aerial and underwater surveys of the bleaching event indicated that as much as 50% of the corals were bleached both at Mexico Rocks and on the barrier reef (CARICOMP, 1997). This paper presents the effects of the 1995 bleaching episode at the Mexico Rocks patch reefs, summarizes short-term coral recovery since that time, and describes long-term effects on patch reef skeletal structure.

Bleaching occurs when stony corals lose or expel all or a portion of their endosymbionts (zooxanthellae). Several factors have been implicated for this disruptive process, including disease (Kushmaro et al., 1966; Gleason & Wellington, 1993; Ritchie et al., 1993), increased UV radiation flux (Jokiel & York, 1984), hyposalinity (Goodbody, 1961; Goreau, 1964), increased sediment flux (Acevedo & Goenaga, 1986), pollution (Neff & Anderson, 1981), and temperature increases, some of which are possibly associated with global warming (Jokiel & Coles, 1990; Glynn, 1991, 1993; Smith & Buddemeier, 1992).

The ability of corals to recover their zooxanthellae after bleaching appears to be species-specific and related to their susceptibility to increases in water temperature, and recovery period can range from months to years (Hays & Bush, 1990; Holden, 1995). Research on bleached and then recovered Montastrea annularis in reefs in the Cayman Islands, for example, indicates that healing is gradual, and involves acquisition of a new population of zooxanthellae and restoration of their densities to normal, non-bleached levels (Hays & Bush, 1990). Additional consequences of bleaching include decreased skeletal growth, repressed gonad growth and reproduction, increased predation pressure on surviving corals, and increased mortality (Jokiel & Coles, 1977, 1984; Glynn, 1988b, 1990, 1993; Hughes, 1989; Brown & Suharsono, 1990; Goreau & MacFarlane, 1990; Szmant & Gassman, 1990). Occasional reports of framework deterioration have also been reported (Glynn, 1988b, 1993; Eakin, 1991).

Study area

Mexico Rocks patch reef complex is located on the outer shelf platform offshore of northern Belize, about 0.3 km seaward of Ambergris Caye, and 0.4 km to the lee of the platform-margin barrier reef. Dimensions of the complex are approximately 1.7 km in length and 0.5 km in width. It has been under consideration for preserve status by the Belize government, and has been the site of baseline research by the authors since 1988 (Mazzullo et al., 1992, 1993; McHenry, 1996; Burke et al., 1998). The complex includes approximately 100 individual patch reefs, which consist predominantly of the *Montastrea annularis* complex of coral heads that range in area from approximately 4–400 m². The largest of these reefs

are about 2.1 m high and have grown to within 0.5 m of mean sea level. The patch reefs have grown atop and on the flanks of a narrow, northeast-trending ridge of karsted Pleistocene limestone. Growth was initiated during the Flandrian transgression at about 3.5 Ka ago (Mazzullo et al., 1992, 1993; Burke et al., 1998).

Water depths immediately around the complex and between patch reefs average 2.7 m, and increase to about 4 m in a seaward direction before shallowing to about 0.9 m toward the barrier reef flat. Temperature, salinity, and pH were recorded at least once a year within and around the complex over the period from 1988 to 1998. Salinity of the water is constant at 38%, and daytime pH ranges from 8.0 to 8.4. Average daily surface ocean temperatures range seasonally from 27 to 29 °C. The semi-diurnal tidal range is less than 0.5 m and typical wave energy, which is qualitatively characterized as moderate (except during storms or unusually calm weather), is driven by onshore and seasonal easterly trade winds and, to a lesser extent, by tides. Some wave energy is input through passes in the barrier reef.

Methods

Twenty-three patch reefs ranging from 4 to 370 m^2 were geologically mapped and biologically surveyed (using line transect and area measurements) in 1990 for determination of coral-algal coverage, percentage of dead coral (herein called areas of degradation), percentage of cavities, and both linear and vertical dimensions of each patch reef framework (cf. Burke et al., 1998 for specific field methods and survey results). A rope line, calibrated at meter intervals, was placed along the long axis of each of the 23 selected reefs. At each meter section, both area measurements of biotic coverage, degradation and cavities, and the biota beneath the rope line were recorded on underwater slates for the length of each patch reef. A steel reinforcement bar calibrated at 0.3 m intervals was used for field measurements of horizontal and vertical dimensions at each meter interval along the rope line. Horizontal measurements included the width of each meter section along the length of the patch reef in meters. At each 1 m interval, vertical measurements were taken at the center of the reef, and at least two locations perpendicular to the long axis. This resulted in a minimum of three vertical measurements for each meter section across the top and flanks of each patch reef. To assure good correspondence between reef topography and contour mapping, wide or cavernous reefs required additional height measurements. All dimensional measurements from each patch reef were used to construct contour maps of each patch that also serve as base maps for annual monitoring of reef health. Yearly assessment of coral-algal coverage, percentage of dead coral, percentage of cavities on the reef surface, and changes in dimensions of the patch reefs, can be compared to these original base maps for each of the 23 patch reefs. This field technique provides a method to monitor the health of reef skeletal structure. An estimate of framework erosion can be determined by comparing original base maps to annual assessments of surface area degradation and cavities present as well as to vertical distance below mean sea level to the tops of the colonies. These data can be useful for monitoring long term, temporal changes in the skeletal framework and coral coverage on patch reefs, or any section of any reef.

Results and discussion

Pre-bleached parameters

Prior to the 1995 hyperthermal episode, patch reef surface area consisted of 47% healthy stony coral coverage, 10% soft coral/sponge coverage, 2% algae, 35% dead coral surfaces that were degraded by biological activity and physical erosion, and 6% cavities (Fig. 1). On the 23 measured patch reefs, the head corals Montastrea annularis (sensu strictu) composed 83%, and Diploria spp. and Dichocoenia sp. together composed 1% of total area of living reef-framework biota (coral species coverage). Other species of Montastrea (e.g., faveolata, franksi, and cavernosa) were observed in the complex, but were localized, and did not constitute a noteworthy portion of the measured patch reefs or of the complex. The remaining 16% of corals consisted of Agaricia spp., Porites spp., Acropora spp., and other branching scleractinids; and reefdwelling Millepora spp.



Figure 1. Patch reef composition. Weighted percentage by area of coral and non-coral biotic coverage, degradation, and surface cavities at the Mexico Rocks patch reef complex, 1988–1997.

Degraded areas on the framework resulting from biological and physical erosion averaged 35% of measured reef surface area (Fig. 1). Additional coral species (e.g., Agaracia spp., Acropora spp., and Porities spp.) and, locally, calcareous algae (e.g., Halimeda spp. and Amphiroa spp.), filamentous (turf) algae, and macro algae (e.g., Padina and Turbinaria) or encrusting and boring sponges, colonized these areas of dead coral, thereby increasing species richness on the patch reefs and comprising 12% of biotic coverage (Mazzullo et al., 1992, 1993; McHenry, 1996; Burke et al., 1998). The distribution of non-scleractinid coral biota was controlled by the amount and distribution of surface degradation and cavities (Burke et al., 1998). Commonly, localized growths of non-calcifying macro algae, turf algae, and encrusting sponges were restricted to large, degraded patch reefs, and were rarely found growing on medium or small patch reefs with little degradation. Calcareous algae and non-encrusting sponges grew in degraded regions among lobes of corals and skeletal cavities.

Cavities on the patch reefs averaged 6% of the complex (Fig. 1). Most of the medium and small clusters of head corals contained few cavities; larger patch reefs generally contained larger cavities. Because of the dominance of head corals, the reefs were clustered domes that had coalesced over time and grown to within 0.5 m of sea level. No significant changes in the geometric shapes of the patch reefs, heights below mean sea level, species coverage, degraded areas, and extent of cavities were noted from 1988 to 1993 (Burke et al., 1998). In fact, the 23 patch reefs in the complex were easily identifiable from year to year based upon these attributes.

Post-bleaching parameters – 1996

Increase in degradation, shift in species distribution, and presence of diseased corals

The patch reefs were re-surveyed in March 1996, 6 mo after the 1995 bleaching event. Evidence of active bleaching was minimal: less than 1% of the corals were bleached. Most of the previously whitened corals had regained their normal color, and presumably, their population of zooxanthellae. Rapid recovery of surviving corals after the bleaching event therefore is indicated.

To determine the extent of coral survival, we re-measured the area of degraded reef framework. Dead reef surfaces had increased on average to 49% of the reef frameworks (Figs 1 and 2). This significant increase (p = 0.029 based on Kruskal– Wallis analyses) was the result of coral mortality and indicates that the 1995 Caribbean bleaching event resulted in significant coral mortality on these patch reefs within 6 months. Most of these 'new' zones of degradation were already colonized by algae, a phenomena that is noted by many Pacific reef workers as a post-bleaching ramification (cf. Jokiel & Coles, 1977; Glynn, 1988a, 1993; Hughes, 1989; Brown & Suharsono, 1990; Goreau & Macfarlane, 1990; Szmant & Gassman, 1990; McCook, 1999).

Increase in standing crop of algae indicates a phase shift in biotic distribution on the reefs. Most of the degraded surfaces at Mexico Rocks had been colonized extensively, and in some cases, entire reefs were nearly overgrown, by noncalcifying turf algae. To further complicate recovery, macro-algae, including intertwining mats of Caulerpa racemosa and meadows of Turbinaria spp. and Padina spp. covered the patch reef surfaces. Before 1996 these algae were restricted to large, degraded portions of reefs in the complex. In 1996, nearly all the patch reefs supported macroalgae, and in some cases, entire reefs were nearly overgrown by Caulerpa racemosa. This increase in macroalgal abundance may have resulted from coral mortality, corals weakened by bleaching with inhibited natural chemical and physical defense systems (cf. Lang, 1973), and lack of algal herbivory. As a consequence, algal standing crop increased, and resulted in a dramatic phase shift in reef biotic distribution (cf. Jokiel & Coles, 1977; Glynn, 1988a, 1993; Hughes, 1989; 1994; Brown & Suharsono, 1990; Goreau & Macfarlane, 1990; Szmant & Gassman, 1990; McCook, 1999).

Coral bleaching has been implicated in the suppression of normal coral physiology including carbonate secretion (Glynn, 1993). The increase in



Figure 2. Patch reef degradation. Percentage of patch reef surface area degraded in 1988–1993, 1996, and 1997. Solid black bar is area degraded prior to bleaching (1993). Solid white bar is area degraded in 1996, 6 months after bleaching. Gray bar is area degraded in 1997, 18 mo after bleaching. The degraded surface area on the patch reefs increased significantly after the 1995 bleaching event.

dead reef surfaces may reflect these physiological complications. For example, corals weakened by bleaching may be unable to ward off pathogens or disease epizooids that ultimately lead to coral mortality. As noted by Glynn (1993, p. 5), increases in coral diseases may be a result of stressors placed on corals by bleaching.

By March, 1996, previously unrecorded coral diseases such as White Plague and Pox were present on the reefs, but were sufficiently uncommon to warrant a category on survey transects. Evidence of white band and plague diseases were present in the form of fresh white spotting of coral surfaces that were devoid of living corals. Black band disease was also present, but has been a persistent, contained disease in the complex for many years.

Post-bleaching - 1997

Increase in degradation and change in patch reef geometry

The patch reefs were again resurveyed in 1997, 18 mo after the bleaching event. At this time, less than 1% of the corals, including the dominant frame builder, *Montastrea annularis (sensu strictu)*, displayed signs of active bleaching. An 'apparent' shift in morphotypes to *M. franksi* was noted but not quantified on several reefs. Localized areas of diseased corals were present in the complex as in the previous year.

The average amount of surface degraded area, however, had increased from 49% to 53% in 1996. This increase in degraded areas from 1996 to 1997 (Fig. 1) is statistically significant (p = 0.0366), and suggests that coral mortality continued to occur throughout 1996–1997. Based upon these results, several implications can be made. First, re-coloration of corals is only a partial measure of reef recovery and occurs rapidly after waters cool. Secondly, recovery of the reef framework lags behind that of individual corals, and may be at risk because fewer corals are producing calcium carbonate due to coral mortality. Evidence for framework decline was present by 1997.

Patch Reefs 15 and 7 (87.7 and 4.2 m^2 in area, respectively), which are located in different areas of the complex, had collapsed and all but disappeared (Fig. 2). A less dramatic, but typical example of skeletal structure collapse is patch reef

13, located in the northern portion of the complex. In 1993, mapping of this patch reef indicated that the total volume of the framework was 26 m³. Remapping of the patch reef in 1997 indicated that the total volume of the reef was 24 m³ which represents an approximate loss of about 2 m³ of carbonate. This decrease in volume was attributable to reduction in the height of the reef below mean sea level. In essence, the large domed Montastrea annularis head that comprised the reef framework had collapsed, and produced an erratically cratered reef topography. Similar signs of mass wasting were present throughout the complex such that patch reefs easily recognizable through geometric shape by the authors in previous years were unrecognizable except by map location in 1997. A study to quantify the significance of this carbonate loss is underway.

Conclusion

Although reports on global coral health suggest that coral reefs remotely located away from centers of human population are not deteriorating (Pennisi, 1997), near shore reefs like Mexico Rocks are more typical of the patch reefs associated with increasing eco-tourism, harvesting, and onshore development. Conditions here are generally well disposed for reef growth; however, lagoonal patch reefs at Mexico Rocks are subjected to more variable environmental conditions and anthropogenic stresses because of their proximity to shore than are remotely located reefs. These shallow water patch reefs were among the first to succumb to bleaching, and their recovery time was slower than those corals that comprise the barrier reef (CAR-ICOMP, 1997).

At Mexico Rocks, the sequence of postbleaching events that was observed on the patch reefs includes (1) significant increase in coral mortality as indicated by dead coral surface coverage and diseased corals, (2) phase shift in biotic distribution, and (3) initial deterioration of reef framework. Re-coloration of surviving corals developed soon after waters began to cool. Deterioration of reef frameworks, however, may be a long-term process.

Based on radiocarbon age dates, patch reefs at Mexico Rocks have grown and diversified for hundreds of years (Burke et al., 1998). In contrast, only a few years were required to initiate reef deterioration after living reefs were subjected to a severe bleaching event, the predictable results of which were coral mortality and algal increase. When corals do not produce calcium carbonate exceeding or in equilibrium with reef erosional processes, reef skeletal structures may undergo mass wasting by biological and physical degradation. Analogous reef skeletal structure demise was observed in the Pacific as a result of the 1982–1983 El Nino by Glynn (1988b, 1993) and Eakin (1991). Further study of these patch reefs is necessary to determine the significance and permanency of framework mass wasting.

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