DOES PROTECTION CULTIVATE MORE RESILIENT REEFS?

ASSESSING THE LONG-TERM EFFECTS OF BELIZE’S NO-TAKE MANAGEMENT ZONES ON THE POST-DISTURBANCE RECOVERY OF CORALS

by

Clare Fieseler

Dr. Larry Crowder, Advisor

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2010

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MP Advisor
ABSTRACT

Coral reefs have emerged as one of the ecosystems most vulnerable to climate variation and change. Under the current trends, disturbance events are likely to increase in rate and severity. It is critically important to create management strategies that enhance the ability of coral reefs to absorb shocks, resist phase-shifts, and regenerate after such perturbations. This project assesses the capacity of no-take management zones to foster coral resilience in Belize in the 10 years after a major disturbance.

In 1998, the Belize Barrier Reef Complex (BBRC) experienced bleaching and hurricane events that effectively halved coral cover. Using video-based reef quantification, this project builds on a robust dataset describing benthic composition immediately before and at three sampling intervals after these major disturbances.

The results of this Master’s Project reveal that protection offered by no-take zones (NTZ) has no detectable effect on changes to benthic composition. Coral assemblages show no long-term recovery on either NTZ or fished reefs. As a result, macroalgae cover increased significantly, perhaps past certain resiliency thresholds. Insufficient protection may be attributed to design factors related to size, proximity to other stressors, and isolation. The results make clear that Belize’s reefs are changing at an increasing rate away from desirable ecological baselines. Conservation and government leaders in Belize are thus urged to look beyond purely spatial options in crafting tools for reef resilience.
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STATEMENT OF THE PROBLEM

There is uncertainty surrounding the resilience of coral reef ecosystems to major disturbances. Under the current trend of changing climatic conditions, disturbance events (i.e. bleaching, diseases, hurricanes) are likely to increase in rate and severity. It is critically important to determine if coral reefs are able to absorb shocks, resist phase-shifts, and regenerate after such perturbations (Obura 2005). The challenge, then, is to apply these findings for viable management strategies. In recent years, remote resilient reefs have served as living laboratories to study the ecological characteristics that enhance resilience to ecosystem-level changes (Graham et al. 2008; Sandin et al. 2008). Less-remote reefs, however, are ecologically altered, susceptible to multiple stressors, and respond much differently to disturbance (Knowlton & Jackson 2008). Degraded regions, like the Caribbean, serve as ground-zero for testing management strategies for resilience-based results.

In the western Caribbean, Belize is often considered a success story for coral reef management; it hosts an extensive and mature network of marine protected areas (MPAs) (GEF/UNDP 2008; IUCN/WCMC 1996; WCS 2008). These protected areas may serve as sufficient tools to foster resilience. In the context of hurricane and bleaching events, MPAs can be tested for efficacy by examining ecological responses in post-disturbance recovery. As revealed by Selig and Bruno (2010), post-bleaching recovery of corals in the Indo-Pacific is significantly related with MPA status. In the Caribbean, unfortunately, the actual likelihood for coral recovery after acute declines is small and the process is lengthy. In 68 Caribbean monitoring sites, there is no evidence of recovery to a pre-storm or pre-bleaching coral state for at least eight years after impact (Gardner et al.
The general impairment of the regeneration potential on Caribbean coral reefs is a great challenge to reef managers. At the same time, it is also cause for a highly rigorous investigation of promising management tools.

**PROJECT GOALS & CONTEXT**

In 1998, the Belize Barrier Reef Complex (BBRC) was significantly impacted by a combined bleaching and hurricane event. Scientists observed an overall loss of 48% hard coral cover, with certain patch reefs experiencing a loss of up to 75% (Aronson et al. 2000; Kramer & Kramer 2000; McField 2001). As of 2010, it remains unclear if coral assemblages are on a trajectory of recovery from these major disturbances. By protecting the overfishing of herbivores, “no take” management zones (NTZs) established before 1998 should afford areas of the BBRC with more system resilience, thus aiding in coral recovery (Bellwood et al. 2004; Selig & Bruno 2010). MPAs have been assessed as tools for fisheries management (Crowder et al. 2000; Halpern & Warner 2002; McClanahan & Arthur 2001), but MPAs have not been sufficiently assessed as tools for mitigating ecosystem responses after major disturbances. This project is part of larger study to determine whether the protection status afforded by Belize’s NTZs is accelerating long-term coral recovery.

The specific goal of this analysis is to determine significant differences in the benthic functional groups between three sets of paired study sites: NTZs and fished (unprotected) reefs. Based on the literature, it is expected that NTZ areas would exhibit a significantly higher coral cover, lower macroalgal cover, and higher cover of recruitment surface, like crustose coralline algae (Baker et al. 2008; Box & Mumby 2007; Burkepile
It is unclear how sponge cover may respond to protection during post-disturbance recovery. Nonetheless, quantifying changes in sponge cover can help discern the comparative changes between functional groups. For the purpose of the larger study involved, “benthic composition” represents one of four key variables used to infer resilience to change. “Coral recruitment,” “diversity,” and “herbivore abundance” complete the indices portfolio.

The Healthy Reefs Initiative (HRI), a program lead by the Smithsonian Institute, sponsored this project. Its partners represent researchers, donors, and conservation organizations based locally and internationally. From 1995 to 2000, HRI Director Dr. Melanie McField conducted the first large-scale quantitative description of reef community structure in Belize (McField 2001). This dataset contains the most comprehensive baseline for reef ecology immediately before the 1998 bleaching event and Hurricane Mitch. It serves are the baseline for this project and a similar project completed in 2005 by Ms. Nadia Bood of the World Wildlife Fund. The results of this project are intended for a HRI publication “Coral Reef Report 2010” and will be used to support HRI’s strategic goal of expanding the area of BBRC under MPA status (McField 2010).

HISTORICAL & COUNTRY CONTEXT

In 1998, scientists observed Belize’s most severe bleaching event to date (Carilli et al. 2009); soon after, a category-five hurricane brought severe wave action and flood-induced nitrification of the reef complex. The combined effects devastated reef
community structure, primarily through loss of coral cover. The reef’s most dominant and storm-tolerant coral species, Monastrea annularis was significantly impacted by bleaching, resulting in an absolute 5.6% decrease in hard coral cover after bleaching (McField 2001). The significant losses of the more structurally delicate species, Agaricia tenufolia and Acropora palmata, can be attributed to both events. Diversity loss accompanied coral loss, leading researchers to postulate a subsequent phase-shift similar to that observed on Jamaica’s reefs (Gibson et al. 1998; Hughes 1994; Kramer & Kramer 2000). The events that catalyzed Jamaica’s shift from a coral-dominated to algal-dominated benthos were temporally linked (i.e., hurricane and reduced herbivory) (Hughes 1994).

On the other hand, reefs are dynamic systems and are capable of recovering from acute disturbances if other perturbations are minimized (Connell 1997). The protection from fishing afforded to Belize’s NTZs increase or retain an abundance of herbivorous fish. Since herbivorous fish regulate algal cover on reefs (Mumby 2006b, 2009a), they promote the availability of area suitable for coral recruitment (Bellwood et al. 2004; McClanahan & Arthur 2001). In this way, Belize’s NTZs could have a positive effect on the abundance of corals.

When the Government of Belize initially created its NTZs, they were purposed as tools for fisheries management (Cho 2005b). Yet, an overwhelming amount of literature devoted to MPA-related research in the past decade suggests that NTZs are effective tools for reef conservation at the ecosystem level (Halpern et al. 2010). In the scientific literature, most scientists chose fish biomass as a response variable for MPA efficacy. Yet, little work has been devoted to clarifying the effectiveness of NTZs in promoting the
recovery of corals after a major disturbance and on a local level (Graham et al. 2008; Selig & Bruno 2010). This project presents a robust assessment of NTZs in this long-term capacity by using a baseline dataset of before-and-after disturbance affects.

Ultimately, the ecological recovery and resilience of the reef ecosystem is absolutely critical to the economy of Belize and the livelihoods of its coastal communities. The pristine condition of Belize’s corals in the 1990s attracted much attention from international divers, driving the development of dive-based tourism. Notably, the World Heritage Convention advisory attributed the BBRC’s global uniqueness to its recognition as a “international underwater attraction(s)” (IUCN/WCMC 1996). It is clear that dive-based tourism has contributed significantly to Belize’s economic growth; tourism is currently the number one foreign exchange earner for the country (Agency 2009). Fishing is also an important Belizean livelihood, especially as an increasing proportion of landings are reaching international markets. In 2002, for instance, the Fisheries Department valued marine export earnings at $72,587,810.34 which contributed to 7% of Belize's GDP. Appropriate management tools for Belize must be identified to ensure ecosystem resilience – and the sustainable use -- in the face of future climate-related disturbances.

STUDY SITES

Largely developed in the 1980s, the MPA network in Belize was envisioned as the central management strategy for the BBRC. As previously mentioned, no-take zoning was established in many MPAs to regulate fishing activity. Focus later shifted toward a broader multi-sectoral approach for marine and land-based activities (Cho 2005a; Sobel
The MPA network was integrated into the Integrated Coastal Management (ICM) program in the mid-1990s. This relationship was later formalized by the 1998 Coastal Zone Management Act, in which the major objectives for Belize’s MPAs were clearly defined:

*The major objectives of marine protected areas are for the conservation of ecosystem and species diversity, protection of commercially valuable species and the management of tourism and recreational activities.* (Coastal Zone Management Authority and Institute 2002)

The ICM managerial body – the Coastal Zone Management Authority (CZMA) – has reported significant problems with integrating the MPA system under the ICM program. Of note, CZMA staff has publicly reported on problems related to community involvement, sustainable financing, and disjointed authority (Cho 2005a).

A network of 18 MPAs currently exist in Belize, including eight marine reserves. At the time of data collection, six MPAs had designated NTZs. The following three serve as the focus of this project: Bacalar Chico Marine Reserve, Hol Chan Marine Reserve, and Glover’s Reef Marine Reserve. These study sites were originally chosen by Dr. McField for the 1997 baseline study based on their length of protection (Table 1). The size and age of the NTZs vary, but the zoning regulations for all three areas prohibit extractive activities.

The NTZs in Hol Chan and Glover’s Reef Marine Reserves were under a management plan prior to the 1998 hurricane and bleaching events. The NTZ in Bacalar Chico was legally implemented with a zoning plan in 2001. Gibson et al. notes that reserve staff had been “informally enforcing the zone regulations for several years” despite the legislative lag (Gibson et al. 2004). For ease of comparison, Bacalar Chio is treated as consistently protected throughout the recovery (1999-2009).
Table 1. MPAs containing the NTZs surveyed for this study.

<table>
<thead>
<tr>
<th>Marine Protected Area</th>
<th>Year Established</th>
<th>Marine Area (ha)</th>
<th>Total Area (ha)</th>
<th>No-Take Zone (ha)</th>
<th>Age of NTZ in 2009 (yrs)</th>
<th>% Marine as NTZ</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bacalar Chico</td>
<td>1995</td>
<td>6,303</td>
<td>11,487</td>
<td>1,699</td>
<td>14</td>
<td>26.95%</td>
</tr>
<tr>
<td>Hol Chan</td>
<td>1987</td>
<td>1,452</td>
<td>1,545</td>
<td>273</td>
<td>22</td>
<td>18.8%</td>
</tr>
<tr>
<td>Glover’s Reef</td>
<td>1993</td>
<td>32,876</td>
<td>32,876</td>
<td>7,226</td>
<td>16</td>
<td>21.98%</td>
</tr>
</tbody>
</table>

Figure 1: Map of the 18 marine protected areas in Belize. The green box indicates the region of focus for this project. This map is sourced directly from the Government of Belize’s Integrated Coastal Management Authority and Institute (http://www.coastalzonebelize.org/maps, 2010). Figure 2 contains a higher resolution map of this enclosed area and indicates paired study sites.
Table 2. GPS locations of the 300m$^2$ sites surveyed. Paired study sites are indicated by adjacent shaded cells. Sites within NTZ in light gray, fished sites in dark grey.

Figure 2. Spatial reference insert from Figure 1. Location of NTZ sites indicated by hatches. Paired control sites indicated by diamond points.

BACALAR CHICO MARINE RESERVE

Established in 1995, Bacalar Chico Marine Reserve encompasses a section of the barrier reef with high relief spur-and-groove formations, a number of patch reefs and seagrass beds (McField 2001). In relation to the other marine reserves in this study, NTZ is of moderate size: 1,699 ha$^2$. The reserve borders Mexico, having associated reef tract on the east and west end of Chetumal Bay. Unique to the BBRC, reef tracks in this reserve mark the only location where the living reef touches coastline. This distinct feature locates reef organisms proximate to land-based sources of sedimentation and toxins.
The reefs within and surrounding the reserve are susceptible to pollution from Chetumal Bay waters, illegal fishing, bleaching, and diseases. Gibson et al. (2004) report that management has been effective in realizing higher fisheries stocks within NTZ (Gibson et al. 2004). In 2000, McField evaluated its comparative management effectiveness in relation to other MPAs on the BBRC; her findings suggest that that management in Bacalar Chico Marine Reserve is less effective than neighboring Hol Chan Marine Reserve.

HOL CHAN MARINE RESERVE

Established in 1987, this reserve encompasses 6475 ha of protected mangroves, seagrass beds, and coral reefs. Fisheries stocks are managed by a 273 ha NTZ, the smallest reserve sampled in this study. The intermediate to deep fore reef in this area is characterized by high-relief spur- and-groove formations (Gibson 1986; McField 2001). Historically, Hol Chan has been the most successful in increasing fish stocks (Gibson et al. 2004). Like Bacalar Chico, however, its reefs are susceptible to local impacts from tourism, pollution, and development.

GLOVER’S REEF MARINE RESERVE

Established in 1993, this reserve encompasses an entire atoll 25 km east of the barrier reef. At the time of sampling, it contained the largest NTZ in the country: 2,733 ha². In the geomorphologic sense, Glover’s Reef Marine Reserve is very different from Hol Chan and Bacalar Chico. It is surrounded by a shallow reef flat that is approximately 450 - 700m in width (Gibson 1988) and has an almost entirely emergent peripheral reef
crest except in three places where it is broken by channels. On its windward side, there is a well-defined spur and groove system. The clear oceanic waters have permitted coral growth to depths of 100 m or more. A gradual fore reef extends from the seaward margins of the peripheral reef to the drop-off. The fore reef ranges in width from about 400 m to 1.5 km with the edge of the drop-off lying at a depth of 15-25 m (Gibson 1988).

The atoll is isolated from major human populations and, for the most part, human activity within its boundaries is mild. Small tourism ventures and fishing occur within and around this site (Gibson et al. 2004). However, satellite imagery after storm events shows evidence that terrestrial run-off can reach this isolated atoll (Andréfouët et al. 2002). The NTZ has been found to exhibit a higher abundance of fish species than the fished area (McClanahan et al. 2001; Gibson et al. 2004). However, management is unsuccessful to protect corals from disease outbreaks, like white band disease, which significantly impacted the reserve just prior to the 1998 hurricane and bleaching disturbances (Aronson et al. 2000).

METHODOLOGY

Research Hypothesis: There are significant differences in the proportional cover of different functional groups of benthos between no-take zones (NTZs) and fished reefs during a post-disturbance recovery period. Specifically, the NTZ should have more coral, less macroalgae, and more cover of crustose coralline algae, fine turfs, and bare space (CTB) than fished reefs.

Null Hypothesis: There are no significant differences in the cover of different
functional groups of benthos between NTZs and fished reefs.

To test this hypothesis, I investigated fore-reef structure at 15-18 m depth within the NTZ of each of the three reserves of study: Bacalar Chico, Hol Chan and Glover’s Reef Marine Reserves. Nearby fished areas of ecologic and geomorphologic similarity were surveyed as paired controlled sites: Mexico Rocks, Tackle Box, and Glover’s Southwest Caye, respectively. A study area of approximately 300 m² was established at each site, corresponding to the geographic coordinates reported by researchers McField et al. (2001) and Bood (2005) from identical sampling efforts in previous years. Baseline data on the condition of these fore-reef communities (i.e. coral, macroalgae, sponge and turf algae cover, and coral species diversity and richness) in the aforementioned depth range were used for comparison to my 2009 datasets (McField et al. 2001, Bood 2006).

Figure 3. A. Diver holding video camera housing 25cm above the benthos for consistent sampling frame. B. Field assistant laying 25m transect along fore reef spurs.

I replicated a video-based reef quantification methodology employed by researchers
McField (1997, 1998) and Bood (2005). The sampling scheme was a haphazard transect design on windward fore-reef spurs which is similar to that of several other monitoring programs (Aronson et al. 1994; Edmunds et al. 1998; Jaap and McField 2001; Miller et al. 2003). Seven replicate 25m transects were videotaped at each site. Transects were oriented along individual reef spurs (i.e. perpendicular to the shelf edge) and demarcated with fiberglass measuring tape. Digital video was taken beside each of the seven 25-m transect line with the camera held 25 cm above the substrate with a scaled reference bar, enabling a consistent recorded swath width of approximately 25 cm. As required by the Duke University dive safety program, my field assistant and I were certified as Scientific Divers by the American Association of Underwater Sciences at the time of sampling.

Once collected, video footage was processed and quantified by four variables: 1.) abundance of hard coral 2.) abundance of macroalgae 3.) abundance of crustose coralline algae, fine turf algae, and bare space and 4.) abundance of sponge. I used Apple software to capture 65 still and non-overlapping frames from each transect. I chose 50 frames at random from among the clearest and least obstructed images of the benthos (Figure 4). Point Count software was used to generate 10 random points on each image and to provide menu selection of coral species and other functional groups for each point (Kohler & Gill 2006).

Biological components beneath each point were identified to hard corals species or functional benthic groups (e.g. macroalgae, sponge, turf, crustose coralline algae). I converted the point-count data to percent cover data for hard corals, macroalgae, sponges and a category that combined crustose coralline algae, fine turfs and bare space (CTB). This later category referenced by Aronson and Precht (2000) is comprised of benthic
groups associated with high levels of herbivory. I sampled 300 points per transect, totaling 2,100 points per site. Each transect was treated as a replicate and quantified for mean cover for each of the four examined variables.

![Figure 4. Examples of still images (2009) captured from raw underwater footage.](image)

Univariate measures of these categories of cover were compared among NTZs and fished reefs using three-way mixed effect ANOVA model with restricted maximum likelihood (REML). Protection status (i.e., unfished and fished) and year (i.e., 1997, 1999, 2005, and 2009) were used as fixed factors, whereas geographic location (i.e., site) was treated as a random factor. The interaction between the two fixed factors was also tested. Site was used as the random factor because it was foreseeable that there would be a degree of natural variability among sites. Based on the results, Tukey’s test of pairwise comparisons (Honestly Significant Differences) clarified significant relationship between different levels of the two fixed effects. Prior to computation of the ANOVAs, proportional data for each combination of factors (coral, macroalgal, CTB and sponge cover) were tested for normality and homogeneity of variances assumptions of parametric statistics using Levene’s tests, and transformed when necessary.

**RESULTS**
HARD CORAL ABUNDANCE

The data for raw hard coral cover were not normally distributed and exhibited heterogeneity of variances. Square root transformation corrected heterogeneity of variance (Levene’s test; p=0.9034) but not normality. Since the square root transformed data passed the homogeneity of variances assumption, the use of Analysis of Variance techniques (ANOVA) was allowed. Levene’s test was used as guidance since ANOVA is robust to non-normal data and there was a balanced sampling design. A three-way mixed model ANOVA detected no significant effect of protection status (unfished vs. fished reefs in paired comparisons; F = 0.1780; p = 0.678). The model detected a significant effect of year (1997, 1999, 2005 and 2009; F = 08.86192; p = 0.0012). The interaction between protection status and year was not significant (F= 0.13; p = 0.9363).

The drop in coral cover from 1997 and 1999 was highly significant (p = 0.0056) in both fished and NTZ areas, as reveled by Tukey’s multiple comparisons test. This is consistent with findings from other literature. The differences from the 1997 baseline and more recent sampling are also significant across sites, suggesting no affect of protection on recovery (1997-2005, p = 0.0019; 1997–2000 p= 0.0057)

<table>
<thead>
<tr>
<th>Year</th>
<th>Protection Status</th>
<th>Mean</th>
<th>+/- SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997</td>
<td>Fished</td>
<td>27.5600</td>
<td>2.5200</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>29.5600</td>
<td>1.8467</td>
</tr>
<tr>
<td>1999</td>
<td>Fished</td>
<td>11.3367</td>
<td>1.2700</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>11.7467</td>
<td>1.2133</td>
</tr>
<tr>
<td>2005</td>
<td>Fished</td>
<td>11.6400</td>
<td>1.5533</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>8.7937</td>
<td>1.1100</td>
</tr>
<tr>
<td>2009</td>
<td>Fished</td>
<td>12.7038</td>
<td>1.5533</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>10.2958</td>
<td>2.5489</td>
</tr>
</tbody>
</table>

Table 3. Coral cover by treatment and time.
Figure 5. Percent coral cover. Error bars represent standard error (SE).

MACROALGAE ABUNDANCE

The data for raw macroalgae cover were not normally distributed and exhibited heterogeneity of variances. Square root transformation corrected heterogeneity of variance (Levene’s test; p = 0.4361) but not normality. Similar to trends in hard coral cover, a three-way mixed model ANOVA detected no significant effect of protection status (unfished vs. fished reefs in paired comparisons; F = 1.0807; p = 0.314) but a significant effect of year (1997, 1999, 2005 and 2009; F = 0.86192; p > 0.0001). The interaction between protection status and year was not significant (F = 0.25; p = 0.8580).

In fished and NTZ sites, macroalgae cover followed a different temporal trajectory than hard coral after 1997. First, Tukey’s multiple comparison test show that it did not experience a significant decline after the 1998 disturbance, suggesting tolerance to
hurricane effects. Secondly, Tukey’s tests reveal a significant increase in macroalgae between 2005 and 2009 across all sites.

<table>
<thead>
<tr>
<th>Year</th>
<th>Protection Status</th>
<th>Mean</th>
<th>+/- SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997</td>
<td>Fished</td>
<td>19.5500</td>
<td>1.3267</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>17.7900</td>
<td>1.7967</td>
</tr>
<tr>
<td>1999</td>
<td>Fished</td>
<td>9.4300</td>
<td>1.6000</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>16.6467</td>
<td>1.7900</td>
</tr>
<tr>
<td>2005</td>
<td>Fished</td>
<td>25.4000</td>
<td>2.3500</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>28.1414</td>
<td>3.2600</td>
</tr>
<tr>
<td>2009</td>
<td>Fished</td>
<td>53.0168</td>
<td>2.3500</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>58.5646</td>
<td>2.9051</td>
</tr>
</tbody>
</table>

Table 4. Percent macroalgae cover by treatment and time.

Figure 6. Percent macroalgae cover. Error bars represent standard error (SE).
ABUNDANCE OF CRUSTOSE CORALLINE, TURF ALGAE, & BARE SUBSTRATE

The data for raw CTB cover were not normally distributed and exhibited heterogeneity of variances. Square root transformation corrected heterogeneity of variance (Levene’s test; p=0.098) but not normality. Similar to the trends in the three previous resilience factors, a three-way mixed model ANOVA detected no significant effect of protection status (unfished vs. fished reefs in paired comparisons; F = 1.0807; p = 0.0) but a significant effect of year (1997, 1999, 2005 and 2009; F = 08.86192; p > 0.0001). There was no interaction between protection and year. The 1998 impacts brought about a significant increase in the cover of CTB by 1999 (p < 0.001), which correlates with the reduction in coral cover. A notable relationship revealed by the Tukey’s multiple comparisons test is the significant reduction in CTB across all sites from 2005 to 2009 (p< 0.003). This seems to correlate with the significant increase in macroalgalae also observed from 2005 and 2009.

<table>
<thead>
<tr>
<th>Year</th>
<th>Protection Status</th>
<th>Mean</th>
<th>+/- SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997</td>
<td>Fished</td>
<td>39.5633</td>
<td>3.5133</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>41.8033</td>
<td>1.8533</td>
</tr>
<tr>
<td>1999</td>
<td>Fished</td>
<td>69.8867</td>
<td>2.0267</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>63.9367</td>
<td>2.2267</td>
</tr>
<tr>
<td>2005</td>
<td>Fished</td>
<td>54.9047</td>
<td>2.5633</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>53.6400</td>
<td>3.7433</td>
</tr>
<tr>
<td>2009</td>
<td>Fished</td>
<td>16.4528</td>
<td>2.5633</td>
</tr>
<tr>
<td></td>
<td>NTZ</td>
<td>36.6667</td>
<td>3.3904</td>
</tr>
</tbody>
</table>

Table 5. Percent CTB cover by treatment and time.
**SPONGE ABUNDANCE**

Data for raw sponge cover were not normally distributed and exhibited heterogeneity of variances. Square root transformation corrected heterogeneity of variance (Levene’s test; p=0.052) but not normality. A three-way mixed model ANOVA (Table 8) detected no significant effects of protection status (unfished vs. fished reefs in paired comparisons; F = 0.125; p = 0.728), and year was only weakly significant (F = 1.102; p = 0.030). The protection status and year interaction were not significant (F = 0.559; p = 0.043). Sponge cover experienced a significant increase across sites in from 2005 to 2009 (p = 0.047). Starting from such low cover, it is difficult to interpret these results in relation to the other comparative increases observed between 2005 and 2009.

*Figure 7.* Percent CTB cover. Error bars represent standard error (SE).
<table>
<thead>
<tr>
<th>Year</th>
<th>Protection Status</th>
<th>Mean</th>
<th>+/- SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997</td>
<td>Fished</td>
<td>1.2667</td>
<td>0.3333</td>
</tr>
<tr>
<td></td>
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<td>0.5567</td>
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**Table 7.** Percent sponge cover by treatment and time.

![Mean Sponge Cover](image.png)

**Figure 8.** Percent sponge cover. Error bar represent standard error (SE).
DISCUSSION

My hypothesis was that the reefs in Belize’s NTZs should exhibit significantly higher coral cover, lower macroalgal cover, and higher CTB cover than fished reefs. Recent studies have identified these and other ecosystem characteristics as potential factors for reducing disturbance impacts (Knowlton & Jackson 2008; Halford & Caley 2009; Mumby 2009a). Contrary to expectation, I detected no significant difference in benthic cover between fished and protected reefs in Belize even a decade after a major disturbance event.

These results fail to reject my null hypothesis. This warrants a discussion of the current role of protection on the BBRC - a concept that can often be evasive and ambiguous in its implementation. If the rate of natural disturbances increases with climate change, managers must respond with forms of protection that are more than sufficient for Belize’s local needs. Before challenging literature-based concepts regarding the ecosystem-level benefits of NTZs (Almany et al. 2009; Halpern & Warner 2002; Selig & Bruno 2010), the limitations of the project must be clarified.

THE NEED FOR MORE DATA

I did not have access to herbivore population data in this project, though other members of the project are processing these data. Therefore, it is difficult to infer information about the functional role of herbivory occurring in the system. Other researchers have documented that herbivorous fish can regulate algal cover on reefs (Mumby 2006; Burkepile & Hay 2008), but they will only promote the availability of recruitment space if their populations are substantial (Mumby 2009b). The “protection”
variable associated with the NTZ sites in this methodology (NTZ = 1; fished = 0) describes the direct protection of reef fish populations from fishing activities. However, the response of herbivore populations as a dependent variable is unknown at this time. While it is tempting to conclude that the protection afforded to certain reefs failed to foster coral recovery, it is important to acknowledge that the “protection” variable relates only indirectly to coral cover. To draw more confident conclusions relating to NTZ performance, the intermediate ecological function in this trophic cascade (i.e., herbivory) needs to be quantified. This will be possible as other project data are processed by colleagues.

This project -- and the larger study it complements -- is further limited by a lack of information regarding compliance rates. Compliance to MPA zoning by user groups is assumed; this assumption is grounded in very limited evidence outside of self-reporting documents by Belize’s Fisheries Department (Gibson et al. 2004). With no significant effect of the “protection” variable, it is plausible that protection legislated within these NTZs is not being realized through compliance or enforcement in the reserves. The term “paper parks” most often describes this phenomenon. It can explain much of the disconnect between management goals and efficacy observed in protected areas worldwide (Bonham et al. 2008). Ignoring moratoriums on mangrove cutting has been extensively reported in Belize’s marine protected areas; other reports suggest various oversights in MPA management on the BBRC (Cho 2005; TNC 2008; WHC 2009). Therefore, the “paper park” phenomenon is a viable explanation for interpreting the failed role of NTZ in apparent coral resilience and recovery.
ADRESSING INADEQUACIES

The temporal regeneration of coral is much slower than that of macroalgal growth, supporting the idea that the 1998 disturbances in Belize were strong enough to act as a catalyst for the observed boom in macroalgal cover (Gardner et al. 2005; Knowlton & Jackson 2008; Mumby 2009b; Sandin et al. 2008). Caribbean coral reefs are widely thought to exhibit two alternate stable states, with one being dominated by coral and the other by macroalgae (Hughes 1994). The results of this project suggest that a post-disturbance shift to a macroalgal dominated state --on both protected and fished reefs of Belize -- has already occurred. The herbivore protection within Belize’s NTZs was not sufficient in compensating for the severity of coral loss and persistence of macroalgal growth. In other words, herbivory may have been enhanced by NTZ but herbivores may not have been able to keep up with macroalgal production.

Insufficient protection may be attributed to design factors, like geographic size, proximity to other stressors, and isolation from other NTZs (Keller et al. 2009). Mumby (2009) reveals that modest increases in grazing may not necessarily allow coral populations to recover, whereas large increases are more likely to exceed threshold levels of grazing intensity and set a trajectory of coral recovery. Therefore, small size – like that of the NTZ in Bacalar Chico Marine Reserve – may act as a bottleneck to sufficient grazing and subsequent increases in coral recruitment. In the case of Glover’s Reef Marine Reserve, the isolation of its fairly-large NTZ may limit the amount of coral recruits available from outside its boundaries (Crowder et al. 2000). This bottleneck would possibly be worse immediately after an acute loss of regional coral cover. Lastly, certain threshold levels of grazing cannot be passed if external stressors, like nitrification,
are accelerating macroalgal growth. In this way, coral cover may be persistently depleted even while increases in grazing reduce some macroalgal cover. If grazing can not keep up with nutrient inputs, recruitment will be limited. Mumby (2009) argues that once levels of coral recruitment become sufficient to overwhelm the population bottleneck, the coral-dominated state can begin to emerge from an algal-dominated state.

Despite the results of this project, Mumby’s argument rationalizes NTZs as possible tools for coral recovery, given that NTZs are designed with resilience in mind. Unfortunately, the current network of MPAs and NTZs in Belize were not designed for this purpose. Spatial relationships, sizes, and proximity to complementary ecosystems is not integrated into current NTZ placement (Cho 2005b). In other words, the NTZs were not designed as a network of reserves but were set up opportunistically. In addition to network design, addressing external chronic stressors may help in optimizing coral resilience. Given that many stressors in marine ecosystems are synergistic, understanding the variation and localization of cumulative stressor effects could help in improving NTZ design (Crain et al. 2008; Crowder et al. 2006). Localized surveys on *Monostrea faveolata* coral species on the BBRC show that coral resilience significantly improved with the decrease of external anthropogenic stressors (Carilli et al. 2009). This evidence of coral recovery on the BBRS – albeit small-scale – is promising for future management strategies.

THE 2005-2009 YEARS: REASONS TO THINK BEYOND NO-TAKE ZONES

Perhaps the most relevant finding of this project is the overwhelming significance of the time period 2005 to 2009. During the 10 year post-disturbance period (1999
though 2009), macroalgal cover, CTB, and sponge cover saw the greatest changes during the most recent four years. As recently revealed about coral reefs in other parts of the world, Belize’s reefs are changing at an increasing rate (Obura 2005; Hoegh-Guldberg et al. 2007; Knowlton & Jackson 2008; Veron et al. 2009). For example, sponge cover has increased three to four-fold from historical averages on three of the six study sites. While sponges are unlikely to dominate the ecosystem in the near future, these findings illustrate the uncertain and unanticipated responses by reef communities that are likely to result from new disturbances and stressors.

If NTZs have the capacity to foster reef resilience in Belize, they must be complemented by strategies that are not place-based. Macroalgae operate as a pervasive and strong interactor in the BBRC food web. While MPA networks (i.e., increased spatial protection) are considered a potentially effective management approach for promoting resilience in the system, enhancing the spatial network in Belize should be completed in conjunction with other strong fisheries regulations (i.e., affecting herbivory) and reductions of land-based nutrient inputs. In other words, reserves will best function when management of factors external to the reserve is also adequate. However, a more integrative management option may be overshadowed by the current push to increase spatial protection by HRI and other conservation groups. Given the lack of recovery response to NTZ protection, it seems justified that the Government of Belize concentrate on ameliorating known stressors to corals in a more integrative fashion, while assuring that reserves have sufficient enforcement and compliance to assure their function.
CONCLUSION

The level of protection currently afforded by Belize’s NTZs doesn’t appear to further coral ability to absorb shocks, resist phase-shifts, or recover by significant proportions. The disturbances of 1998 significantly reduced coral cover on both NTZ and fished reefs. As a result, macroalgal cover increased significantly, perhaps past a certain proportional threshold. I recommend that thresholds be delineated to determine if Belize’s reefs have entered into a stable state of macroalgal domination. As of 2009, no significant recovery of coral assemblages was observed in either NTZ or fished reefs.

These results are surprising, given the substantial praise and attention given to Belize’s creation of MPAs. As tools for reef management, NTZs are not currently sufficient for resilience-based results. Possible inadequacies include enforcement, network design, and management. Belize’s NTZs hold great potential to enhance resilience, but their purpose, design, and approach may need to be re-examined.

This project reveals valuable information on the ecological status of Belize’s reefs and suggests likely factors hampering reef resilience within and outside of NTZs. As the stressors facing the BBRC become more numerous and synergistic, Belize must adapt its management strategies to a changing environment and emergent human behaviors (i.e., marine debris). The integration of various resource management strategies can more efficiently tackle ecological and human needs. Conservation and government leaders are thus urged to look beyond purely spatial options in crafting tools for reef resilience.
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