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Coral Reef Populations in the Caribbean: Is There a Case for Better Protection against Climate Change?

Michael James C. Crabbe
Institute for Biomedical and Environmental Science and Technology, Faculty of Creative Arts, Technologies and Science, University of Bedfordshire, Luton, UK
Email: james.crabbe@beds.ac.uk

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ABSTRACT

Knowledge of factors that are important in coral reef growth help us to understand how reef ecosystems react following major environmental disturbances due to climate change and other anthropogenic effects. This study shows that despite a range of anthropogenic stressors, corals on the fringing reefs south of Kingston harbour, as well as corals on fringing reefs on the north coast of Jamaica near Discovery Bay can survive and grow. Skewness values for *Sidastrea siderea* and *Porites astreoides* were positive (0.85 - 1.64) for all sites, implying more small colonies than large colonies. Coral growth rates are part of a demographic approach to monitoring coral reef health in times of climate change, and linear extension rates (mm·yr^{-1}) of *Acropora palmata* branching corals at Dairy Bull, Rio Bueno, and Pear Tree Bottom on the north coast of Jamaica were c. 50 - 90 mm·year^{-1} from 2005-2012. The range of small-scale rugosities at the Port Royal cay sites studied was lower than that at the Discovery Bay sites; for example Rio Bueno was 1.05 ± 0.15 and Dairy Bull the most rugose at 2.3 ± 0.16. Diary Bull reef has for several years been the fringing reef with the most coral cover, with a benthic community similar to that of the 1970s. We discuss whether Jamaica can learn from methods used in other Caribbean countries to better protect its coral reefs against climate change. Establishing and maintaining fully-protected marine parks in Jamaica and elsewhere in the Caribbean is one tool to help the future of the fishing industry in developing countries. Developing MPAs as part of an overall climate change policy for a country may be the best way of integrating climate change into MPA planning, management, and evaluation.

Keywords: Demographics; Belize; Jamaica; MPAs; GDP; Hurricanes; Fishing; Bleaching; Climate Change; Global Warming

1. Introduction

Coral reefs throughout the world are under severe challenges from environmental factors including overfishing, destructive fishing practices, coral bleaching, ocean acidification, sea-level rise, algal blooms, agricultural run-off, coastal and resort development, marine pollution, increasing coral diseases, invasive species, ocean acidification, rising sea level, changing circulation patterns, increasing severity of storms, changing freshwater influxes and hurricane/cyclone damage (e.g. [1-4]). The fringing reefs around Jamaica constitute one of the best documented areas of reef decline in the Caribbean, where significant loss of corals and macroalgal domination has been due to hurricanes [5-7], overfishing [8,9], die-off of the long-spined sea urchin *Diadema antillarum* in 1983-1984 [10], and coral disease [11]. Nutrient enrichment does not appear to have been a causal factor in the development of these reef macroalgal communities [12].

Warming ocean (sea surface) temperatures due to climate change are considered to be an important cause of the degradation of the world's coral reefs. Marine protected areas (MPAs) have been proposed as one tool to increase coral reef ecosystem resistance and resilience (*i.e.* recovery) to the negative effects of climate change. However, few studies have evaluated their efficacy in achieving these goals.

While healthy reefs usually have high numbers of coral recruits and juvenile corals, degraded systems typically have limited numbers of young colonies [13,14]. To manage coral reefs it is important to have an understanding of coral population demography-structure and dynamics. Ideally, this involves the quantification of numbers of individual colonies of different size classes-the population structure-through time, in addition to quantifying coral growth rates, recruitment and survival.

Knowledge of coral population structure helps in understanding how reefs react following disturbance, and provides us with an early warning system for predicting future reef health.

Here we studied recent non-branching coral population structure at sites near Discovery Bay on the north coast of Jamaica, and at sites near Kingston Harbour, on the south coast of Jamaica.

Kingston harbour is a major trans-shipment post for the Caribbean, and in 2002 a major ship channel (East Channel) was constructed by Rackham's Cay, near Port Royal to accommodate vessels with draft of up to 14.5 metres and beams of 42 metres [15]. In addition, the harbour is highly polluted, mostly due to an excessive input of sewage [16-19]. This study shows that despite these and other environmental stressors, corals can survive on the fringing reefs south of the harbour.

MPAs provide *in-situ*-based management of marine ecosystems through various degrees and methods of protective actions. As impacts of climate change strengthen they may exacerbate effects of existing stressors and require new or modified management approaches; for example MPA networks may be an improvement over individual MPAs in addressing multiple threats to the marine environment. We discuss whether Jamaica can learn from methods used in other Caribbean countries to better protect its coral reefs in times of climate change.

2. Materials and Methods

2.1. Reef Sites

GPS coordinates were determined using a hand-held receiver (Garmin Ltd.).

In Discovery Bay, Jamaica, surface areas of non-branching corals between 5 - 9 m depth, were measured using SCUBA at four sites [Rio Bueno (18˚28.805'N; 77˚21.625' W), M1 (18˚28.337'N; 77˚24.525'W), Dancing Ladies (18˚28.369'N; 77˚24.802'W), Dairy Bull (18˚28.083'N; 77˚23.302'W) and Pear Tree Bottom (18˚27.829'N; 77˚21.403'W)] along the fringing reefs surrounding Discovery Bay, Jamaica. Overall, surface areas of 209 non-branching corals were measured in 2012 (**Table 1(a)**).

This work was conducted at Discovery Bay during August 8-August 10 in 2012.

For sites near Port Royal, Jamaica, surface areas of non-branching corals were measured using SCUBA at seven sites south of Port Royal. Six of these were fringing reefs around the cays South of Port Royal: SE Barrier Cay (17˚53.714'N, 76˚48.226'W); Lime Cay (17˚54.948'N, 76˚49.134'W); Gun Cay (17˚55.901'N, 76˚50.141'W); Drunkenman's Cay (17˚54.128'N, 76˚50.736'W), the face of the ship channel made in 2002 along the reef of Rackham's Cay (17˚55.571'N, 76˚50.307'W) and Maiden Cay (17˚54.506'N, 76˚48.728'W); a seventh site was the wreck of the ship *Edina*, south of the barrier reef (17˚49.525'N, 76˚50.723'W). Depth of samples at six of the sites was between 5 - 12 m, at the *Edina* wreck it was between 22 - 28 m.

Overall, surface areas of 347 non-branching corals were measured in 2010 (**Table 1(b)**) and of 451 non-branching corals in 2012 (**Table 1(c)**).

This work was conducted in April 14-16 in 2010 and July 30-August 2 in 2012. SE Barrier Cay was only studied in 2010, owing to time limitations caused by tropical storm Ernesto in 2012.

2.2. Sampling

Details of data sampling have been described for North Jamaica and South Jamaica [6,20]. In summary, corals 2m either side of transect lines were photographed for archive information, and surface areas measured with flexible tape as described previously using SCUBA. For non-branching corals, this was done by measuring the widest diameter of the coral and the diameter at 90˚ to that.

In all cases except that of the Edina wreck near Port Royal, which was 22 - 28 m, depth of samples was between 5 - 12 m, to minimise variation in growth rates due to depth [21]. To increase accuracy, surface areas rather than diameters of live non-branching corals were measured [6]. Sampling was over as wide a range of sizes as possible. Colonies that were close together (<50 mm) or touching were avoided to minimise age discontinuities through fission and altered growth rates [22,23]. In this study we ignored *Montastrea annularis* colonies, because their surface area does not reflect their age [22], and because hurricanes can increase their asexual reproduction through physical damage [23].

Computer digital image analysis for coral linear extension rates was undertaken using the UTHSCSA (University of Texas Health Science Center, San Antonio, Texas, USA) Image Tool software (see [6]). One-factor ANOVA was used; ±error values represent standard errors of the data.

Skewness [24] (sk) was used to estimate the distribution of small and large colonies in the coral populations. Negative skewness implies more large colonies than small colonies, while conversely positive skewness implies more small colonies than large colonies.

2.3. Rugosity

Rugosity (R) was determined according to the formula:

$$R = Sr/Sg$$

where Sr = real surface distance between two points, and Sg = straight line geometric distance between two points. This was calculated over a 20 m distance, performed in

Table 1. Coral species and numbers studied at sites around Port Royal in 2010 and 2012, and around Discovery Bay in 2012. (a) Corals at sites around Discovery Bay in 2012; (b) Corals at sites around Port Royal in 2010; (c) Corals at sites around Port Royal in 2012.

(a)

2012					
Species	Dairy Bull	M1	Pear Tree	Rio Bueno	Total
ss	37	12	2	7	58
pa	19	10	12	9	50
mean mean	6	25	4	1	36
dip strig	2			8	10
ag ag	5		2	3	10
mont cav	6			2	8
dip laby	4	4	3		11
eusmilia	2		1		3
manicina					0
muss ang					0
mycetophyllia	10				10
col natans	4	5	1	3	13
Total	95	56	25	33	209

(b)

2010								
Species	Rackhams	Edina wreck	Drunkenmans	SE barrier	Maiden	Lime	Gun	Total
ss	78	20	12	21	5	23	10	169
pa	10	6	28	15		2	7	68
mean mean	9	3	6	4	3	4	1	30
dip streg	1	16	8	12		4		41
ag ag		1						1
mont cav		8	4	2		2	10	26
dip laby			2	2	2	3	1	10
eusmilia						1		1
col natans							1	1
Total	98	54	60	56	10	39	30	347

(c)

2012							
Species	Rackhams	Edina wreck	Drunkenmans	Maiden	Lime	Gun	Total
ss	35	17	22	42	43	33	192
pa	6	9	10	27	20	9	81
mean mean		1	8	1	7	5	22
dip strig		5	8	1	4	4	22
ag ag		8	1	8	7	1	25
mont cav	1	8	11	10	7	8	45
dip laby	1	2	1	3			7
eusmilia				4	1		5
manicina						3	3
muss ang			1				1
mycetophyllia	1	2	7	14	12		36
col natans		1		1	6	4	12
Total	44	53	69	111	107	67	451

Legend. Coral species: ss, *Sidastrea siderea* (Ellis, 1786); pa, *Porites astreoides* (Lamarck, 1816); meanmean, *Meandrina meandrites* (Linnaeus, 1758); Dip strig, *Diploria strigosa* (Dana, 1848); agag, *Agaricia agaricites* (Linnaeus, 1758); mont cav, *Montastrea cavernosa* (Linnaeus, 1758); dip laby, *Diploria labyrinthiformis* (Linnaeus, 1758); eusmilia, *Eusmilia fastigiata* (Pallas, 1766); col natans, *Colpophyllia natans* (Houttuyn, 1772); manicina, *Manicina areolata* (Linnaeus, 1758); muss ang, *Mussa angulosa* (Pallas, 1766); mycetophyllia, *Mycetophyllia lamarckiana* (Milne Edwards 1848). GPS coordinates of all sites are given in the text. Gaps in the tables indicate 0.

triplicate, at each site, using photographic image analysis verified by the chain method, as described previously for Discovery Bay sites [25].

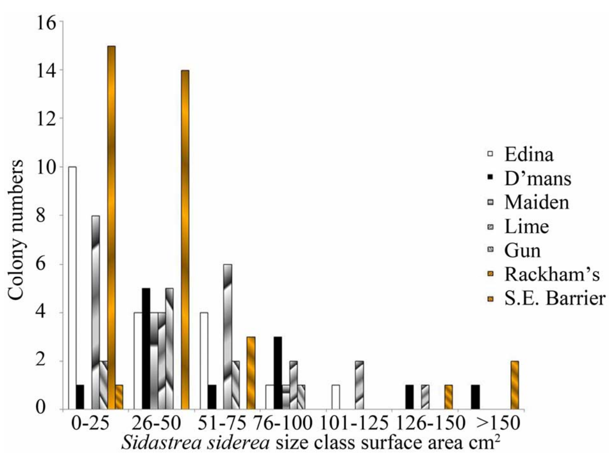
3. Results

The reefs fringing the cays around Port Royal in Jamaica did not exhibit extensive three-dimensional complexity; this is exemplified by their rugosities: Rackhams cay, 1.42 ± 0.15; Edina wreck, 1.37 ± 0.17; Drunkenman's cay 1.41 ± 0.16; SE Barrier cay, 1.39 ± 0.15; Maiden cay, 1.1 ± 0.1; Lime cay, 1.16 ± 0.12; and Gun cay, 1.17 ± 0.2.

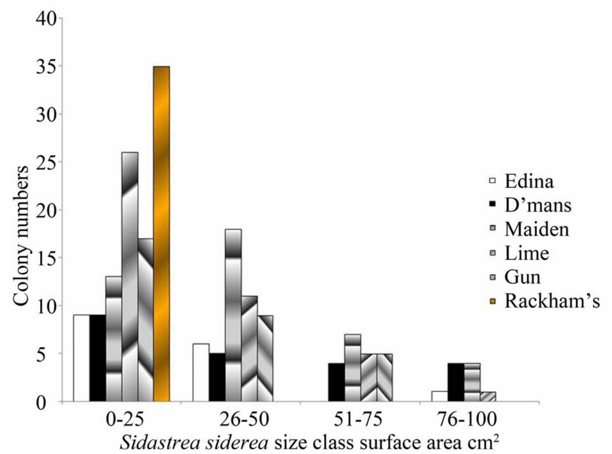
Despite their low three-dimensional complexity, there was a high proportion of small size classes of non-branching corals that included new recruits and juveniles on these reefs. This is illustrated for both 2010 and 2012 in **Figures 1(a)** and **(b)** for *Sidastrea siderea* and in **Figures 1(c)** and **(d)** for *Porites astreoides*. These patterns are typical of other species of corals, where the numbers of corals are smaller. Skewness values for both species were positive (for example 1.36 for *Sidastrea siderea* at the Edina site in 2010, and 0.85 for *Porites astreoides* at Drunkenman's Cay in 2012), implying more small colonies than large colonies.

There was also a high proportion of small size classes of non-branching corals on the fringing reefs around Discovery Bay on the Jamaican north coast in 2012. This is illustrated for *Sidastrea siderea* and *Porites astreoides* in **Figures 2(a)** and **(b)**. Once again, this pattern was typical of other coral species. Skewness values for both species were positive, with Dairy Bull having the highest values (for example 1.64 for *Sidastrea siderea* and 1.12 for *Porites astreoides* in 2012 at Dairy Bull), implying more small colonies than large colonies.

Coral growth rates are part of a demographic approach to monitoring coral reef health in times of climate change, and **Figure 3** presents linear extension rates ($mm \cdot yr^{-1}$) of *Acropora palmata* branching corals (n = 4) at Dairy Bull,



(a)



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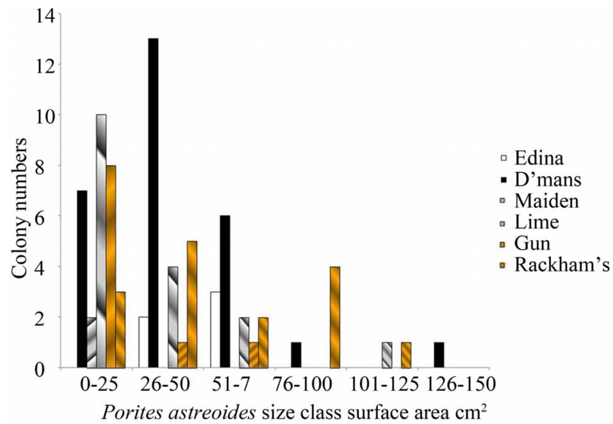
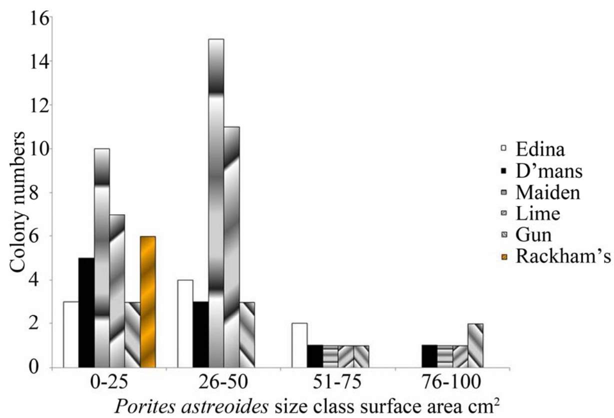


(c)



(d)

Figure 1. Size classes of non-branching corals on the fringing reefs on the Port Royal Cays. (a) ***Sidastrea siderea*** **in 2010; (b)** ***Sidastrea siderea*** **in 2012; (c)** ***Porites astreoides*** **in 2010; (d)** ***Porites astreoides*** **in 2012.**

Rio Bueno, and Pear Tree Bottom on the north coast of Jamaica from 2005-2008, and from 2009-2012. Growth rates are similar to those reported previously [21]. There were no significant differences between the sites, or across the time periods of the study. Where growth rates

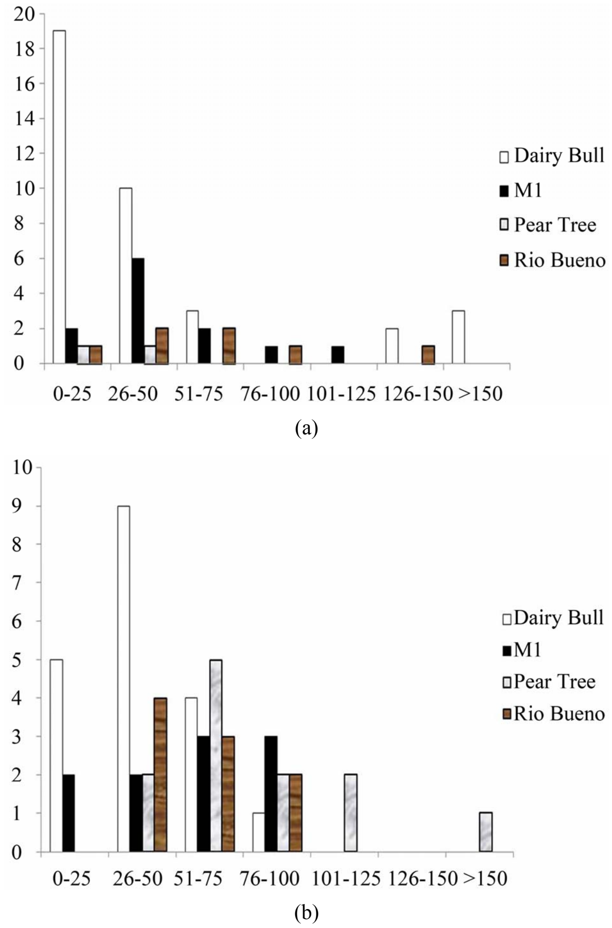


Figure 2. Size classes of non-branching corals on the fringing reefs near Discovery Bay, Jamaica. (a) ***Sidastrea siderea*** **in 2012; (b)** ***Porites astreoides*** **in 2012.**

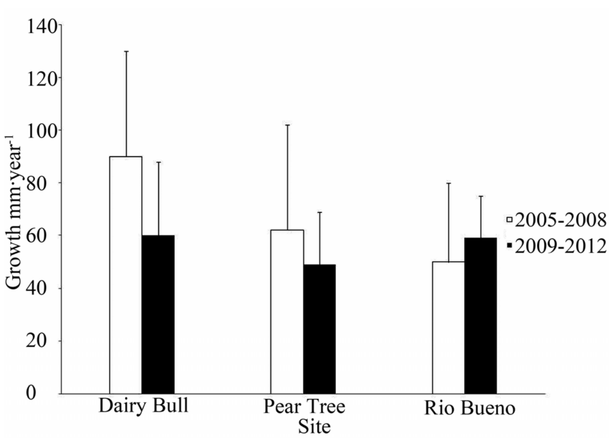


Figure 3. Linear extension rates of ***Acropora palmata*** **at sites near Discovery Bay, Jamaica, from 2005-2008 and 2009-2012. Error bars represent standard errors of the data (n = 4).**

were higher, they tended to be higher at Dairy Bull reef. With the increase of *Diadema antillarum* at Rio Bueno in recent years, clearing the macroalgae, many healthy *A. palmata* colonies have appeared at the Rio Bueno site from 2006.

4. Discussion

Mesoscale rugosity (larger scale) has been found important for predicting intra-habitat variation in coral reef fish assemblages, and explicitly in predicting the impacts of coral mortality on ecosystem process and services. This particularly relates to large, tall (>50 cm) corals [26]. Loss of architectural complexity in Caribbean reefs due to climate change appears to be linked to physical impacts, such as hurricanes and bio-erosion [3].

The range of small-scale rugosities at the Port Royal cay sites studied was lower than that at the Discovery Bay sites [25]; for example Rio Bueno was 1.05 ± 0.15 and Dairy Bull the most rugose at 2.3 ± 0.16. Diary Bull reef has for several years been the fringing reef with the most coral cover, with a benthic community similar to that of the 1970s [21], and it was the subject of the study which suggested a rapid phase-shift reversal [27].

In addition to climate change, Jamaican reefs are subject to a number of acute and chronic stressors, the last including overfishing and continuing coastal development, including the much-publicised development on land adjacent to Pear Tree Bottom reef and the resurfaceing of the North Jamaican coastal highway [28]. Faster rates of macroalgal growth, higher rates of algal recruitment iron enrichment from Aeolian dust, lack of acroporid corals, lower herbivore biomass and missing groups of herbivores all predispose the Caribbean to low resilience, relative to the Indo-Pacific region [29].

The Kingston harbour area has been impacted for many decades [16,19] and there have been efforts at mitigation, particularly in association with the ship channel [30]. In addition, some coral communities may adapt to chronic stressors, for example sedimentation, over long periods of time [31].

After the 2005 bleaching event there was a major loss of live coral cover [32,33], and it is encouraging that the population size studies show that there are numbers of both small and large colonies at both Discovery Bay and Port Royal sites. Also, the linear extension rates of *A. palmata* branching corals at Dairy Bull, Rio Bueno, and Pear Tree Bottom on the north coast of Jamaica were maintained from 2005-2012.

The influence of *M. annularis* colonies on the reef, acting as structural refugia [27], with maintenance of the biological legacies, may have facilitated this recovery. In addition, there have been no major hurricanes or bleaching events due to climate change since 2005 until this study which have impacted Jamaican fringing reefs.

Unlike other areas in the Caribbean, Jamaica has few Marine Protected Areas (MPAs) or Marine Reserves.

MPAs have been suggested as means for enhancing local resilience and population growth of marine species [e.g. 34,35]. However, some studies have highlighted continued climatic and other impacts in regions of MPAs [36-38].

Belize has the highest annual capture production-the annual volume of aquatic species caught by country for all commercial, industrial, recreational and subsistence purposes, in 2010, for countries in the Caribbean for which data is available [39]. The Belize continental shelf also includes the MesoAmerican Barrier reef; this World Heritage Site has levered the adoption of Marine Protected Areas (MPAs) for Belize. In 2010, Belize had 11.86% of its territorial waters as MPAs. This compares with Jamaica, 4.2%; Trinidad and Tobago, 2.8%; Barbados, 0.1%; St. Kitts and Nevis, 0.5%; St. Lucia, 0.1%; St. Vincent and the Grenadines, 0.6%. **Figure 4** shows the relationship between percentage of a country's territorial waters as MPAs with volume of catch ($r^2 = 0.88$); there is a similar correlation ($r^2 = 0.66$) with fisheries as a percentage of GDP [40] for the country concerned. Catch is a function of effort, and capture production may reflect this in addition to the influence of MPAs.

Recognition of the importance of fisheries in a country's GDP may be a factor in empowering conservation policy as well as local stakeholder action to conserve coral reefs. Having a large percentage of its territorial waters as MPAs, and in a coordinated network of MPAs, may reflect in the value of a country's fishing industry.

The 2012 IUCN report [41] on Caribbean coral reefs points to the decline of Caribbean reefs from c. 50% cover in the 1970 to just 8% today. They call for strictly enforced local action to improve the health of corals, including limiting fishing through catch quotas, an extension of MPAs, a halt to nutrient run-off from the land, and a reduction on the global resilience on fossil fuels.

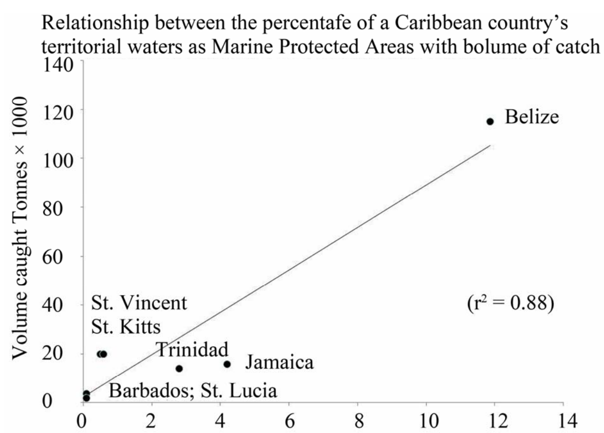


Figure 4. Relationship between the percentage of a country's territorial waters as Marine Protected Areas (MPAs) with volume of catch ($r^2 = 0.88$).

Through the IUCN-coordinated Global Coral Reef Monitoring Network (e.g. [42]) there are moves to strengthen the data available at a worldwide level.

Protection using *fully protected* MPAs is one of the few management tools that governments and local communities can use to combat large scale environmental impacts. In Bajja California, such an MPA provided protection to marine populations both within and outside the protected area [43]. The degree of protection is critical [44]. At Glover's Reef Marine Reserve, a no-take policy has not resulted in increases in grazing fish abundance, although commercially important fish, such as black grouper (*Mycteroperca bonaci*) and lane snapper (*Lutjanus synagris*) and invertebrates have increased [45]. This could be for many reasons, not least if they are poorly managed and maintained over long periods, particularly when there are stressful environmental events such as hurricanes of high sea surface temperatures (SSTs).

In an environment where carbon dioxide concentrations predicted to occur at the end of this century would significantly reduce coral settlement and crustose coralline algae cover, and would so reduce successful coral recruitment and larval settlement [46,47], establishing and maintaining fully-protected marine parks in Jamaica and elsewhere in the Caribbean is one tool to help the future of the fishing industry in developing countries.

Where protection in MPAs was not found to reduce the effect of warm temperature anomalies on coral cover declines [48], then shortcomings in MPA design, including size and placement, may have contributed to the lack of an MPA effect. It may be that the benefits from single MPAs may not be great enough to offset the magnitude of losses from acute thermal stress events. Although MPAs are important conservation tools, their limitations in mitigating coral loss from acute thermal stress events suggest that they need to be complemented with policies aimed at reducing the activities responsible for climate change. One way forward is to have networks of MPAs [49], and they could be more effective in conjunction with other management strategies, such as fisheries regulations and reductions of nutrients and other forms of land-based pollution. Developing MPAs as part of an overall climate change policy for a country [50] may be the best way of integrating climate change into MPA planning, management, and evaluation.

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