

## ORIGINAL ARTICLE

# Ecological assessment to detect imminent change, Admiral Cockburn Land and Sea National Park, Turks and Caicos Islands

Angela Dikou, Colin Ackerman, Carly Banks, Alex Dempsey, Michael Fox, Meagan Gins, Patricia Hester, Adam Parnes, Stephanie Roach, Jessica Rohde, Cliff Spital, Mackai Tapleshay & Luke Thomas

The School for Field Studies, Center for Marine Resource Studies, South Caicos, Turks and Caicos Islands, British West Indies

## Keywords

Caribbean; coral reefs; environmental impact assessment; island development; power analysis.

## Correspondence

Angela Dikou, Department of Ichthyology and Aquatic Sciences, University of Thessaly, Volos 38446, Greece.  
E-mail: angeladikou@hotmail.com

Accepted: 23 February 2009

doi:10.1111/j.1439-0485.2009.00291.x

## Abstract

Coral reefs of the Turks and Caicos Islands (TCIs) constitute some of the few pristine coral reef systems in the world and play a crucial role in the islands' economy because they support rich fisheries catches and tourism development. Ambitious development plans involving increase in fishing and tourism pressures are about to bring changes in coastal zone resources of the TCIs associated with increased sediments and nutrients and reduced predation by herbivorous fish on coral reefs. Understanding change is critical when attempting to protect the resources that these coral reefs support and to adopt proper management strategies. Yet, an environmental assessment program to detect imminent human-induced changes on the surrounding reefs of the TCIs is lacking. Thus, (i) we obtained baseline data on benthic composition and coral community structure at seven reef sites of representative reefs of the TCIs within the Admiral Cockburn Land and Sea National Park (ACLSNP) of South Caicos Island and (ii) performed *a priori* statistical power analysis to calculate replication requirements for safely and confidently detecting small ( $\delta = 0.1$ ), medium ( $\delta = 0.3$ ), and large ( $\delta = 0.5$ ) effect sizes for a number of relevant to anticipated changes, univariate, benthic indices and for power  $\beta = 0.95$ . The platforms of the margin reefs studied (9–12 m depth) appeared rather variable regarding benthic composition but quite homogeneous regarding hard coral community structure. Mean percent cover of algal functional groups was  $0.1 \pm 0.3$  (mean  $\pm$  sd) percent for coralline algae and *Halimeda*,  $0.1 \pm 0.6$  (mean  $\pm$  sd) percent for macroalgae,  $21.7 \pm 33$  (mean  $\pm$  sd) percent for turf algae and  $4.8 \pm 4.0$  (mean  $\pm$  sd) percent for hard coral cover. The dominant benthic component, however, was carbonate substrate (mean  $\pm$  sd =  $30.4 \pm 34.3$ ), thus indicating an accreting reef framework. Mean hard coral density, colony size and recruit density were  $5.5 \pm 1.8$  (mean  $\pm$  sd) corals per 20-m line transect,  $13.0 \pm 2.3$  (mean  $\pm$  sd) cm maximum colony diameter, and  $1.3 \pm 1.4$  (mean  $\pm$  sd) recruits per square foot, respectively. Due to high natural variance, hard coral colony size and density were practically the most sensitive indices in detecting even small size changes on benthos. Also, the geometric mean of log-transformed colony size-frequency distributions of the most abundant hard coral taxa, *i.e.* *Montastrea annularis*, *Agaricia* spp., *Siderastrea* spp. and *Porites asteroides* were practically sensitive for the same purpose. We hope that the study will optimize the spatial component of a necessary environmental impact assessment program on coral reefs of the TCIs once the natural spatial variability of the system has been assessed and sensitive, benthic, univariate indices have been identified for representative reference coral reef sites of the TCIs.

## Problem

Coral reefs play a crucial role in the Turks and Caicos Islands (TCIs) economy because they support fisheries and tourism development (Anonymous 2005). In addition, they constitute some of the few pristine coral reef systems in the world within a region hard hit by (Buddemeier *et al.* 2003) and particularly sensitive to (Buddemeier & Fautin 2002; Hughes *et al.* 2003) anthropogenic effects leading to either 'top-down' (reduction in herbivory) or 'bottom-up' (increase in nutrients) induced 'phase shifts' from coral to algal reefs (Littler & Littler 1984). Furthermore, change in coastal zone resources of the TCIs is imminent due to ambitious development plans involving increase in fishing and tourism pressures (TCI Government 2004; Anonymous 2007).

Expected alterations in the ambient waters of coral reefs of the TCIs due to programmed development involve elevated levels of sediments and nutrients and decreased herbivory. Appropriate indicators of expected alterations should involve relevant structural aspects of the reef framework-building hard coral communities (Dikou & van Woosik 2006a) and functional aspects of the benthos (Littler *et al.* 2006). Scleractinian, or hard corals are sensitive to alterations in nutrient and sediment status (Pastorok & Bilyard 1985; Rogers 1990; Fabricius 2005) and in herbivore abundance (McManus *et al.* 2000; Mumby *et al.* 2006) of their ambient waters. Benthic composition, on the other hand, may be a more informative indicator of ecosystem health as it incorporates more organisms than studies that focus on hard coral species only (Brown & Howard 1985). Benthic composition determines the maximum level of coral recruitment (Hughes *et al.* 2000), especially for self-seeded reefs (Black *et al.* 1991; Roberts 1997), as well as the substrate available for hard corals in the competition for bare space with the faster growing algae (Bak & Engel 1979; Sammarco 1980; Miller & Hay 1998; McCook *et al.* 2001; Birrell *et al.* 2004), and therefore the capacity for growth and recovery after perturbation (Connell 1997; Miller *et al.* 2000; Gardner *et al.* 2003). The importance of analyzing the composition of benthos on coral reef systems in relation to past, present and future anthropogenic effects has been noted (Bak & Luckhurst 1980; Hughes *et al.* 1987). Furthermore, there have already been documented coral to algal whole reef 'phase shifts' due to anthropogenic effects such as increase in sediment load due to land clearing and subsequent runoff, dredging, and coastal construction (Sanders & Baron-Szabo 2005; Dikou & van Woosik 2006b); in nutrients load due to sewage discharge (Maragos *et al.* 1985); and trophic cascades associated with

overfishing of herbivorous fish and quincidental mass mortality of the grazer sea urchin *Diadema antillarum* (Jackson *et al.* 2001).

Understanding and detecting imminent change on coral reefs of the TCIs is critical when attempting to protect the resources that these ecosystems support and to adopt proper management strategies. *A priori* statistical power analysis is a useful tool for study design as it allows the *ad-hoc* selection of the most sensitive indicators and the calculation of the sample size needed to detect a biologically/ecologically/environmentally meaningful magnitude of change in context (Fairweather 1991; Morrisey 1993). Its application in environmental impact assessments, however, has been rather limited so far (e.g. Ferraro & Cole 1990), supporting a false sense of security against public and environmentally costly Type II errors (Peterman 1990a,b).

Yet, an environmental assessment program to detect imminent human-induced changes on the surrounding reefs of the TCIs is lacking. Such a program requires a sampling design that would separate impacts from natural (spatial and temporal) variability through series of sampling times 'Before' and 'After' the disturbance, at the 'Impacted' site and at multiple 'Control' (reference) sites (BACI designs, Green 1979; Underwood 1992). This design should have adequate statistical power (Fairweather 1991; Benedetti-Cecchi 2001) to detect effects that are real, biologically/ecologically/environmentally important, and relevant to future anthropogenic interventions associated with development of the TCIs. The impact is defined as a change from 'Before' to 'After' the disturbance (the temporal component of the program) in the difference between the 'Impacted' and the average of the 'Control' sites (the spatial component of the program) and is detected as the interaction between the factors space (with two levels: 'Impacted' and 'Control' conditions) and time (with two levels: 'Before' and 'After' conditions). This study aims to optimize the spatial component of an environmental impact assessment program for coral reefs of the TCIs by providing power estimates of the difference between sites in a number of relevant benthic, univariate, community descriptors, which would correspond to the main effect of 'Impacted' *versus* 'Control' condition in a real environmental impact assessment study. In particular, it provides: (i) 'before' impact baseline data on benthic composition and coral community structure that allow the assessment of the natural spatial variability of the system and (ii) relevant and sensitive benthic indicators based on replication requirements for confidently detecting changes of small, medium and large size, using assessments at reef sites of representative reefs of the TCIs within the Admiral Cockburn Land and Sea National Park (ACLSNP) of South Caicos Island. The procedure is

readily applicable to the assessment of anthropogenic impacts on coral reefs in general.

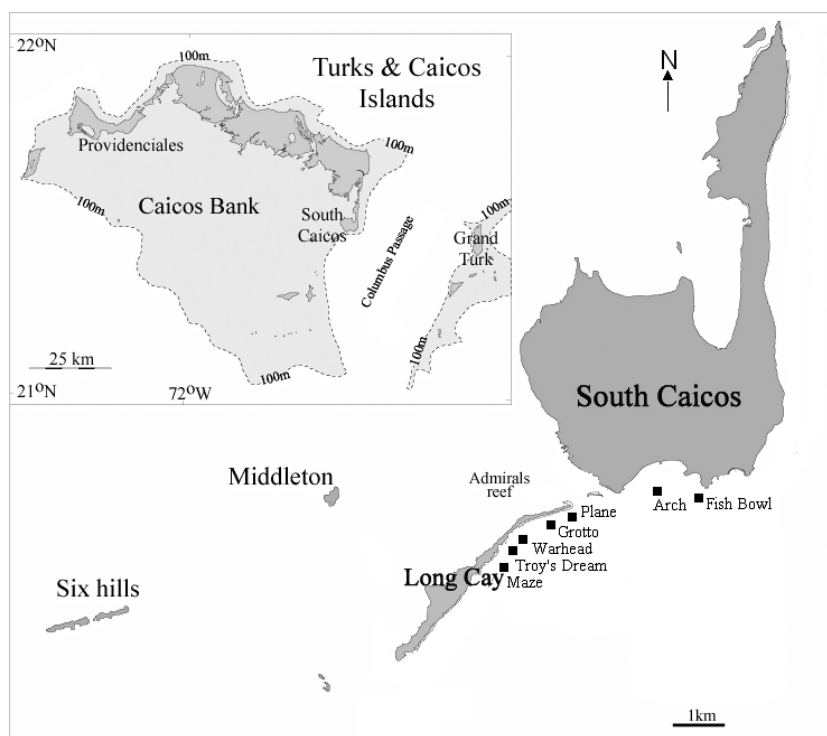
## Material and Methods

The TCIs are located at 21°53'N, 71°47'W in the Caribbean Sea. They consist of eight islands and roughly 40 cays situated among two banks; the Caicos and Turks Banks (Fig. 1). These islands are surrounded by over 300 km of coral reefs and, due to the generally low human population, they are considered to be in a literally pristine condition. The island of South Caicos lies on the southeastern end of the Caicos Island chain along the western edge of the Columbus or Turks Island Passage (Fig. 1). All reef sites surveyed were situated along reef flats on the outer reefs of the Caicos Bank; five reef sites were located on the southeast side of Long Cay (Maze, Troy's Dream, Warhead, Plane, Grotto) and another two were located south of East Bay (Fishbowl, Arch) (Fig. 1).

Surveys in November and December 2006 and in April and May 2007 with SCUBA provided three types of data: benthic cover, hard coral colony size, and hard coral recruits. Cover by main benthic components included live hard coral, filter feeders (sponge, soft coral and other alcyonaria, zoanthids), macroinvertebrates (molluscs, echinoids), turf algae (including encrusting fleshy algae), coralline algae (and *Halimeda*), calcareous substrate, macroalgae, dead hard coral, coral rubble, rock, and sand. For

benthic cover and hard coral colony size data we used time-efficient replicate 20-m-long line transects (Loya 1972). Transects were laid haphazardly at each reef site, parallel to the coast (Bouchon 1981) to ensure homogeneity of ecological conditions along each transect, and at depths between 9 and 12 m. At each site, 9–14 replicate transects were used depending on time and weather constraints. Data on hard coral recruits were obtained at regular intervals along each line transect using three replicate 0.09 square meter quadrats and a magnifying lens; coral recruits were defined to be <50 mm in diameter for large-size hard coral species and <20 mm for small-size species (Miller *et al.* 2000). All authors participated in field assessments after training by the same person (A. Dikou).

Univariate indices were extracted from the aforementioned data to describe and compare the community structure of the seven coral reef sites: percent cover of main benthic components, hard coral colony cover (percent), density (number of corals per 20-m line transect), Pielou's evenness (based on species cover,  $J'_C$ ) and size, (maximum diameter in centimetres) and hard coral recruit density (number of recruits per 0.09 square meter). To test for significant differences in percent live coral cover, hard coral colony density, Pielou's evenness, size, and recruit density among the seven reef sites, one-way analysis of variance (ANOVA) was employed. Data on percent live coral cover and on coral colony size



**Fig. 1.** Map of South Caicos showing the location of the seven study sites.

Maze: 21°28'160"N, 71°33'031"W;  
 Troy's Dream: 21°28'853"N, 71°32'157"W;  
 Plane: 21°28'831"N, 71°31'020"W;  
 Grotto: 21°28'876"N, 71°32'031"W,  
 Warhead: 21°28'815"N, 71°32'236"W;  
 Arch: 21°28'961"N, 71°32'031"W;  
 Fishbowl: 21°29'101"N, 71°30'505"W.

needed a double square root and a logarithmic transformation, respectively, to satisfy the assumptions of normality and homogeneity of variance of ANOVA. Data on hard coral evenness and recruit density could not satisfy both assumptions and therefore its non-parametric equivalent, the Kruskal–Wallis test, was employed.

Relationships among benthic substratum components were explored using Spearman rank correlations ( $r_s$ ) on means of the seven reef sites and ( $n = 7$ ) and on the whole dataset ( $n = 81$ ) to detect even small-size relationships. Ordination of reef sites in two dimensions reflecting their similarity/dissimilarity in their benthic composition was accomplished by employing multidimensional scaling (MDS) on an underlying triangular similarity matrix based on Bray–Curtis similarity of raw data (percent cover by different benthic components). Significant differences in mean percent cover of benthic components among reef sites were tested with a permutation procedure, the one-way analysis of similarity (ANOSIM). Clustering (based on group-average) was used to group sites into relatively homogeneous groups. MDS, ANOSIM and clustering were performed using PRIMER 5.0 (Clarke & Warwick 2001). All other statistical tests were carried out using STATISTICA 5.0.

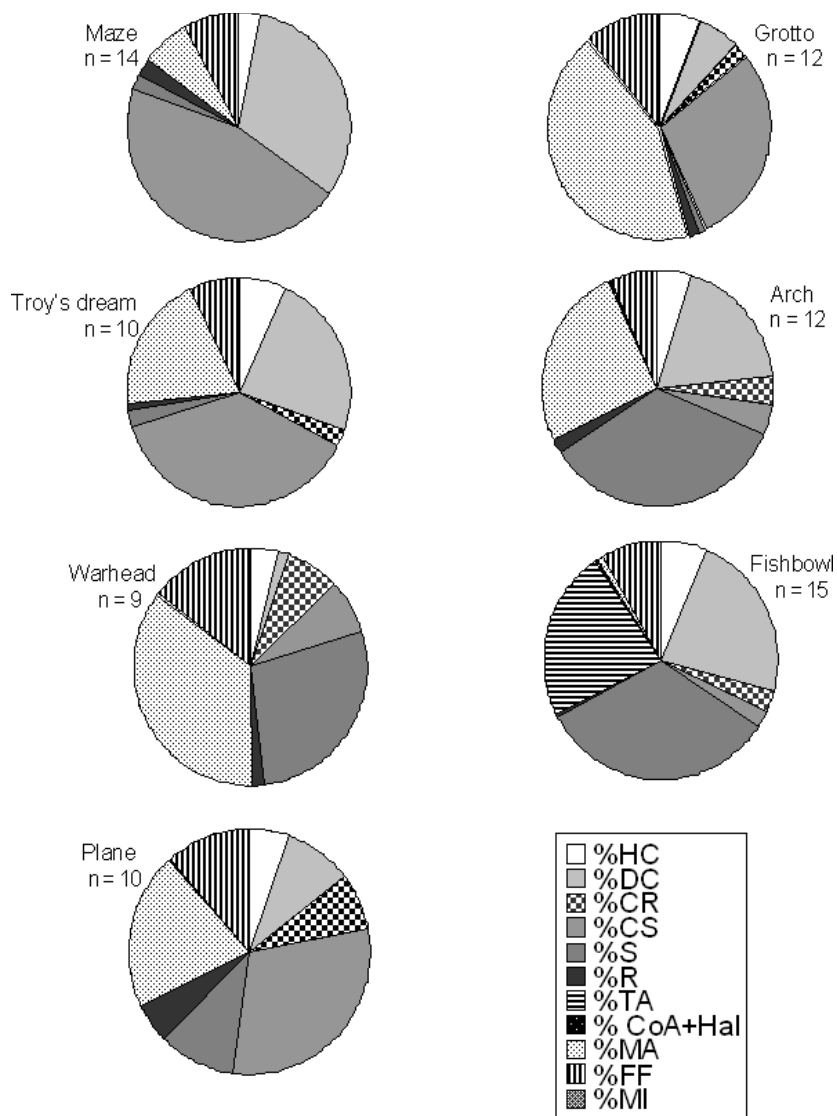
Also, log-transformed colony size-frequency distributions (based on maximum colony diameter) of *Agaricia* spp., *Siderastrea* spp., *Favia fragum*, *Porites asteroides* and *Montastrea annularis* were constructed. These taxa abound in shallow waters of the studied reefs and of Caribbean reefs, in general. It is speculated that the majority of *Agaricia* spp. colonies were *Agaricia agaricites* with a small contribution of *Agaricia humilis*. All *Siderastrea* spp. colonies were either *Siderastrea radicans* or *Siderastrea siderea* but *in situ* classification was difficult to perform. Logarithmic transformation of coral-size data produces a better view of population structure in scleractinian corals (Bak & Meesters 1998) and provides species- and site-specific responses to alterations in environmental conditions (Meesters *et al.* 2001). Parameters characterizing the aforementioned distributions were the geometric mean size, skewness (describes the asymmetry around the mean), kurtosis (describes the peakedness near the central mode), the 95th percentile (an indication of the maximum colony size), standard deviation, and the probability that the sample is from a normal distribution (Sokal & Rohlf 1995).

Power represents the probability that a test will result in the correct rejection of the null hypothesis ( $H_0$ ) and is thus expressed as  $1 - \beta$ . Type II error is the result of accepting  $H_0$  when an impact exists. Type II error, therefore, is the failure to detect an impact, and occurs with a probability of  $\beta$  (Toft & Shea 1983; Rotenberry & Wiens 1985; Andrew & Mapstone 1987; Simberloff 1990). Power is

dependent upon the level of significance ( $\alpha$ ), the sample size ( $n$ ), the effect size (ES), and the inherent variability in data. The higher the power of a test, the more statistically likely it is to show an effect that actually exists (Toft & Shea 1983; Rotenberry & Wiens 1985; Osenberg *et al.* 1994). The number of replicates, or  $n$ , is directly related to power; as  $n$  increases, so does power. The effect size is the magnitude of change in a parameter. A system with small natural variance will require a smaller  $n$  to detect a given effect size at a certain power, and as that variance increases, so does the required number of replicates (Osenberg *et al.* 1994). Sample sizes necessary to detect small, medium and large effect sizes corresponding to a 10%, 30% and 50% change, respectively, were calculated for all reefs collectively. The equation described in Clarke & Green (1988) was used to estimate sample sizes for each parameter:  $n > (k + 1) + \sqrt{(k^2 + 1)}$ ;  $k = (\sigma/\delta)^2(2 + \phi^{-1}(P))^2$ , where  $\sigma^2$  is the variance of the population,  $\delta$  is the size of the biological change and  $\phi^{-1}(P)$  is the inverse of the normal distribution function. This equation is an approximation of the sample size acceptable for  $n > 4$  (Clarke & Green 1988). The effect size (ES) was calculated for each parameter by multiplying its grand mean (mean of all seven reef sites) by ( $\delta =$ ) 0.1, 0.3 and 0.5 for small, medium and large effect sizes, respectively. Sample sizes were calculated for level of power equal to 0.95 [ $\phi^{-1}(0.95) = 1.64$ ] and for level of significance ( $\alpha$ ) equal to 0.05.

## Results

Dominant benthic components were calcareous substrate, turf algae, and dead coral, while mean live coral cover ranged between  $2.3 \pm 1.19$  (mean  $\pm$  sd) and  $6.9 \pm 5.70$  (mean  $\pm$  sd) percent of benthic space (Fig. 2). Given the replication used, the relative cover of the main benthic components differed among sites, albeit to a small degree (one-way ANOSIM, Global  $R = 0.083$ ,  $P = 0.0080$ ). At 30% dissimilarity, three clusters of similar sites were formed by Warhead, Fishbowl and Arch; Maze and Troy's Dream; Plane and Grotto (Fig. 3). Percent turf algal cover correlated negatively and significantly with percent dead coral cover ( $r_s = -0.79$ ,  $P = 0.0362$ ,  $n = 7$ ). There was also a marginally significant negative relationship between percent calcareous substrate cover and percent sand cover ( $r_s = -0.75$ ,  $P = 0.0522$ ,  $n = 7$ ). Furthermore, weak, yet significant, relationships were detected among algal functional groups on the reefs studied when all replicates, and not just means of reef sites, were used. In particular, percent turf algal cover correlated negatively with percent hard coral cover ( $r_s = -0.28$ ,  $P = 0.0102$ ,  $n = 82$ ) but positively with percent reef-building algal cover ( $r_s = 0.28$ ,  $P = 0.0101$ ,  $n = 82$ ) and percent macroalgal cover ( $r_s = 0.35$ ,  $P = 0.0013$ ,  $n = 82$ ).

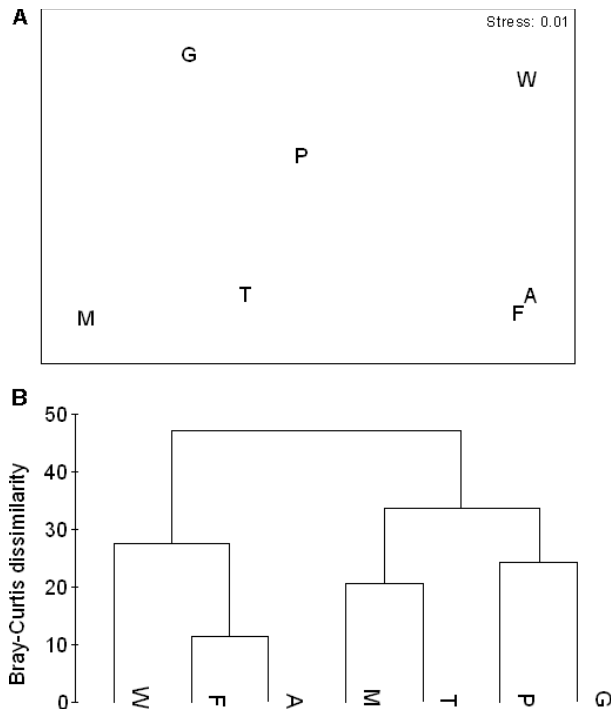


**Fig. 2.** Benthic partitioning at seven reef sites within the ACLSMP using 20-m line transects. HC, hard coral; DC, dead coral; CR, coral rubble; CS, calcareous substrate; S, sand; R, rubble; TA, turf algae; CoA + Hal, coralline algae + *Halimeda*; MA, macroalgae; FF, filter feeders; MI, macroinvertebrates.

Double square root-transformed percent of live coral cover ( $F_{6,81} = 1.2492$ ,  $P = 0.2913$ ; mean  $\pm$  sd =  $4.8 \pm 4.0$ ), hard coral colony density ( $F_{6,81} = 1.4659$ ,  $P = 0.2019$ ; mean  $\pm$  sd =  $5.6 \pm 1.8$ ), and log-transformed mean hard coral colony size ( $F_{6,81} = 1.3484$ ,  $P = 0.2469$ ; mean  $\pm$  sd =  $13.0 \pm 5.9$ ) did not differ among the seven reef sites (Fig. 4). Also, hard coral recruit density did not differ among the seven reef sites (Kruskal–Wallis  $H_{6,386} = 6.7320$ ,  $P = 0.3460$ ) or between the two sampling seasons (Kruskal–Wallis  $H_{1,386} = 0.5980$ ,  $P = 0.5980$ ; mean  $\pm$  sd =  $1.3 \pm 1.4$ ) (Fig. 4). There was no pronounced dominance (Pielou’s evenness, Kruskal–Wallis  $H_{6,80} = 1.9323$ ,  $P = 0.9258$ ) of any of the scleractinian coral taxa encountered in any of the seven reef sites, and although patterns were not consistent within the seven reef sites, *Agaricia*

spp., *M. annularis*, and *Siderastrea* spp. showed the highest relative cover as a percent of the total live hard coral cover when all reef sites were considered (Table 1).

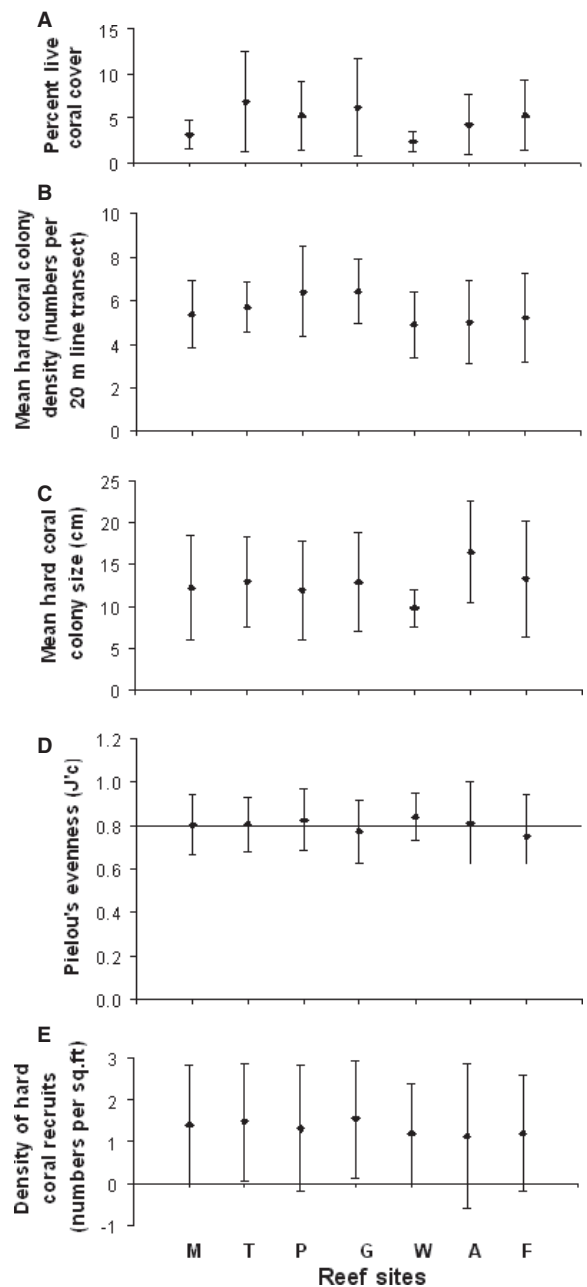
Logarithmically transforming colony-size data of *Agaricia* spp., *Siderastrea* spp., *M. annularis*, *F. fragum* and *P. asteroides* greatly improved normality of frequency distributions with four of five distributions being statistically non-significantly different from a normal distribution (Table 2, Fig. 5). *Favia fragum* was the smallest species, with an average colony size of 1.5 cm maximum colony diameter, while *M. annularis*, the largest species, averaged around 17 cm maximum colony diameter. Ninety-fifth percentiles of these two species were 9 cm and 65 cm maximum colony diameter, respectively. These two species also exhibited the largest standard deviations around



**Fig. 3.** (A) 2-Dimensional MDS and (B) dendrogram based on Bray-Curtis dissimilarity among means of 11 benthic variables for seven reef sites within the ACLMP. M, Maze; T, Troy's dream; P, Plane; G, Grotto; W, Warhead; F, Fishbowl; A, Arch. Lines of different line patterns indicate different groups. Benthic variables and replication are presented in Fig. 2.

the geometric mean colony size (Table 2). None of the coral-size distributions examined was symmetrical around the mean, but they were negatively skewed. This indicated colony-size distributions with relatively fewer colonies in the smaller size classes and a predominance of larger colonies. Of the five distributions in the dataset, two displayed negative kurtosis and three were positive with no clear pattern.

Due to high natural variance, even the large ES specified ( $\delta = 0.5$ ) lay within the natural variation of all the specified benthic and hard coral community parameters, with the exception of hard coral colony density. Thus, a large number of replicates are required to detect even a 50% change in the mean of the selected parameters (Table 3a). We consider the medium effect size ( $\delta = 0.3$ ) specified for all parameters of great biological/ecological importance on the studied coral reefs (see Discussion). The most sensitive parameter in detecting even small effect sizes was hard coral colony size (as maximum colony diameter) followed by hard coral colony density, whilst the least sensitive parameter in detecting even large effect sizes was percent coralline algae and *Halimeda*



**Fig. 4.** Variation (mean  $\pm$  sd) in (A) percent live hard coral cover, (B) colony density, (C) colony size, (D) evenness, and (E) recruitment density in seven reef sites within the ACLMNP, 9–12 m depth. Reef sites as in Fig. 3.

cover (Table 3a). The geometric mean of log-transformed colony-size frequency distributions (based on maximum colony diameter) was a far more practically sensitive index in detecting small size effects for all hard coral taxa examined, except *F. fragum*, compared to skewness (Table 3b).

**Table 1.** Relative hard coral cover (mean  $\pm$  sd) as percent of the total live coral cover at seven reef sites within the ACLSMP, 9–12 m depth, n = sample size.

Hard coral species	Reef sites							
	Maze (n = 14)	Troy's Dream (n = 10)	Plane (n = 10)	Grotto (n = 12)	Warhead (n = 9)	Arch (n = 12)	Fishbowl (n = 14)	Grand (n = 81)
<i>Acropora cervicornis</i>		3.3 $\pm$ 10.5	0.8 $\pm$ 2.4					0.5 $\pm$ 3.8
<i>Agaricia</i> spp.	8.9 $\pm$ 11.3	15.5 $\pm$ 10.3	18.6 $\pm$ 18.5	9.9 $\pm$ 9.3	23.0 $\pm$ 17.2	27.0 $\pm$ 25.3	18.9 $\pm$ 20.1	17.1 $\pm$ 17.5
<i>Colpophyllia natans</i>		1.5 $\pm$ 3.1				1.1 $\pm$ 3.9		0.4 $\pm$ 1.9
<i>Dendrogyra cylindricus</i>			2.2 $\pm$ 7.0		1.0 $\pm$ 3.0	3.1 $\pm$ 10.8	2.0 $\pm$ 7.3	1.2 $\pm$ 5.7
<i>Dichocoenia stokesii</i>	0.5 $\pm$ 1.9		3.4 $\pm$ 8.6	0.4 $\pm$ 1.4	0.6 $\pm$ 1.8		1.0 $\pm$ 2.7	0.8 $\pm$ 3.4
<i>Diploria strigosa</i>	1.9 $\pm$ 5.1	0.1 $\pm$ 0.4						0.3 $\pm$ 2.2
<i>Diploria labyrinthiformis</i>		3.5 $\pm$ 5.9	0.9 $\pm$ 2.8	1.6 $\pm$ 4.5				0.8 $\pm$ 3.0
<i>Diploria stokesii</i>	10.7 $\pm$ 17.6	0.7 $\pm$ 2.1	0.4 $\pm$ 1.1	0.3 $\pm$ 1.0	7.9 $\pm$ 16.1	2.9 $\pm$ 7.7	2.9 $\pm$ 6.1	3.8 $\pm$ 10.3
<i>Eusmilia fastigiata</i>			1.7 $\pm$ 4.6	1.1 $\pm$ 2.1		2.0 $\pm$ 4.6		0.7 $\pm$ 2.6
<i>Favia fragum</i>	1.1 $\pm$ 2.6	1.9 $\pm$ 3.2	7.9 $\pm$ 12.6	5.8 $\pm$ 6.9	1.5 $\pm$ 1.9	2.7 $\pm$ 5.7	3.2 $\pm$ 5.8	3.4 $\pm$ 6.5
<i>Isophyllastrea rigida</i>	3.1 $\pm$ 5.9		2.5 $\pm$ 4.5	1.2 $\pm$ 4.2	1.2 $\pm$ 3.6			1.1 $\pm$ 3.6
<i>Madracis mirabilis</i>				1.9 $\pm$ 4.5		6.5 $\pm$ 22.5	2.3 $\pm$ 8.6	1.6 $\pm$ 9.4
<i>Manicina areolata</i>	0.6 $\pm$ 2.4							0.1 $\pm$ 1.0
<i>Meandrina meandrites</i>			0.6 $\pm$ 1.2	3.4 $\pm$ 6.6	12.2 $\pm$ 16.4		0.5 $\pm$ 1.9	2.0 $\pm$ 6.9
<i>Millepora alicornis</i>	0.2 $\pm$ 0.2	0.2 $\pm$ 0.3	0.1 $\pm$ 0.3	0.1 $\pm$ 0.3	0.2 $\pm$ 0.3	0.1 $\pm$ 0.2	0.2 $\pm$ 0.4	0.2 $\pm$ 0.3
<i>Millepora complanata</i>	1.4 $\pm$ 5.3					0.3 $\pm$ 1.2		0.3 $\pm$ 2.2
<i>Montastrea annularis</i>	16.6 $\pm$ 22.3	31.7 $\pm$ 28.1	16.3 $\pm$ 14.7	14.1 $\pm$ 18.7	4.9 $\pm$ 14.2	14.5 $\pm$ 17.6	18.5 $\pm$ 21.0	6.8 $\pm$ 20.6
<i>Montastrea cavernosa</i>		3.6 $\pm$ 6.1	3.8 $\pm$ 5.3	9.1 $\pm$ 15.6	7.1 $\pm$ 12.0		0.4 $\pm$ 1.7	0.5 $\pm$ 3.8
<i>Porites astreoides</i>	24.3 $\pm$ 22.4	5.3 $\pm$ 6.3	14.1 $\pm$ 13.5	10.3 $\pm$ 11.3	7.0 $\pm$ 12.1	9.9 $\pm$ 8.9	7.7 $\pm$ 7.8	11.7 $\pm$ 14.1
<i>Porites porites</i>	4.2 $\pm$ 9.3	5.2 $\pm$ 8.4	10.3 $\pm$ 24.9	8.3 $\pm$ 11.8	11.1 $\pm$ 11.4	0.9 $\pm$ 3.2	8.4 $\pm$ 11.3	6.7 $\pm$ 12.5
<i>Siderastrea</i> spp.	8.9 $\pm$ 9.7	18.6 $\pm$ 16.9	10.4 $\pm$ 11.0	28.9 $\pm$ 21.4	16.6 $\pm$ 26.9	15.2 $\pm$ 18.0	15.3 $\pm$ 21.9	16.1 $\pm$ 18.9
<i>Stephanocoenia michilini</i>	2.3 $\pm$ 7.3	5.0 $\pm$ 8.5	5.0 $\pm$ 6.7	1.1 $\pm$ 3.0		2.5 $\pm$ 8.7	4.0 $\pm$ 9.6	2.9 $\pm$ 7.2

**Table 2.** Distribution parameters of logarithmically transformed colony size (maximum diameter) data for five taxa of hard corals from seven reef sites within the ACLSMP.

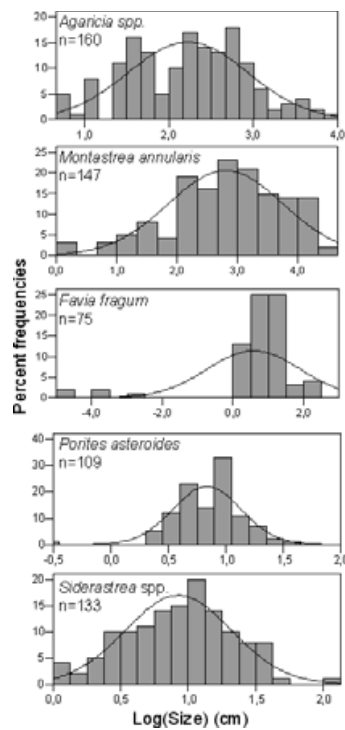
Hard coral species	$\mu$	SD	$g_1$ ( $\pm$ SE)	$g_2$ ( $\pm$ SE)	95%	$P_{\text{norm}}$	n
<i>Agaricia</i> spp.	2.1 $\pm$ 0.11	0.71	-0.0573 ( $\pm$ 0.1919)	-0.4850 ( $\pm$ 0.3815)	3.47	0.0740	160
<i>Montastrea annularis</i>	2.8 $\pm$ 0.15	0.95	-0.6539 ( $\pm$ 0.2000)	0.3584 ( $\pm$ 0.3975)	4.17	0.2000	147
<i>Favia fragum</i>	0.58 $\pm$ 0.30	1.32	-2.6433 ( $\pm$ 0.2774)	7.6573 ( $\pm$ 0.5482)	2.08	0.0100	75
<i>Porites astreoides</i>	0.84 $\pm$ 1.69	0.30	-0.7960 ( $\pm$ 0.2315)	3.0143 ( $\pm$ 0.4590)	1.26	0.2000	109
<i>Siderastrea</i> spp.	0.93 $\pm$ 0.07	0.39	-0.1304 ( $\pm$ 0.2101)	-0.0696 ( $\pm$ 0.4171)	1.60	0.2000	133

$\mu$  = geometric mean size (cm);  $g_1$  = skewness ( $\pm$  SE);  $g_2$  = kurtosis ( $\pm$  SE); 95% = 95th percentile;  $P_{\text{norm}}$  = probability that data follow a normal distribution (Kolmogorov–Smirnov test using Lilliefors adjusted probability); n = total number of colonies measured.

## Discussion

The platforms of the margin reefs of South Caicos studied appeared quite homogeneous regarding hard coral community structure but rather variable regarding benthic composition, even when this was portioned into living (mean  $\pm$  sd = 35.0  $\pm$  29.8) and non-living components (mean  $\pm$  sd = 65.0  $\pm$  29.8). Benthic partition is in accordance with descriptions of comparable sites at the TCIs (Sullivan *et al.* 1994; Riegl *et al.* 1999) and in the broader Caribbean three decades ago (Bak & Luckhurst 1980) but is in sharp contrast to more recent studies in the same region documenting the dominance of benthos by macro-

algae (Andres & Witman 1995; Chiappone *et al.* 1997). According to the relative dominance paradigm of Littler & Littler (1984), Littler *et al.* (2006), reef-building corals would be dominant over reef-building (coralline and encrusting algae including *Halimeda*) and palatable algae (turf and macro algae) in a high herbivory but low nutrient environment. Although mean turf algal cover (mean  $\pm$  sd = 21.7  $\pm$  33.0) was the highest among the four algal functional groups on the reef sites studied, the dominant benthic component was carbonate substrate, thus indicating an accreting reef framework. Despite the lack of urchins on these reef flats (only one specimen was recorded under the total of 82, 20-m line transects and



**Fig. 5.** Size-frequency distributions (based on maximum colony diameter) of measured hard coral colonies with overlaid normal distributions for five taxa and seven reef sites within the ACLSMP.

only a handful were observed inside crevices adjacent to the line transects), herbivorous, large-size fish apparently abundant in ambient waters (Hoshino *et al.* 1999) because local fisheries are still largely based on the gastropod *Strombus gigas* and the spiny lobster *Panulirus argus* (Rudd 2003). Numerous *Diadema antillarum*, as well as other urchin species, however, are found on shallower reefs around South Caicos (J. Rohde, personal observations). On these relatively pristine Caribbean reefs, a weak negative relationship between turf algal cover and hard coral cover and a weak positive relationship between reef-building algal cover and macroalgal cover were detected when a large number of replicates was used ( $n = 82$ ). In the low herbivory and high nutrient environment of Jamaican fore reefs, in contrast, there was a strong, negative relationship ( $r_s = -0.94$ ,  $P = 0.0048$ ,  $n = 6$ ) between coralline and macroalgal cover (Andres & Witman 1995) with macroalgal cover averaging  $79 \pm 2.9$  (mean  $\pm$  sd) percent.

On a benthos largely dominated by carbonate substrate, at the beginning of the reef, and at relatively shallow waters (9–12 m depth), percent live coral cover was low. Percent live coral cover progressively increases towards the reef crest, after which live corals of large size dominate the benthos up to 50 m depth (A. Dikou, personal observations), similar to classic accounts of coral reef zonation (Loya 1972; Porter 1972; Bouchon 1981; Done 1982). The

four most abundant hard coral species found in this study were in agreement with other studies conducted at the TCIs and the greater Caribbean (Loya 1976; Edmunds *et al.* 1990; Andres & Witman 1995; Riegl *et al.* 1999). In terms of cover, *Agaricia* spp. was the most abundant taxon followed by the reef-builder *M. annularis*, *Siderastrea* spp. and *P. asteroids*, and the small-size *F. fragum* was ubiquitous (73 colonies under 82, 20-m line transects). Mean recruitment rate of hard corals (mean  $\pm$  sd =  $20.8 \pm 22.4$  recruits·m<sup>-2</sup>) was double than that documented using same methods and considerations at Florida Keys, FL, USA (Edmunds *et al.* 1998; Miller *et al.* 2000).

Expansion of tourism and fisheries at the TCIs in the near future is likely to induce ‘bottom-up’ (due to increase in sediment and nutrient loads) and ‘top-down’ (due to reduction in fish herbivory) alterations in reef community structure and trophic dynamics similar to other developed islands in the Caribbean (Sullivan 2004). Although increase in sediments and nutrients usually take place in concert, a recent review has highlighted that an increase in turbidity and/or sedimentation impairs mainly growth, recruitment and survival of hard corals, while an increase in nutrients impairs mainly reproduction of hard corals (Fabricius 2005). Also, hard corals’ competitive inferiority over algae and filter feeders for space and light is further enhanced in reduced herbivory and/or increased nutrients environments (McManus *et al.* 2000; McCook *et al.* 2001). Thus, individual and/or synergistic effects are expected to be demonstrated, among others, through reduction in abundance of herbivorous fish in ambient waters, relative increase of palatable algal cover over reef-building algal cover (including scleractinian corals), increase in abundance of filter feeders, decrease in coral colony density, decrease in coral recruitment rates, and relative increase of sediment-resistant over sediment-intolerant hard corals along with concomitant changes in their colony-size frequency distributions. McClanahan & Obura (1997) found that genera that were more abundant under high sediment load conditions had larger mean size colonies compared with those in clearer waters, whereas genera more abundant on ‘reference’ reefs had smaller mean size colonies. Thus, differences in the direction and magnitude of changes in colony-size distributions of the abundant, sediment-resistant *Siderastrea* spp., *P. asteroides* and *Agaricia* spp. (Loya 1976; Cortes & Risk 1985) compared to that of the sediment-intolerant but nutrients-benefiting *M. annularis* (Loya 1976; Tomascik & Sander 1985) are expected. Meesters *et al.* (2001) evaluated the sensitivity of the log-transformed colony size-frequency distributions (based on colony aerial cover in square centimetres) of 13 hard coral species as indicators of heavy coastal urbanization on fringing reefs of Curaçao, Netherland Antilles, by comparing degraded reefs to upstream



**Table 3.** Required number of replicates (n) to detect small, medium and large effect sizes (ES) denoted by corresponding  $\delta$  values of 0.1, 0.3 and 0.5, respectively, at the ACLSNP for indices of (a) benthic partition and hard coral community structure and (b) log-transformed colony size-frequency distributions of hard corals.

Indices	Grand mean	Grand $\sqrt{\sigma^2}$	$\delta = 10\%$			$\delta = 30\%$			$\delta = 50\%$		
			small ES	k	n	medium ES	k	n	large ES	k	n
<b>(a) Benthic partition</b>											
Live coral cover (%)	4.8	4.0	0.48	940.6	1882	1.43	104.51	210	2.4	37.6	76
Dead coral cover (%)	16.4	26.8	1.64	3548.2	7097	4.91	394.25	789	8.2	141.9	285
Turf algae (%)	21.7	33.0	2.17	3063.1	6127	6.51	340.34	682	10.8	122.5	246
Macroalgae (%)	0.1	0.6	0.01	24346.5	48694	0.04	2705.17	5411	0.1	973.9	1949
Coralline algae and <i>Halimeda</i> (%)	0.1	0.3	0.01	24447.0	48895	0.02	2716.34	5434	0.0	977.9	1957
Filter feeders (%)	8.3	9.0	0.83	1557.9	3117	2.49	173.10	347	4.1	62.3	126
Coral colony density (number of corals per 20-m line transects)	5.5	1.8	0.55	134.1	269	1.65	14.90	31	2.8	5.4	12
Coral recruit density (number of corals per square foot)	1.3	1.4	0.13	1580.9	3163	0.39	175.65	352	0.7	63.2	127
Coral colony size (maximum diameter in cm)	13.0	2.3	1.30	41.5	84	3.90	4.61	10	6.5	1.7	5
<b>(b) Log-transformed colony size frequency distributions</b>											
<i>Agaricia</i> spp. ( $\mu$ )	2.10	0.7	0.2	151.5	304	0.63	16.83	35	1.1	6.1	13
<i>Montastrea annularis</i> ( $\mu$ )	2.80	0.9	0.3	152.5	306	0.84	16.95	35	1.4	6.1	13
<i>Favia fragum</i> ( $\mu$ )	0.58	1.3	0.1	6862.7	13726	0.17	762.52	1526	0.3	274.5	550
<i>Porites astreoides</i> ( $\mu$ )	0.84	0.3	0.1	169.0	339	0.25	18.78	39	0.4	6.8	15
<i>Siderastrea</i> spp. ( $\mu$ )	0.93	0.4	0.1	233.0	467	0.28	25.89	53	0.5	9.3	20
<i>Agaricia</i> spp. ( $g_1$ )	-0.06	2.4	0.0	2377735.0	4755471	-0.02	264192.78	528387	0.0	95109.4	190220
<i>Montastrea annularis</i> ( $g_1$ )	-0.65	2.4	-0.1	18220.4	36442	-0.20	202.49	4050	-0.3	728.8	1459
<i>Favia fragum</i> ( $g_1$ )	-2.64	2.4	-0.3	1094.4	2190	-0.79	121.60	244	-1.3	43.8	89
<i>Porites astreoides</i> ( $g_1$ )	-0.80	2.4	-0.1	12215.3	24432	-0.24	1357.26	2716	-0.4	488.6	978
<i>Siderastrea</i> spp. ( $g_1$ )	-0.13	2.4	0.0	457 458.1	914917	-0.04	50828.68	101658	-0.1	18298.3	36598

$\mu$  = geometric mean (in cm);  $g_1$  = skewness;  $\sigma^2$  = variance;  $k = (\sigma/\delta)^2(2 + \phi^{-1}(P))^2$  as in Clarke & Green (1988).

Data were pooled from seven reef sites.

control areas. They concluded that *Colpophyllia natans* and *Diploria labyrinthiformis* were relatively sensitive to environmental conditions. Both species are present, though in low abundance, on the studied reefs.

While the direction of anticipated change in the variables studied on these reefs may be projected, the magnitude of a biologically/ecologically/environmentally important change is more difficult to decipher. The latter depends on the specific variable-indicator employed; the nature, severity, frequency and synergy of impacts; the ecology of the coral reef system these impacts are inflicted upon; and on what policy/public consider a un/acceptable amount of change for the coral reefs of interest. For example, chronic pollution by sewage along the west coast of Barbados was manifested, among others, through a 36.4% and 90.4% increase in suspended particulate matter concentration and chlorophyll a concentration, respectively, which, in turn, led to an 89.3%, 60.8%, 61.4%,

and 79.5% decrease in skeletal linear extension rate of *M. annularis*, hard coral colony density, percent live coral cover, and hard coral recruit density, respectively, and a 413.1% increase in percent fleshy algal cover at the vicinity of the sewage discharge compared to a reference site approximately 50 km away from the sewage discharge (reef flat zones of stations BR and GS of Tomascik & Sander 1985, 1987; Tomascik 1991). We believe that a 10% and 30% change in the variables studied on these reefs should set off an alarm and prompt a response to degradation, respectively, given the protected status of these relatively pristine studied reefs and the demonstrated widespread degradation of coral reefs in the Caribbean (Buddemeier *et al.* 2003).

Furthermore, a possible impact is likely to affect not only the variables simulated in this study but also the variability of the system. Warwick & Clarke (1993) noted that, in a variety of environmental impact studies at the

community level, the variability among samples from impacted areas was much greater than that from reference sites. Thus, the level of replication estimated for the community level variables of Table 3a may be the minimum required. On the other hand, impact assessment studies on the growth of hard corals at the population level indicate species-dependent change in variance (e.g. Tomascik & Sander 1985; Meesters *et al.* 2001). Thus, the level of replication estimated for the growth-related, population level variables of Table 3b may be either relaxed or conservative depending on the sensitivity/tolerance of the hard coral species studied to the specific stressor.

A *priori* power analysis indicated that among the univariate indices examined, coral colony size (maximum diameter in centimetres) and density along with the geometric mean of log-transformed size-frequency distributions of *M. annularis*, *Siderastrea* spp., *Agaricia* spp. and *P. asteroides* were practically the most sensitive indices in detecting even small changes on benthos. Power analysis does not add to an understanding of nature itself but it does allow some quality control to be exercised over inferences derived from environmental studies. Strengthened inferences give more reliable information about interactions in nature upon which to base management decisions (Fairweather 1991).

The present study provided baseline data on the benthic composition of coral reefs from 'before' impact, representative, 'control' sites and on its natural spatial variability. We applied a simple, yet robust, method for calculating required sample size for specified effect sizes and power; and evaluated the relative sensitivity of a number of benthic indicators to detect anticipated changes on coral reefs of the TCIs. Thus, it may optimize the spatial component of an environmental impact assessment program, which, when coupled with (currently lacking) water quality data within a required BACI type of sampling design (Green 1979; Underwood 1992; Benedetti-Cecchi 2001), will be able to confidently assert the presence (or absence) of biologically/ecologically/environmentally significant human-induced change on heterogeneous coral reef communities. The results of such a program, in turn, may guide adaptable development decisions, and thus ensure ecological and socio-economic sustainability for the local communities of the TCIs.

### Acknowledgements

We thank The School for Field Studies, Center for Marine Resource Studies, TCI, for educational, financial, and logistical support throughout this study. Figure 1 was originally drawn and generously offered by Dr J. Claydon. We thank Prof. L. Benedetti-Cecchi and an anonymous reviewer for astute comments on this work.

### References

- Andres N.G., Witman J.D. (1995) Trends in Community Structure on a Jamaican Reef. *Marine Ecology Progress Series*, **118**, 273–288.
- Andrew N.L., Mapstone B.D. (1987) Sampling and the description of spatial pattern in marine ecology. *Oceanography and Marine Biology: An Annual Review*, **25**, 39–90.
- Anonymous (2005) Coral reefs: valuable assets for all of US-economic value of TCI reefs. Times of the Islands, Newsletter of the Department of Environment and Natural Resources, pp. 51. Available at: <http://www.environment.tc/information/reading/documents/GreenPagesProof-F05-6.pdf>. Accessed 17 June 2008.
- Anonymous (2007) Final Workshop for the National Development Plan, September 21, 2007, Providenciales. Press Releases Archive, Department of Economic Planning and Statistics, Turks and Caicos Islands. Available at: <http://www.depstc.org/pressreleases/2007/sep07/0921a.html>. Accessed 17 June 2008.
- Bak R.P.M., Engel M.S. (1979) Distribution, abundance and survival of juvenile hermatypic corals (Scleractinia) and the importance of life history strategies in the parent coral community. *Marine Biology*, **54**, 341–352.
- Bak R., Luckhurst B. (1980) Constancy and change in coral reef habitats along depth gradients at Curacao. *Oecologia*, **47**, 145–155.
- Bak R.P.M., Meesters E.H. (1998) Coral population structure: the hidden information of colony size-frequency distributions. *Marine Ecology Progress Series*, **162**, 301–306.
- Benedetti-Cecchi L. (2001) Beyond BACI: optimization of environmental sampling designs through monitoring and simulation. *Ecological Applications*, **11**, 783–799.
- Birrell C.L., McCook L.J., Willis B.L. (2004) Effects of algal turfs and sediment on coral settlement. *Marine Pollution Bulletin*, **51**, 408–414.
- Black K.P., Moran P.J., Hammond L.S. (1991) Numerical models show coral reefs can be self-seeding. *Marine Ecology Progress Series*, **74**, 1–11.
- Bouchon C. (1981) Quantitative study of the scleractinian coral communities of a fringing reef of Reunion Island (Indian Ocean). *Marine Ecology Progress Series*, **4**, 273–288.
- Brown B.E., Howard L.S. (1985) Assessing the effects of 'Stress' on reef corals. *Advances in Marine Biology*, **22**, 1–63.
- Buddemeier R.W., Fautin D.G. (2002) Large-scale dynamics: the state of the science, the state of the reef, and the research issues. *Coral Reefs*, **21**, 1–8.
- Buddemeier R.W., Ware J.R., Gardner T.A., Côté T.A., Gill J.A., Grant A., Watkinson A.R. (2003) Coral reef decline in the Caribbean. *Science*, **302**, 391–393.
- Chiappone M., Sullivan K., Luka R. (1997) Status of reefs in Central Bahamas on a large-scale survey. *Proceedings of the 8th International Coral Reef Symposium*, **1**, 345–350.

- Clarke K.R., Green R.H. (1988) Statistical design and analysis for a 'biological effects' study. *Marine Ecology Progress Series*, **46**, 213–226.
- Clarke K.R., Warwick R.M. (2001) *Changes in Marine Communities: an Approach to Statistical Analysis and Interpretation*. PRIMER-E. Plymouth Marine Laboratory, Plymouth: 172.
- Connell J.E. (1997) Disturbance and recovery of coral assemblages. *Coral Reefs*, **16**, 101–113.
- Cortes J.N., Risk M.J. (1985) A reef under siltation stress: Cahuita, Costa Rica. *Bulletin of Marine Science*, **36**, 339–356.
- Dikou A., van Woessik R. (2006a) Partial colony mortality reflects coral community dynamics: a fringing reef study near a small river in Okinawa, Japan. *Marine Pollution Bulletin*, **52**, 269–280.
- Dikou A., van Woessik R. (2006b) Survival under chronic stress from sediment load: spatial patterns of hard coral communities in the Southern islands of Singapore. *Marine Pollution Bulletin*, **52**, 7–21.
- Done T. (1982) Patterns in the distribution of coral communities across the central Great Barrier Reef. *Coral Reefs*, **1**, 95–107.
- Edmunds P.J., Roberts D.A., Singer R. (1990) Reefs of the Northeastern Caribbean. I. Scleractinian populations. *Bulletin of Marine Science*, **46**, 780–789.
- Edmunds P.J., Aronson P.J., Swanson D.W., Levitan D.R., Precht W.F. (1998) Photographic versus visual census techniques for the quantification of juvenile corals. *Bulletin of Marine Science*, **62**, 937–946.
- Fabricius K.E. (2005) Effects of terrestrial runoff on the ecology of corals and coral reefs: Review and synthesis. *Marine Pollution Bulletin*, **50**, 125–146.
- Fairweather P.G. (1991) Statistical power and design requirements for environmental monitoring. *Australian Journal of Marine and Freshwater Research*, **42**, 555–567.
- Ferraro S.P., Cole F.A. (1990) Taxonomic level and sample size sufficient for assessing pollution impacts on the Southern California Bight macrobenthos. *Marine Ecology Progress Series*, **67**, 251–262.
- Gardner T.A., Côté I.M., Gill J.A., Grant A., Watkinson A.R. (2003) Long-term region-wide declines in Caribbean corals. *Science*, **301**, 958–960.
- Green R.H. (1979) *Sampling Design and Statistical Methods for Environmental Biologists*. Wiley, New York: 257 pp.
- Hoshino K., Brandt M., Manfrino C., Riegl B., Steiner S.C.C. (1999) Assessment of the Turks and Caicos Islands (Part II: Fish communities). In: Lang J.C. (Ed.), *Status of Coral Reefs in the Western Atlantic: Results of Initial Surveys, Atlantic and Gulf Rapid Reef Assessment (AGRRA) Program*. National Museum of Natural History, Washington, D.C.: 481–499.
- Hughes T.P., Reed D.C., Boyle M.J. (1987) Herbivory on coral reefs: community structure following mass-mortalities of sea urchins. *Journal of Experimental Marine Biology and Ecology*, **113**, 39–59.
- Hughes T.P., Baird A.H., Dinsdale A., Molchanivskyj N.A., Pratchett M.S., Tanner J.E., Willis B.L. (2000) Supply-side ecology works both ways: the link between benthic adults, fecundity, and larval recruits. *Ecology*, **81**, 2241–2249.
- Hughes T.P., Baird A.H., Bellwood D.R., Card M., Connolly S.R., Folke C., Grosberg R., Hoegh-Guldberg O., Jackson J.B.C., Kleypas J., Lough J.M., Marshall P., Nystrom M., Palumbi S.R., Pandolfi J.M., Rosen B., Roughgarden J. (2003) Climate change, human impacts, and the resilience of coral reefs. *Science*, **301**, 929–933.
- Jackson J.B.C., Kirby M.X., Berger W.H., Bjorndal K.A., Botsford L.W., Bourque B.J., Bradbury R.H., Cooke R., Erlanson J., Estes J.A., Hughes T.P., Kidwell S., Lange C.B., Lenihan H.S., Pandolfi J.M., Peterson C.H., Steneck R.S., Tegner M.J., Warner R.R. (2001) Historical overfishing and the recent collapse of coastal ecosystems. *Science*, **293**, 629–637.
- Littler M.M., Littler D.S. (1984) Models of tropical reef biogenesis: the contribution of algae. In: Round F.E., Chapman D.J. (Eds). *Progress in Phycological Research*, Vol. 3. Biopress, Bristol: 323–364.
- Littler M.M., Littler D.S., Brooks B.L. (2006) Harmful algae on tropical coral reefs: bottom-up eutrophication and top-down herbivory. *Harmful Algae*, **5**, 565–585.
- Loya Y. (1972) Community structure and species diversity of hermatypic corals at Eilat, Red Sea. *Marine Biology*, **13**, 100–123.
- Loya Y. (1976) Effects of water turbidity and sedimentation on the community structure of Puerto Rican corals. *Bulletin of Marine Science*, **26**, 450–466.
- Maragos J.E., Evans C.W., Holthus P.F. (1985) Reef corals in Kaneohe Bay six years before and after termination of sewage discharges. *Proceedings of the 5th International Coral Reef Symposium*, **4**, 189–194.
- McClanahan T.R., Obura D. (1997) Sedimentation effects on shallow coral communities in Kenya. *Journal of Experimental Marine Biology and Ecology*, **209**, 103–122.
- McCook L.J., Jompa J., Diaz-Pulido G. (2001) Competition between corals and algae on coral reefs: a review of evidence and mechanisms. *Coral Reefs*, **19**, 400–417.
- McManus J.W., Meñez L.A.B., Reyes K.N.K., Vergara S.G., Ablan M.C. (2000) Coral reef fishing and coral-algal phase shifts: implications for global reef status. *ICES Journal of Marine Science*, **57**, 572–578.
- Meesters E.H., Hilterman M., Kardinaal E., Keetman M., de Vries M., Bak R.P.M. (2001) Colony size-frequency distributions of scleractinian coral populations: spatial and interspecific variation. *Marine Ecology Progress Series*, **209**, 43–54.
- Miller M.W., Hay M.E. (1998) Effects of fish predation and seaweed competition on the survival and growth of corals. *Oecologia*, **113**, 231–238.
- Miller M.W., Weil E., Szmant A.M. (2000) Coral recruitment and juvenile mortality as structuring factors for reef benthic communities in Biscayne National Park, USA. *Coral Reefs*, **19**, 115–123.

- Morrisey D.J. (1993) Environmental impact assessment – a review of its aims and recent developments. *Marine Pollution Bulletin*, **26**, 540–545.
- Mumby P.J., Dahlgren C.P., Harborne A.R., Kappel C.V., Micheli F., Brumbaugh D.R., Holmes K.E., Mendes J.M., Broad K., Sanchirico J.N., Buch K., Box S., Stoffle R.W., Gill A.B. (2006) Fishing, trophic cascades, and the process of grazing on coral reefs. *Science*, **311**, 98–101.
- Osenberg C.W., Schmitt R.J., Holbrook S.J., Abu-Saba K.E., Flegal A.R. (1994) Detection of environmental impacts: natural variability, effect size, and power analysis. *Ecological Applications*, **4**, 16–30.
- Pastorok R.A., Bilyard G.R. (1985) Effects of sewage pollution on coral-reef communities. *Marine Ecology Progress Series*, **21**, 75–189.
- Peterman R.M. (1990a) Statistical power analysis can improve fisheries research and management. *Canadian Journal of Fisheries and Aquatic Science*, **47**, 2–15.
- Peterman R.M. (1990b) The importance of reporting statistical power: the forest decline and acidic deposition example. *Ecology*, **71**, 2024–2027.
- Porter J.W. (1972) Patterns of species diversity in Caribbean Reef Corals. *Ecology*, **53**, 745–748.
- Riegl B., Manfrino C., Hermoyian C., Brandt M., Hoshino K. (1999) Assessment of the Turks and Caicos Islands (Part I: Stony corals and algae). In: Lang J.C. (Ed.), *Status of Coral Reefs in the Western Atlantic: Results of Initial Surveys, Atlantic and Gulf Rapid Reef Assessment (AGRRA) Program*. National Museum of Natural History, Washington, D.C.: 460–480.
- Roberts C.M. (1997) Connectivity and management of Caribbean coral reefs. *Science*, **278**, 1454–1457.
- Rogers C.S. (1990) Responses of coral reefs and reef organisms to sedimentation. *Marine Ecology Progress Series*, **62**, 185–202.
- Rotenberry J.T., Wiens J.A. (1985) Statistical power analysis and community-wide patterns. *The American Naturalist*, **125**, 164–168.
- Rudd M.A. (2003) Fisheries landings and trade of the Turks and Caicos Islands. *Fisheries Centre Research Reports*, **11**, 149–161.
- Sammarco P.W. (1980) *Diadema* and its relationship to coral spat mortality: grazing, competition, and biological disturbance. *Journal of Experimental Marine Biology and Ecology*, **45**, 245–272.
- Sanders D., Baron-Szabo R.C. (2005) Scleractinian assemblages under sediment input: their characteristics and relation to the nutrient input concept. *Palaeoecology*, **216**, 139–181.
- Simberloff D. (1990) Hypotheses, errors, and statistical assumptions. *Herpetologica*, **46**, 351–357.
- Sokal R.R., Rohlf F.J. (1995) *Biometry. The Principles and Practice of Statistics in Biological Research*. W.H. Freeman and Company, New York: 887 pp.
- Sullivan S.K. (2004) Large-scale ecological impacts of development on tropical islands systems: comparison of developed and undeveloped islands in the central Bahamas. *Bulletin of Marine Science*, **75**, 295–320.
- Sullivan K.M., Chiappone M., Lott C. (1994) Abundance patterns of stony corals on platform margin reefs of the Caicos Bank, Southeastern Bahamas. *Bahamas Journal of Science*, **5**, 2–11.
- TCI Government (2004) *Policy for the Management and Development of the Fisheries Sector, 4th Draft*. Department of Environment and Coastal Resources, Providenciales.
- Toft C.A., Shea P.J. (1983) Detecting community-wide patterns: estimating power strengthens statistical inference. *The American Naturalist*, **122**, 618–625.
- Tomascik T. (1991) Settlement patterns of Caribbean scleractinian corals on artificial substrata along a eutrophication gradient, Barbados, West Indies. *Marine Ecology Progress Series*, **77**, 261–269.
- Tomascik T., Sander F. (1985) Effects of eutrophication on reef-building corals. I. Growth rate of the reef-building coral *Montastrea annularis*. *Marine Biology*, **87**, 143–155.
- Tomascik T., Sander F. (1987) Effects of eutrophication on reef-building corals. II. Structure of scleractinian coral communities on fringing reefs, Barbados, West Indies. *Marine Biology*, **94**, 53–75.
- Underwood A.J. (1992) Beyond BACI: The detection of environmental impacts on populations in the real, but variable, world. *Journal of Experimental Marine Biology and Ecology*, **161**, 145–178.
- Warwick R.M., Clarke K.R. (1993) Increased variability as a symptom of stress in marine communities. *Journal of Experimental Marine Biology and Ecology*, **172**, 215–226.