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REEFS: SIMPLE, QUICK, INEXPENSIVE TECHNIQUES

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ABSTRACT

With coral cover and diversity declining on many coral reefs, a clearer understanding of large-scale reef dynamics is imperative. This paper presents a sampling program designed to quantify the sessile biotas of Caribbean reefs on large spatiotemporal scales. For each reef sampled, data are gathered along replicate, 25-m transects located within the habitat of interest. Herbivore impact is estimated by fish and echinoid censuses along the transects. High-resolution videotapes are used to estimate the percent cover of corals, algae, and other substratum occupants, and to estimate coral diversity. Finally, topographic complexity is measured along the transects. In at least some reef habitats, this index of three-dimensional structure provides a measure of the total disturbance regime, with flatter areas having been subjected to more intense, more frequent, and/or more recent sources of coral mortality. The techniques and statistical analyses described in this paper are simple, quick and inexpensive. Repeated sampling on multiple reefs will enable the investigator to detect changes in community structure and to test hypotheses of the causes of those changes.

INTRODUCTION

Coral reefs are complex, diverse, productive tropical ecosystems in which multiple physical and biological processes covary in space and time (Huston 1985). Discerning the contributions of those processes to the community structure of reefs has been and will continue to be extremely difficult. The question of the appropriate scales at which to search for pattern and process is fundamental to unraveling these multiple causal connections (Jackson 1991, 1992; Karlson and Hurd 1993). Are ecological parameters such as coral cover and diversity determined primarily by small-scale processes, such as the local level of herbivore activity (Sammarco 1980; Lewis 1986), or are larger-scale, regional disturbances more important? Such questions are becoming increasingly

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germane as we confront the possibility that human interference is altering coral reef ecosystems (e.g., Brown 1987).

Over the past two decades, the "health" of many coral reefs worldwide has deteriorated, as measured by indicators such as the diversity and abundance of reef-building corals (Rogers 1985; Dustan and Halas 1987; Hatcher et al. 1989; Grigg and Dollar 1990; Porter and Meier 1992; Ginsburg 1994). Hurricanes (Woodley et al. 1981; Rogers et al. 1982, 1991; Edmunds and Witman 1991; Hubbard et al. 1991; Bythell et al. 1993), coral bleaching (Oliver 1985; Brown and Suharsono 1990; Williams and Bunkley-Williams 1990; Glynn 1993), coral diseases (Gladfelter 1982; Rützler et al. 1983; Edmunds 1991), mortality of the sea urchin *Diadema antillarum* (Lessios et al. 1984; Levitan 1988; Lessios 1988; Carpenter 1990) and outbreaks of the seastar *Acanthaster planci* (Moran 1986; Endean and Cameron 1991) may all be involved. At present, it is unknown whether any or all of these problems are due to recent human activities, or whether they are part of natural, long-term trends or cycles (Brown 1987; Richards and Bohnsack 1990).

In addition, reefs are directly affected by human activities, particularly fishing, sedimentation, eutrophication and pollution (Brown 1987, 1988; Hatcher et al. 1989; Rogers 1990; Richmond 1993; Sebens 1994). These stresses reduce coral survivorship and growth and may promote macroalgal growth. Furthermore, marine pollution may increase the susceptibility of corals to disease (Brown 1988; Peters 1993).

Reef dynamics are governed by multiple causes operating on multiple scales (Grigg and Dollar 1990). For example, the recent decline in coral cover and increase in macroalgal cover on the well-studied reef at Discovery Bay, Jamaica appears to have resulted from a variety of interacting processes: Hurricane Allen (1980), which opened substratum for colonization; feeding by corallivorous snails, which increased the mortality of hurricane-fragmented corals; the *Diadema* dieoff, which reduced herbivory drastically; and a history of overfishing by humans, which had previously removed herbivorous reef fishes (Hughes et al. 1987; Knowlton et al. 1990; Hughes 1994). Likewise, the catastrophic effects of the 1982-83 El Niño event on eastern Pacific coral reefs were due to the interaction of physical changes and a variety of biological processes (Glynn 1990).

One of the greatest challenges to ecology is determining the relative importance of numerous causes to ecosystem structure and function (Quinn and Dunham 1983). Yet, even choosing the correct range of scales on which to understand coral reef dynamics seems a forbidding task. It is becoming increasingly clear that quadrat-scale observations (square meters or smaller) have less explanatory power than larger-scale observations (Jackson 1991). Those larger scales range from the landscape within a reef (hundreds of m² to hectares), to the area encompassing multiple reefs within a locality (hundreds of km²), to an entire region such as the Caribbean.

The need for large-scale and long-term (>5 yr) monitoring programs for coral reefs

has been emphasized over the past few years (Rogers 1988, 1990; Ogden and Wicklund 1988; D'Elia et al. 1991; Jackson 1991; Ray and Grassle 1991; Smith and Buddemeier 1992; Bythell et al. 1993; Glynn 1993; Hughes 1994), although less attention has been given to the hypotheses that should be driving the research (Hughes 1992). Few such studies have been undertaken, in part because hypotheses are so difficult to formulate and test (Hughes 1992). Such hypotheses might, for example, include an inverse relationship between the degree of disturbance and coral cover; the intermediate disturbance hypothesis (Connell 1978; Rogers 1993), which postulates that coral diversity should be highest at intermediate frequencies and intensities of disturbance; and complex relationships between herbivory, nutrients, coral and algal cover, and diversity (Littler et al. 1991; Knowlton 1992; Hughes 1994). However, with so many possible causes of community change, and with the ecological implications of each putative cause unknown, highly variable and/or controversial, it is hard to know which variables to monitor in order to test hypotheses adequately.

This paper outlines procedures for comparing coral reef community structure and disturbance regimes across space and through time, with a view toward eventually testing hypotheses such as those listed above. The methods were developed during continuing studies of four western Atlantic coral reefs: Carrie Bow Cay, on the Belizean Barrier Reef; Discovery Bay, Jamaica; and Conch and Carysfort Reefs, off Key Largo, Florida. Since funding for monitoring programs is difficult to obtain and since field time is usually limited, our goal was to create an accurate, relatively inexpensive, "rapid assessment" sampling program for Caribbean coral reefs, with sufficient statistical power to detect biologically meaningful differences. Video transects are used to describe the sessile biota, and fish and echinoid censuses provide estimates of the intensity of herbivory. Since the importance of each type of reef disturbance has not been conclusively demonstrated, we advocate combining them in a single disturbance index, topographic complexity. Topographic complexity reflects the total disturbance regime in at least some reef habitats, integrated on a time scale of years.

METHODS

1. Sampling Design

Any study that compares reefs must be standardized with respect to habitat. The design described here is for a common Caribbean fore-reef habitat: the spur-and-groove habitat of windward-facing reefs at 12-15 m depth. Ten 25-m long transects are sampled at each site. The transects are placed along the central axes of replicate, haphazardly chosen spurs. Sandy areas and the edges of the spurs are avoided. In the present study, each transect was placed on a different spur, except at Discovery Bay. A number of spurs at Discovery Bay were >25 m wide; 2 transects, spaced >10 m apart, were surveyed on some of those wide spurs. It is important to choose sites that will accommodate at least 25-30 transects, so that the transects sampled during different visits to a site will not be identical.

The use of band transects is dictated in this particular application by the elongate shape of the spurs, and in general by the desire to encompass as great an area as possible in each sample. The 25-m transect length was chosen to be as great as possible while still restricted to a narrow depth and habitat range on the spurs. A 25-m transect, when extended over a 3-m depth range, requires spurs that slope at angles of no more than 7°.

For each transect, scuba divers unreel a 25-m fiberglass surveyor's tape, laying it taught along the center of the spur. The tape is left undisturbed for 3-5 min, at which point the divers commence surveys of the mobile fauna.

2. Fish and Echinoid Censuses

Coral reef researchers have yet to agree on a single, reliable method for quantifying the activity of herbivores (Steneck 1983). The simplest measure is the abundance of herbivores, which correlates with their impact among habitats within reefs (Hay and Goertemiller 1983; Lewis and Wainwright 1985). Therefore, before the fish are disturbed further, a visual census is conducted along each transect. A diver swims along the tape at a standard slow speed, recording the number of parrotfish (Labridae, formerly Scaridae) and surgeonfish (Acanthuridae) within a visually-estimated 2 m on either side. The fishes are classified as small (≤ 10 cm Standard Length), medium (10-25 cm), or large (>25 cm). The small transect width minimizes the underestimate of true fish population density inherent in the transect method (Sale and Sharp 1983). Divers then carefully explore the 100 m^2 area, recording the number and species identities of damselfish (Pomacentridae), as well as censusing *Diadema* and other regular echinoid species. The echinoids can be extracted from their shelters and measured (test diameter) with calipers. Size distributions of echinoid species can then be used to estimate their impacts on algal assemblages (Levitan 1988). It should be noted that daytime censuses underestimate echinoid densities; more accurate estimates can be obtained by censusing at night (e.g., Carpenter 1981, 1986).

An alternative approach is to examine the *process* of herbivory by counting the number of bites that parrotfish and surgeonfish take per unit time from small areas of algal turf (Steneck 1983). This technique is more time-consuming than counting fish. A more serious concern is that some habitats on some reefs are currently so overgrown with fleshy macroalgae that finding even a square meter of algal turf would be problematic. For example, at Discovery Bay in 1992 macroalgal cover was >90 %, coral cover was <3 %, and the cover of algal turfs, crustose coralline algae, and bare space combined was <6 % at 15 m depth (as assessed by the video technique described below; Table 1, p. 11). Large differences in the availability of algal turfs could lead to differences in fish foraging behavior, independent of fish abundance. It is important to recognize that both the abundance and bite frequency methods yield short-term "snapshots" of herbivory, which may not adequately reflect longer-term variability.

3. Percent Cover and Scleractinian Diversity

Photography provides the only practical means of sampling large areas underwater,

given the time constraints of scuba diving. Littler (1980) discussed the advantages of photography over recording data *in situ*. High-resolution video technology makes the approach all the more attractive because videotaping is easier and less time-consuming than still photography. While still photographs provide better resolution than videotapes, the resolution of videotapes is adequate for work of the type described here. Videotapes enable the investigator to cover a far greater area per unit sampling effort. In addition, video exposures are automatic, continuous and do not require developing.

In this method, a diver swims slowly along the transect, videotaping a 40-cm wide swath of reef from a height of approximately 40 cm, using a high-resolution (Hi-8) video camera in an underwater housing, fitted with a wide-angle lens. A 15-cm gray plastic bar is attached to a rod that projects forward from the video housing. The bar, which is held at the level of the substratum during taping, provides scale in the videotaped images and also ensures that the camera is held a constant distance from the bottom.

Individual video frames are displayed on a high-resolution monitor in the laboratory. A clear plastic sheet with 10 random dots is laid over the monitor screen, and the sessile organisms underlying the dots are recorded (Sebens and Johnson 1991; see **Sample Sizes**, p. 8, for number of dots per frame). The videotape is then advanced to a new, non-overlapping position. Each 25-m transect yields 50 video "quadrats", for a total of 500 points per transect. The point count data are used to calculate percent cover and the Shannon-Wiener diversity index, *H'*, for each transect. Since this and similar sampling methods tend to be biased against the inclusion of rare species, presence-absence data are also gathered for coral species by viewing the entire videotape of each transect (Chiappone and Sullivan 1991).

By holding the camera perpendicular to the substratum, swimming slowly along the transect, and using a pair of video lights (50 or 100 watts each), it is possible to produce clear stop-action images. Corals, sponges, and some gorgonians and macroalgae can be identified to species, down to a diameter of approximately 5 cm. A drawback of this method is the difficulty of distinguishing fine algal turfs, crustose coralline algae and bare space from the tapes; these are lumped into a single category, which can be resolved by closeup, still photography if desired.

Ecologists have devoted a great deal of effort to developing and comparing methods for quantifying coral reef community structure, with variable results (e.g., Loya 1978; Dodge et al. 1982; Ohlhorst et al. 1988; Chiappone and Sullivan 1991; Porter and Meier 1992). Porter and Meier (1992) examined some of the biases introduced to surface area estimates by photographic and video techniques. Such errors include non-orthographic projection, in which coral heads that stick up above the surrounding substratum are closer to the camera lens and therefore artificially enlarged, and parallax. These problems are difficult to correct (Porter and Meier 1992).

Whorff and Griffing (1992) found that point counts from video frames overestimated

the percent cover of intertidal barnacles and bivalves, compared to computer image-processing of the video frames. More dots per frame yielded better percent cover estimates. On the other hand, Foster et al. (1991) concluded that point counts from photographs underestimated cover in multilayered, temperate subtidal assemblages, compared to point counts done in the field. The point count method is not as accurate as planimetry of the individual colonies in each frame or fully-automated image processing. However, planimetry is so time-consuming as to be impractical. Image processing is also difficult at present because subtle color and pattern differences must be detected; most corals and algae are quite similar in color, and reasonably-priced, hand-held video lights provide limited color saturation. The point count method is capable of detecting significant among-site differences in percent coral cover and diversity (see Sample Sizes, p. 8). Video has its problems and biases like other techniques, but it remains a simple, cost-effective comparative method.

Another concern is the seasonality of macroalgal growth (Carpenter 1981; Hughes et al. 1987). Seasonal changes within a site could change estimates of coral cover, as more or less living coral is obscured by the algae (J. C. Lang, personal communication). The constraints of time, logistics and funding do not always permit the investigator to standardize sampling by season, and the error in coral cover estimates caused by seasonal variations in algal growth are unknown. In the present study, Carrie Bow Cay was sampled in the late spring, Discovery Bay in the winter, and Conch and Carysfort Reefs in the fall of 1992. If algal growth is maximal in the summer and algal destruction by storms is maximal in the winter, then, all other things being equal, macroalgal cover, and the error in coral cover estimates due to macroalgal cover, should have been greatest in Florida (after the summer's algal growth), intermediate at Carrie Bow Cay (before the summer's growth), and least at Discovery Bay (during the stormy season). In fact, macroalgal cover was highest at Discovery Bay, intermediate at Conch Reef and Carrie Bow Cay, and lowest at Carysfort Reef (Table 1, p. 11). In this study, differences in macroalgal cover among sites apparently outweighed any error in coral cover estimates associated with seasonal differences.

The species diversity of scleractinian corals is evaluated as species richness, S, and as the Shannon-Wiener index, H'. S is measured for each site as the asymptote of the rarefaction curve (cumulative species numbers plotted against number of transects videotaped). Species richness is not calculated for each transect since reefs with lower coral cover are expected to have lower S per transect simply because fewer colonies are sampled (Magurran 1988). The Shannon-Wiener index is calculated for each transect as $H'=-\Sigma(p_i[\ln p_i])$, where p_i is the proportion of the ith species in the sample. Vast size differences among coral colonies and colony fragmentation and fusion obscure the meaning of H' calculated from numbers of "individuals". Unless the investigator is prepared to establish the genetic identity of all ramets, we recommend H' indices based on areal coverage for a general characterization of reef communities.

4. An Integrated Measure of Disturbance

Historical records of disturbance do not exist for most coral reefs. Even where such records are available, there is no obvious way to sum the different disturbances to reflect the total disturbance regime; one cannot simply score three disturbance points for a hurricane and two for a bleaching event. We present topographic complexity as a technique for measuring disturbance, along with its rationale and sources of error, so that the individual investigator can decide whether or not it will be useful.

Topographic complexity is measured by carefully conforming a 5-m length of fine brass chain to the substratum adjacent to the central part of each transect tape. The chain is conformed to the finest topographic features that the 17-mm links permit; it is carefully inserted into small cavities and into the spaces within thickets of foliose and branching corals. The 5-m chain length was chosen so that the procedure could be completed in a reasonable length of time (10-15 min); the chain must be carefully straightened before it is conformed to the substratum. A complexity index, C, is calculated as $C=1-d/l$, where d is the horizontal distance covered by the conformed chain (measured against the transect tape) and l is its length when fully extended (e.g., Risk 1972; Rogers et al. 1982; Aronson and Harms 1985; Hubbard et al. 1990; Connell and Jones 1991).

Disturbances that lead to the partial or complete mortality of coral colonies decrease this topographic complexity. Hurricanes decrease complexity directly by toppling branching and head corals (e.g., Rogers et al. 1982; Kaufman 1983), although this is not true in all reef habitats (Rogers et al. 1991; see below). In addition, once dead coral skeletons are exposed by a disturbance of any sort, they are colonized by bioeroders, including bivalves, sponges, sipunculans, polychaetes and echinoids, which break down the reef framework (Hutchings 1986). Disturbances that cause partial to complete mortality of coral colonies, provide fresh substratum for bioeroders, and in fact lead to increased bioerosion rates include hurricanes (Moran and Reaka-Kudla 1991), El Niño-induced coral bleaching (Glynn 1990) and damselfish territoriality (Kaufman 1977), although damselfish can have a negative effect on bioerosion by excluding echinoids from their territories (Eakin 1988; Glynn 1990). Naturally and artificially high levels of nutrients on reefs also increase bioerosion rates (Highsmith 1980; Tomascik and Sander 1987; Hallock 1988). Overfishing off the coast of Kenya increased bioerosion and decreased topographic complexity as well, by releasing burrowing echinoids from predation (McClanahan and Shafir 1990). Furthermore, coralline algae, which are important in cementing the reef framework (and thus important in maintaining topographic complexity), are suppressed by macroalgae, which are promoted by nutrient input and the removal of herbivores (Littler and Littler 1985; Lewis 1986; Carpenter 1990). Coral growth, by contrast, generally increases topographic complexity at the scale under consideration (Dahl 1976). To a first approximation *in certain situations*, the topographic complexity index should be inversely related to total disturbance, with lower values indicating flatter terrain and suggesting more frequent, more recent and/or more intense disturbance.

No measure of disturbance is free of bias, including topographic complexity. One source of error is that coral mortality does not lead to the immediate loss of structure because bioerosion takes time. In addition to this time lag, the relationship between coral mortality and physical complexity is not always direct. Some coral species can survive breakage in storms and even reproduce asexually as a result (*Acropora* spp.: Highsmith et al. 1980; Tunnicliffe 1981); low complexity accompanied by high coral cover is a possible consequence. Conversely, bioerosion following partial or complete mortality of massive coral heads could initially increase complexity rather than decrease it.

While these problems introduce error to estimates of disturbance, that error should be minimal in the spur-and-groove down to 15 m depth. Throughout Florida and the Caribbean, those habitats are now or were formerly (before disturbance) dominated by branching or other delicate coral species, including *Acropora cervicornis* (Belize, Jamaica, the Florida Keys and many other localities), branching *Porites* spp. (some reefs in St. Croix, U. S. Virgin Islands), and *Agaricia tenuifolia* (Belize). For all of these corals, complexity in their habitats should decline rapidly following mortality. Topographic complexity would not be as useful an indicator of disturbance in certain other reef habitats, such as shallow-water hardground areas, which are characterized by isolated head corals on limestone pavements. Similarly, deep-reef areas dominated by corals with a plating morphology could have high coral cover but low topographic complexity.

Another potential complication in separating disturbance and herbivory effects is an observed positive correlation of fish abundance and topographic complexity. Herbivorous (and other) fish avoid low-relief areas, including those which have been disturbed (Hay and Goertemiller 1983; Kaufman 1983). Areas with fewer herbivores may experience increased algal cover, decreased coral cover, and increased bioerosion, leading to even flatter topography (reviewed in Hutchings 1986). This feedback loop does not appear to be a problem in the reefs studied (see Sample Sizes).

SAMPLE SIZES

The techniques outlined above are intended for testing hypotheses on large scales, ranging from a landscape scale (among spurs in the spur-and-groove habitat within reefs), to a subregional scale (among reefs within an area such as the Florida Keys), to a regional scale (among reefs throughout the Caribbean). In order to determine the appropriate sample sizes for statistical comparisons among reefs, preliminary surveys were conducted during 1992-93 in the spur-and-groove habitats of the four sites mentioned in the **Introduction**. Ten transects were completed at each site, and an additional 10 transects were videotaped only (see comments below on species richness). Two investigators can sample a site in 3-4 days, assuming 2-3 "full" transects per dive and 2 dives per day. Where funding and equipment are available, the use of nitrox diving techniques increases bottom time substantially, increasing the number of transects that can be completed per dive.