Section 2: Examples of Monitoring Methods

Coral reef monitoring methods can generally be divided into techniques that measure benthic organisms (i.e., coral substrate, macroinvertebrates, or keystone species) and methods that estimate populations of mobile species, commonly reef fish. Low-tech methods for benthos include quadrat, line intercept, point-quarter and habitat heterogeneity sampling to assess coral cover and community structure over time. Photography and videography are increasingly popular since they reduce in-water time and can be conducted with minimal training but there are limitations associated with each method. A variety of reef fish census methods (i.e., belt transects, stationary plots, indicator species) are available depending on the level of expertise of the divers and questions being addressed by the monitoring program.

Chapter 6:

Sampling May be Haphazardous to Your Reef Monitoring Program

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Abstract

Because of the difficulty of collecting data underwater and the time and depth constraints imposed by scuba diving, most monitoring programs designed to assess coral cover or to document changes in coral cover on reefs over time have been lacking in spatial scale and statistical rigor. Although reef scientists understand the importance of random sampling, few if any monitoring programs have sampling designs based on random selection of study sites. This paper describes a statistically rigorous approach to sampling of coral cover and cover by other organisms along random transects. Also, a comparison of results from videotaping of transects to data from the more widely-used linear chain transect method indicated very similar values for total living coral and for the dominant coral species, but higher values for macroalgae and gorgonians with the video technique.

Keywords

reef, monitoring, coral, video, random

Introduction

Recent assessments of the status of coral reefs worldwide have painted a grim picture of reef degradation. By one estimate, 10% of the reefs have been destroyed, 60% will be lost within the next 40 years, and only 30% of the reefs can be considered stable (Wilkinson 1992). Some scientists argue that existing data are inadequate to draw these far-reaching conclusions, pointing out that most of the world's reefs have not been evaluated. There is not even general agreement about the areal extent of these marine ecosystems —a recent estimate of 225,000 km2 (Spalding and Grenfell 1997) compares to one widely cited previous estimate of 600,000 km2 (Smith 1978). Any generalization about the current status of or trends in reef systems worldwide can only be as good as the data upon which it is based, and high quality data from a sufficient number of widely distributed sites are not available.

Reef scientists recognize the need to increase the spatial and temporal scale of their examinations and to use more rigorous statistical approaches to address these pressing concerns about reef change on a global basis. Interest in long-term monitoring of reefs has grown in recent years as a result of several international symposia and initiatives, which have called for greater understanding of coral reef ecosystems, but few long-term data sets, exist for these critical ecosystems. In a recent review of quantitative studies of coral abundance, Dr. Joseph Connell was able to find only 23 studies of four or more years in duration, and data were available for only three locations from the Eastern Caribbean: the US Virgin Islands (including our data from reefs off St. John), the Netherlands Antilles, and Barbados (Connell 1997). Most of these studies were conducted on a very small spatial scale. Whether our intention is to generalize about the current status of coral reefs (specifically, the amount of living coral) on a local, regional or global basis or to determine if and how they are changing over time, we must be certain that the data we are collecting are **representative of the areas about which we are making conclusions.** We also have to avoid inappropriate extrapolation from small areas and small data sets to larger geographic areas.

Although numerous manuals on coral reef monitoring recommend random selection of study sites (or quadrats, transects, etc.), none to date addresses exactly how these sites will be selected (English et al. 1994; Oxley 1994; Rogers et al. 1994). Few if any monitoring programs have sampling designs based on **random** selection of sites within reefs chosen for study. As Roger Green notes, " Putting samples in "representative" or "typical" places is not random sampling" (Green 1979). Underwater research does not lend itself easily to random sampling because it is very difficult for divers to locate the exact positions of (sometimes widely-distributed) transects determined before they enter the water. Because of logistical, financial, and time constraints, most scientists have used **haphazard** sampling in which the diver surveys a portion of the reef, e.g., a transect, and then swims a random number of fin kicks or meters away to the subsequent sampling site, and so forth. In this case, the location of each transect is dependent on the one which precedes it, i.e., the sampling units are not independent of each other. In contrast, truly random sampling ensures that each transect (sampling unit) has an equal probability of being selected for examination. If the sample size (i.e., number of transects) is large enough, haphazard transects may give results (e.g., coral cover values) which are very similar to those from random transects, but it is not strictly appropriate to analyze the data from the haphazard transects using statistical analyses which require independent samples. The inherent variability (patchiness) in coral reef structure presents real challenges for sampling of these ecosystems. How can we be sure that our sampling is random and that it meets the criteria for rigorous statistical analysis? How important is it that we do this?

The primary purpose of this paper is to propose a statistically rigorous approach to sampling cover of reef organisms and substrate along transects, with emphasis on living coral and algae. It is an attempt to

address the shortcomings of previous studies, which have lacked sufficient statistical rigor. Although we are just now starting to test this approach on the reefs we study, we feel it warrants consideration by others, especially those who are just setting up reef monitoring programs. It involves random selection of locations for transects which can then be surveyed with a variety of different methods. We have decided to use a video method primarily, with a digital video camera in an underwater housing to provide data on cover by reef organisms. In addition, we briefly compare the video method to the more widely used linear chain transect method and discuss some of the disadvantages and advantages of each. Specifically, our purpose has been to develop a statistically defensible protocol for monitoring coral cover on reefs around St. John, US Virgin Islands. We want to test the following null hypothesis: H0 = There is no change in total hard coral cover within the lower forereef zone on selected reefs around St. John over time. We want to determine the magnitude of differences in mean coral cover that can be detected between years at the same study site, given samples of fixed size and a prescribed level of statistical significance. (The details of the statistical treatment of the data collected with this method will not be discussed here.)

Several parameters can be used to describe the structure of a particular reef, including coral cover, density of colonies, condition of colonies, and size frequency distributions. Although all of these are useful, we have decided to emphasize monitoring of coral cover (as well as algal cover) for several reasons. Most of the stresses that affect corals around St. John cause loss of coral tissue. If the corals fail to recover, the affected areas will be rapidly colonized by algae. Major stresses therefore will result in a decrease in the amount of "cover" of living coral on the reef and, usually, an increase in the amount of algae. These changes represent a significant shift from framework-building organisms to organisms that do not contribute to the reef's structure.

Because this protocol is being developed for the National Park Service (which administers Virgin Islands National Park on St. John) we are especially interested in being able to document the effects of anthropogenic (i.e., more "manageable") stresses. However, St. John has experienced several severe hurricanes in the last 2 decades, and these storms (along with several coral diseases) have reduced coral cover on the reefs, rendering them more susceptible to detrimental human activities (e.g., Rogers et al.1997). Monitoring should provide information on the responses of the reefs to a combination of natural and human stresses and should help to differentiate natural rates of change from those induced by humans.

Video Method

Rationale for using the video method

Given the complexity of coral reefs, their 3-dimensional relief or topography, and high diversity of organisms, a combination of photographic and non-photographic methods and different sampling approaches are required to assess changes in their structure over time (see Rogers et al. 1994). The method we used in the past was linear chain transects at fixed transects. We have now decided to use underwater video method for monitoring of reefs around St. John as our primary method. This method is based on procedures described in detail in Aronson et al. (1994) and Wheaton et al. (1996). Benthic cover is calculated from unique digital images of transects using acetate overlays containing random points or using an image analysis program.

Because SCUBA diving imposes severe depth and time constraints on scientists attempting to collect data, most reef monitoring studies have been based on very small areas. Use of video cameras has given us the opportunity to efficiently collect larger amounts of certain types of data, giving increased return

for the time spent underwater (see Carleton and Done 1995). For example, it takes less than 15 minutes to record a video transect 20 m long but sometimes over 4 hours to collect information along the same transect using the linear chain transect method. Although the chain transect method provides data on the 3-dimensionality of the reef surface and on organisms which are not visible from a vantage point directly above the reef (the "planar" view recorded with video), it is not suitable for collecting data over a large spatial scale. The video method requires a skilled diver but no expertise in identification of reef organisms in the field, the basis of the chain transect method. In other words, it can be used by non-scientists. The video method can provide a substantial amount of useful information while minimizing a diver's time in the water, and unlike most other methods, provides an archivable, visual record of the reef. Video transect images can be stored on write-to CD or DVD. It is now possible to use image analysis (and associated point count software, such as that developed by Dr. Phil Dustan's group at the College of Charleston, South Carolina) to store the identification of the substrate associated with each pixel. Archived images can be sent to other researchers, and the same image and pixels can be identified for quality control.

Disadvantages of the video method

In spite of its numerous advantages, the video method has its limitations. Changes in the amount of living coral recorded with still photography or video document changes in the amount of living coral tissue visible in planar (2-dimensional) view. This technique will under-represent colonies with a more vertical morphology (e.g., pillar coral) and is more suitable for monitoring reef zones with species exhibiting encrusting and hemispherical morphologies. The video method is not appropriate for monitoring recruitment of hard corals because resolution is not adequate to provide images of recruits less than about 4 cm in diameter. Also, sometimes it is very difficult to differentiate macroscopic algae ("macroalgae") from algal "turf" species because of the planar perspective. In areas with high sedimentation rates, sediment particles may obscure the underlying algae resulting in underestimates of the algal cover. These are important considerations given that increases in macroalgae are considered one of the clearest indications of severe stress to reefs. Image analysis takes a very considerable amount of experience, expertise, and time (an estimated 2 or more hours per 20 m transect) whereas chain transect data can be processed relatively quickly.

Sampling methods and designs

Here we describe the approach we intend to take on reefs off St. John. At each of our study sites, we plan to define a statistical population (in this case, a population of transects) composed of spatial units of sampling that are limited to locations within a reef zone. We define the population to be sampled at each reef by superimposing a uniformly-spaced (virtual) grid of locations based on 10-m intervals between all locations. The population therefore comprises a finite number of locations on the reef that must be identified using Global Positioning System (GPS) technology or an underwater surveying system. To ensure that sample units in the population are non-overlapping, each unit is defined as a 10-m transect of fixed width that begins at one of the locations on the uniformly-spaced grid and is oriented in the same direction as every other transect. (It is generally best to sample along depth contours because depth exerts a major control on reef development. Changes in cover and species composition often occur with changes in depth.) Although compass orientation remains the same for all transects within a population, it may differ among study sites. For example, an exception to common compass orientation is needed where spur-and-groove reef formations can result in unacceptably large differences in depth of a single transect. At such locations, it is essential to develop an unambiguous set of instructions that allows divers to select compass orientation during the survey without producing overlapping transects or in-

cluding sand channels in the videotransects. (For example, one rule might be that each diver ensures that there is 10 m of hard bottom along a depth contour, and if this is not the case, an alternate site must be selected.) These "rules" must be sufficient to ensure that all divers will make the same decisions.

Implementation of the sampling design

Here we describe in greater detail how we intend to survey one of our study sites, Newfound Bay, in an attempt to clarify several of the points raised above. The reef in Newfound Bay, on the north side of St. John, has a well defined and extensive lower forereef zone located seaward of the reef crest and the upper forereef zone. The lower forereef slopes gradually from a depth of about 15' (4.6m) to a depth of 30' (9.1m), where the slope increases abruptly until a flat, sandy plain is reached in about 45-50' (13.7-15.2m) of water. The lower forereef zone in Newfound Bay actually comprises two distinct areas (eastern and western) that are separated by a sand channel. We are proposing to sample the western forereef zone.

The western forereef is about 200 m (= distance alongshore) x 90 m (= distance offshore). We plan to begin by defining the boundaries of the reef zone we want to sample. We then define the population (of transects) by superimposing a uniformly-spaced grid of locations based on 10-m intervals between all locations. A 10-m transect is associated with each location. All transects will be oriented parallel to shore and point in the same direction to ensure that they are non-overlapping. This also limits the range of depths that may be encountered within a single transect. In a perfectly rectangular reef we would expect a statistical population of about 180 transects (=(200/10)*(90/10+1)) to fit within the perimeter; however, the actual population size of the western forereef will obviously depend on its shape.

Prior to the actual survey of the western forereef, a pilot survey will be used to determine whether the population should be stratified by depth intervals. In the pilot survey duplicate 10-m transects will be video-recorded at depths of 15' (4.6m), 25 (7.6m), and 35' (10.7m). Transect locations will be selected haphazardly, attempting to include portions of the forereef that are representative. Based on the results of the pilot survey, the population of transects (for the entire reef zone or for the shallow and deeper zones) will be identified. Randomly selected transects will be sampled, analyzed for substrate composition (cover), and analyzed statistically.

Once the results of the pilot survey have been analyzed, we will collect videorecordings of a random sample of n transects taken from the population of N transects as follows. In advance of the survey we know the spatial coordinates, say (x_i, y_i), for each of the transects ($i = 1, ..., N$) relative to one another and within the limitations of the GPS (or other underwater surveying system). By randomly permuting the sequence of integers $(1, 2, ..., N)$ and selecting the first n elements of the permutation, we obtain a simple random sample of n transects. Next, divers will be sent to make a video recording of a 10-m transect (delimited by surveying tape) at each of the n locations. The actual order in which the n randomly selected transects are videotaped will be chosen to minimize the time spent underwater and the time needed to move from one transect location to another.

The locations of the transects to be sampled are determined prior to the diver entering the water. Where differential GPS technology is available, buoys attached to weights can be placed at the latitude and longitude of the randomly selected positions which mark the beginning of each transect. This method is only feasible when sufficiently accurate GPS is available. Also, it is difficult to translate the surface coordinates accurately to the locations on the bottom. An additional significant concern is the potential for damaging reef organisms when lowering the weights attached to the buoys. Under certain circumstances (e.g., rough sea conditions) this approach will be too destructive to use. There are also (expensive!)

underwater surveying systems (e.g., Desert Star's Aqua Map system) which depend on use of SONAR signals from fixed reference units (baseline stations) and a mobile unit carried by a diver which makes location of the next transect to be sampled much more efficient. We plan to test such a system in the near future.

Comparisons of data collected with the linear chain transect and video methods

It is useful to briefly compare the results from the video method with the widely used, less expensive chain transect method. The chain transect method (described in detail Rogers et al. 1994) and videotransect methods do not sample the same portion of the reef. The former involves counting of all surfaces (organisms, substrate, etc.) directly under a line while the latter photographs a swath of the bottom about 45 cm wide. The chain method provides an estimate of the cover of the 3-dimensional reef surface while the video method gives a planar view. Here we compare the results of both of these methods to measure cover by corals and other substrate along five 20 m permanent transects at Newfound Bay (Table 1). The study site, in 25' (7.6m), is a forereef zone dominated by *Montastrea annularis*. Estimates of total coral cover and cover by the dominant coral species were surprisingly similar. Although all values for macroalgae were less than 10%, the video method consistently gave higher values. In some cases, it was difficult to distinguish macroalgae from algal turf species in the video. (This can easily be done by a diver using the chain method.) Only gorgonian holdfasts are sampled with the chain method while any of the gorgonian in the video camera's field of view can be included in the sample. Consequently, cover by gorgonians is much higher using the video method. (Note that large gorgonians can obscure organisms beneath them.) Because the chain method involves measuring surfaces which are not visible in the planar view and these surfaces typically have algal-covered dead coral, we would expect the category which we refer to as "dead coral with algae" to be higher with the chain method. However, the two methods show no significant difference between mean values for this category. Of course, this discussion is only cursory, and more work must be done to adequately characterize the differences between these methods (see CARICOMP 1997; Green and Smith 1997). Reefs dominated by branching coral species would probably give somewhat different results.

Conclusions

The selection of an appropriate sampling design and method must be driven by the specific purpose of the monitoring and the available resources. In addition, variations in the overall physical configuration (morphology) of the reef to be sampled may necessitate modifications or use of a different method altogether. The major purpose of this paper has been to propose an approach to sampling of coral cover, which is based on the videotaping of randomly selected transects. Roger Green points out, "It requires effort to ensure that sampling is random" (Green 1979). The question is, can we afford the time and expense to do random sampling? Can we afford not to? How much confidence can we have in the conclusions we draw from sampling that is haphazard? We have learned a lot about coral reefs in spite of the limitations of our previous approaches to sampling and a lack of statistical rigor. The difficulty and expense have to be weighed against the quality of the data that are collected. People who are interested in collecting data on cover by corals and other organisms for regional assessments or who are designing reef monitoring programs to track changes in coral cover over time should *consider* using the video method at randomly selected transects. Perhaps this paper will stimulate lively debate about this approach!

Table 1: Comparison of data on cover by reef organisms and substrate at Newfound Bay using chain transects and video method.

	Transect 1		Transect 2		Transect 3		Transect 4		Transect 5		Mean (SD)					
CATEGORY	chain	video	chain	video	chain	video	chain	video	chain	video	chain	video	mean ch	se	mean vid	se
CORAL	18.7	16.5	24.7	23.5	26.6	31.4	29.4	31.8	28.5	26.6	25.6 (4.3)	26.0 (6.4)	25.6	4.3	26.0	6.4
Montastrea annularis	10.7	11.2	20.1	18.7	21.1	22.9	15.5	16.8	16.0	19.1	16.7 (4.1)	17.7 (4.3)	16.7	4.1	17.7	4.3
GORGONIANS	1.3	14.3	1.3	10.1	0.8	8.2	1.2	8.1	0.3	4.6	1.0 (0.4)	9.1 (3.6)	1.0	0.4	9.1	3.6
SPONGES	2.7	2.3	3.4	2.0	2.8	0.6	2.7	1.8	3.6	2.7	3.0 (0.4)	1.9 (0.8)	3.0	0.4	1.9	0.8
ZOANTHIDS	0.1	0.2	0.7	0.2	0.0	0.0	0.0	0.0	0.6	0.0	0.3 (0.3)	0.1 (0.1)	0.3	0.3	0.1	0.1
MACROALGAE	2.7	7.4	5.1	9.5	5.3	8.8	3.4	7.3	4.8	6.1	4.3 (1.1)	7.8 (1.4)	4.3	1.1	7.8	1.4
OTHER, LIVE	0.2	0.0	0.8	0.0	0.0	0.2	0.3	0.2	0.2	0.0	0.3 (0.3)	0.1 (0.1)	0.3	0.3	0.1	0.1
DCA	46.1	48.6	34.4	49.5	37.8	38.8	39.4	43.8	35.5	49.7	38.6 (4.6)	46.1 (4.7)	38.6	4.6	46.1	4.7
SAND	18.6	7.9	21.5	4.2	24.2	11.5	16.0	5.7	16.1	8.4	19.3 (3.6)	7.5 (2.8)	19.3	3.6	7.5	2.8
UNKNOWN	na	2.7	na	1.0	na	0.4	na	1.2	na	1.9						

NOTE: "DCA" refers to dead coral with turf algae. Also includes rubble, pavement and coralline algae.
"UNKNOWN" refers to indistinguishable sample points.

Acknowledgements

My thanks to Jeff Miller, Ginger Garrison, Ellen Gorman and Rob Waara for many hours of hard work collecting reef data with video and chain transect methods. Special thanks to Jeff Miller for his meticulous work analyzing the videotransects and producing Table 1 and to Ginger Garrison for analysis of the chain transect data. I am grateful to Bob Dorazio for his substantial contributions on sampling design. Jeff Miller and Ginger Garrison made many helpful comments on the manuscript.

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Chapter 7:

Emphasis on Colony Size Distributions in Coral-Reef Surveys

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Abstract

The point-quarter method has been used by the University of Guam to assess the coral communities in Micronesia and American Samoa since the 1970s because the emphasis on size distribution data for the corals allows more forensic and predictive insights than do the extrinsic statistics of living coral cover, population density, and species diversity. As long as the objective is to document change in a coral community through time at a particular location, the standardized use of quadrats or transects is good because of the reduced variance of the extrinsic data. However, if the objectives are to deduce past events, identify the ecological processes, and to assess the condition of reefs, the intrinsic data from the point-quarter method are especially true and effective.

Keywords

benthic monitoring, coral colonies, size distribution, point-quarter method

Introduction

The University of Guam Marine Laboratory has surveyed and monitored particular transects on coral reefs since the early 1970s. The point-quarter method was selected because of the emphasis on size distributions as the measurements that give the most interpretive and predictive information. The point-quarter method also provides measurements of living coral cover and colony abundance per unit area for each species and for the entire community, as well as measurements of species diversity. However, the scale of reference by which parameters such as living coral cover, species diversity, or colony abundance indicate a healthy or degraded reef community vary with habitat, and with topographic and geographic setting. These parameters are useful in judging the condition of reef communities when provided in a time series for a particular location. But when comparing different sites without time series, a coral community with lower values in living coral cover or species diversity may be either healthier or more degraded than a community at another site with higher values of these measurements.

In recording living coral cover in Palau, for example, the communities in current-swept passes through the reef or on vertical walls are dominated by sponges, ascidians, alcyonaceans and ahermatypes such as *Tubastraea micrantha*, but there is nothing unhealthy or degraded about the sparse hermatypic coral community that shares the space. Also, a reef slope in the central Pacific, for example at Takapoto in the Tuamotus, may have only a third the number of coral species as in a comparable survey in Japanese waters because of the decrease in species diversity from west to east across the Pacific (Paulay 1997). But the less diverse reefs of Polynesia seem to be growing as vigorously and functioning as productively as the far more diverse reefs of northern Japanese waters (Birkeland 1996). These are extreme cases to make the examples clear, but the same differences in geography and habitat apply on a small scale to make comparisons among locations using living coral cover, colony abundance, or species inappropriate for judging reef degradation or health.

In contrast, size distributions of coral populations integrate the effects of past events and allow the prediction of future changes (Grigg 1975; Bak and Meesters 1998). For example, coral reefs were surveyed in American Samoa in 1985 at Aunu'u Island and at the Rainmaker Hotel. The coral populations at Aunu'u consisted almost entirely of small colonies, while those near the Rainmaker Hotel were nearly all large colonies (Fig. 1). Focusing on living coral cover could give a mistaken interpretation that the reef at the Rainmaker was in better condition than that at Aunu'u because it had 19.2% living coral cover (mean coral colony diameter = 22.4 cm, with a maximum in the transect at 277 cm), while at Aunu'u the living coral cover was an order of magnitude less (1.8%) and the colonies were all much smaller (mean coral colony diameter = 6.2 cm with a range of 1-17 cm). By observing the size distributions, however, the small coral recruits predominating the reefs at Aunu'u indicated that the reefs were recovering vigorously from a catastrophe a few years previously. This was a true interpretation. A phalanx of *Acanthaster planci* were observed devastating the rich *Acropora* reef in 1978 and following the processes of recruitment and growth over time in the coral community verified that the reef was in the process of recovering vigorously (Fig. 1). The lack of recruits at the Rainmaker indicated that the reef was dying by attrition. Although there was no striking mortality of adults as there was at Aunu'u in 1978, the reef was deteriorating because the large coral colonies were not being replaced in the community. This was probably because of pollution from the two major fish canneries about one kilometer further in the harbor. There may be an improvement in recruitment in the near future because the wastewater discharge pipe was extended further out past the Rainmaker Hotel in 1992. These interpretations and predictions were correctly made from size distributions and would probably have been incorrectly made from living coral cover and colony abundance.

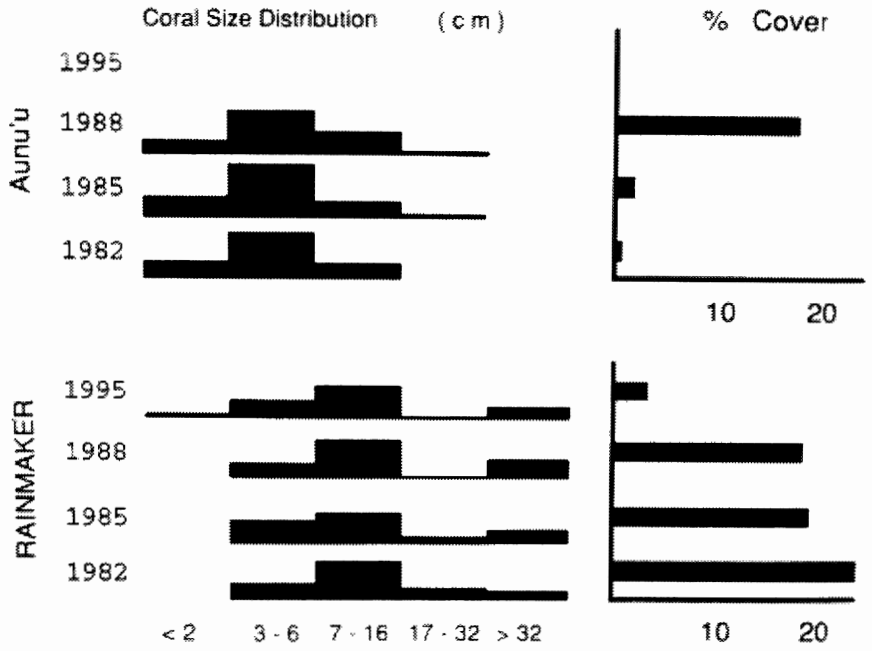

Fig. 1 - Size distribution of coral colonies and living coral cover at two sites in American Samoa.

Fagatele - Transect 2 12 m depth - July 1995	Size distribution of colonies Diameters in cm								
corals	n	$\bar{Y}$	s	w	frequency	density per m²	relative % density	% cover	relative % cover
Montipora venosa	33	4.1	1.6	1.0-8.1	0.62	5.79	0.39	0.02	0.002
Montipora grisea	11	20.2	10.8	8-49	0.38	1.93	0.13	0.47	0.06
Porites (Synarea) rus	10	12.1	1.8	2.4-30.8	0.43	1.75	0.12	0.17	0.02
Pocillopora eydouxi	4	25.9	4.6	20.8-27.4	0.19	0.70	0.05	0.78	0.10
Montipora monasteriata	3	20.2	3.9	16-23.5	0.14	0.53	0.04	0.47	0.06
Montipora verrilli	3	8.1	7.6	3.5-16.9	0.10	0.53	0.04	0.09	0.009
Galaxea fascicularis	3	6.6	0.4	6.2-7.1	0.05	0.53	0.04	0.05	0.006
Porites sp.2	3	5.5	0.6	4.9-6.0	0.10	0.53	0.04	0.03	0.004
Acropora cerealis	2	27.2	5.3	23.5-30.9	0.10	0.35	0.02	0.86	0.11
Favona sp.3	2	4.4	2.7	2.4-6.3	0.10	0.35	0.02	0.02	0.002
Favona variane	1	65	-	-	0.045	0.18	0.01	4.89	0.60
Favia pallida	2	9.4	10.0	2.3-16.5	0.10	0.35	0.02	0.10	0.01
Leptoria phrygia	1	7.5	-	-	0.05	0.18	0.01	0.06	0.007
Favites russelli	1	6	-	-	0.05	0.18	0.01	0.04	0.005
Acropora azure	1	5.9	-	-	0.05	0.18	0.01	0.04	0.005
Acropora hyacinthus	1	5	-	-	0.05	0.18	0.01	0.03	0.004
Montipora ehrenbergii	1	4.9	-	-	0.05	0.18	0.01	0.03	0.003
Porites annae	1	3	-	-	0.05	0.18	0.01	0.01	0.001
Psammocora samoensis	1	1.7	-	-	0.05	0.18	0.01	0.003	0.0004
COMMUNITY	84	10.5	11.1	-		14.7	1	8.16	1

Fig. 2 - An example of the table of information provided by QuattroPro from the distance, length x width data acquired by the point-quarter method.

The procedure for the point-quarter method is described in detail in UNESCO (1983), Brower and Zar (1984) and Birkeland and Lucas (1990). From random or haphazard points on the reef, three measurements are made of the nearest coral colony in each quadrant: distance from the random point, greatest diameter of the colony, and the diameter perpendicular to the greatest. These three data for each colony are entered into a QuattroPro program that provides a table for each location (Fig. 2), which gives for each coral species: density (abundance of colonies per unit area), relative abundance in the community, frequency (a measure of patchiness of distribution), living coral cover, relative percent coral cover of the species in the coral community, mean colony size (Y), and standard deviation (s) and range (w) in colony size. These statistics are also calculated for the coral community as a whole and are given at the bottom of the table. The information provided in the table allows for the calculation of species diversity on the basis of number of colonies or on the basis of percent living coral cover of each species.

From the simple three-part measure for each colony, nearly all of the usual statistics for coral reef community structure are provided. But the especially strong contributions of the point-quarter method are the predictive and forensic capabilities provided by the emphasis on colony size distribution. A prevalence of small colonies and broken remains of large colonies indicate a recent disturbance but a generally healthy environment for coral recruitment and growth, as at Aunu'u following an outbreak of *Acanthaster*. The prevalence of large colonies and almost no small colonies with only 20% living coral cover indicates a more insidious chronic environmental change that is detrimental to recruitment, such as the pollution and sedimentation near the Rainmaker Hotel.

A mix of large colonies providing the topographic structure of the community and small colonies indicating population turnover characterizes a healthy coral community. Grigg (1975) illustrates how the degree of mix in size distributions of corals can be used to interpret both the relative suitability and stability of habitats for coral species. Bak and Meesters (1998) explain how size-frequency data for corals can be used to determine differences between coral species in life-history characteristics of juvenile input and longevity, the size for each species at which the total colony death is replaced by partial colony mortality as the more prevalent event, and differences within species in responses to environmental differences in localities and time.

Although the characteristics of the remains of coral colonies (Fig. 3) are additional information not actually obtained as part of the core method of the point-quarter, line-intercept, transect, or quadrat methods, the emphasis on colony size distributions compels a focus on the condition of the different size classes, and poses questions on the causal mechanisms that might be by-passed when the emphasis is on extrinsic statistics.

It has been argued that because of partial mortality, fragmentation and reattachment, the size distribution of corals does not correlate with age. There are particular cases in which this can be a problem for interpretation, but in most cases the morphology of the coral can indicate its past for several years. An arborescent or corymbose coral that has reattached shows a horizontal branch or plate with a series of abruptly vertical side branches (Fig. 3a). Massive corals with subcolonies separated by limestone, because of partial mortality, often show the connection in the dead colony (Fig. 3b).

Advantages of the point-quarter method, in addition to being more predictive and interpretive, are that there are no zeroes, the stations can be spread throughout the area to be sampled, the data are independent and so there are larger degrees of freedom and proportion of independent assessments, and the data are intrinsic. Quadrats or transects can be empty, but the point-quarter method actually quantifies rarity by measuring distance to the nearest coral, no matter how far it is. While all the data in a transect or quadrat are related, allowing only $n = 1$, no matter how many corals are counted in the transect or

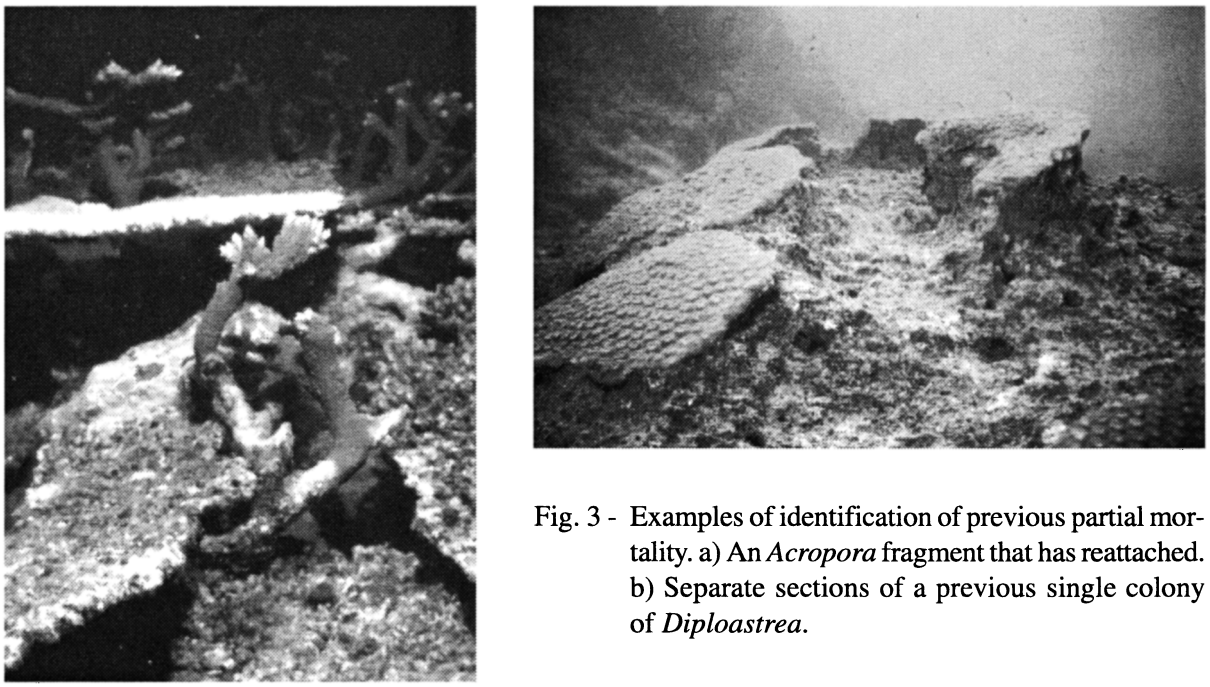

Fig. 3 - Examples of identification of previous partial mortality. a) An *Acropora* fragment that has reattached. b) Separate sections of a previous single colony of *Diploastrea*.

quadrat (Hurlbert 1984), all data are independent with the point-quarter method which gives one to two orders of magnitude more degrees of freedom in the data analysis and almost certainly a set of statistics more representative of the parameters being estimated. Furthermore, each set of four coral colonies can be taken at wider or narrower distances around a bay or along a coast, depending on the area being surveyed, so that the whole area is represented. In other words, the points or quadrats within a transect are associated with the transect while the hammer tosses for each point in a point-quarter survey can be spread out to cover the region being surveyed.

Advantages of intrinsic data

Extrinsic statistics are heuristic in that they provide a measure of abstract attributes of a "whole that is more than the sum of the parts" (e.g., community characteristics such as diversity). But information on attributes of individual species can be lost at this level. As a simplified example to illustrate this matter, consider the difference in distribution of individual die tosses and the average of two or more tosses (Fig. 4). The intrinsic probability distribution of all six sides of a die is equal at 16.7%. The probability of getting a "6" or a "1" is equal to the probability of getting a "3" or a "4". In contrast, the mean (extrinsic) statistic of any size sample of die tosses will be about "3.5". The average of die tosses does not reflect the probability distribution of individual tosses. Likewise, extrinsic statistics such as living coral cover or species diversity do not reflect the past history of the coral community as clearly as do the size distributions of individual species.

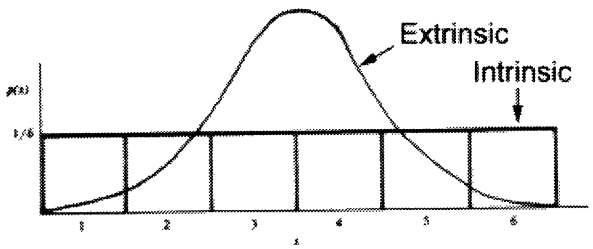

Fig. 4 - A comparison of extrinsic and intrinsic statistical views of a population of die tosses.

The most gratifying attribute of the point-quarter method is that the statistics are intrinsic, that is, the variance in the data is based on direct measures of the corals. Statistics from data taken on transects or quadrats are extrinsic, and so the variance in the data is influenced by the size of the transect or quadrat as well as community characteristics such as patterns of dispersion.

Disadvantages of the point-quarter method

A disadvantage of the point-quarter technique is that it is possible, in special cases where coral colonies overlap vertically, as when corymbose or tabletop *Acropora* are found abundantly in tiers or extend over a smaller massive or encrusting coral, for the estimate of living coral cover to be greater than 100%. Although this almost never happens, it is awkward to have over 100% cover.

A second disadvantage of the point-quarter method is that the true variance of intrinsic data is usually greater than the artificially restricted variance of extrinsic data. The variance of extrinsic data can be artificially limited by the size of the quadrat or transect. For example, if the quadrat is small relative to the abundance of coral, one might only get zeroes, whereas with the point-quarter measurements, the distance to the coral is measured no matter how great. The measurement does not artificially stop at zero, but provides a direct measure of scarcity, which increases the variance. Conversely, if corals are relatively large and very abundant compared to the size of the quadrats, then quadrats might be artificially restricted to one per quadrat or 100% cover, whereas the point-quarter method actually measures the dimensions of the coral which can provide large numbers not restricted by quadrat size. The advantage of the intrinsic data of the point-quarter is that it gives statistics that more truly estimate the population parameters, while the disadvantage is that the variances are greater.

Dangers of standardization

Standardized surveys or sampling techniques are essential for statistical tests. If testing a hypothesis, monitoring environmental quality according to standards set by law, doing comparative regional studies, or monitoring changes through time at a particular place, it is important that the survey uses standardized methods and the data are collected in a consistent manner. Each survey technique has its own biases, but as long as it is used consistently, the biases should be constant among replicates and will not confound the comparisons.

For example, Hillmann-Kitalong and Birkeland (1987) surveyed zooplankton with paired nets pushed before a boat, and with paired nets towed behind the boat. There were no significant differences in total amounts of zooplankton in pushnet and pullnet samples for dry weight of zooplankton or for total numbers of zooplankton, but there were consistent differences between the types of zooplankton that were prevalent in the two kinds of net. Motile zooplankton was consistently more prevalent in the pushnet samples and passive drifters were more prevalent in the pullnet samples (Table 1). There were generally no significant interactions in results between type of net and date or location; therefore any bias should be consistent across the sampling program. If we were testing hypotheses about the effects of pollution on zooplankton or monitoring changes in zooplankton through time, either pushnet or pullnet sampling would be satisfactory.

However, if we were estimating parameters or developing a model to understand the system or make predictions, then we would obtain different results from the two methods. The pushnet samples would lead us to deduce lower fecundity and greater survival of larvae for the fishes, while the pullnet samples would lead us to assume greater fecundity and lower larval survival in the life history of fishes. As

Table 1 Mean abundances (no. m^{-2}) and standard deviations of the eight most prevalent groups of zooplankton in pushnet and pullnet samples (n = 24 for both sets of samples). The data are from UNESCO 1983.

Taxa	Push $\bar{Y} \pm S_y$	Pull $\bar{Y} \pm S_y$
Acartia sp.	387 ± 98	184 ± 55
decapod zoeae	79 ± 29	25 ± 11
fish larvae	3.6 ± 0.74	1.3 ± 0.30
fish eggs	253 ± 61	378 ± 81
invertebrate eggs	168 ± 43	202 ± 49

Hamner and Carleton said (1979 pg. 10) "The use of imprecise but distinctive sampling techniques provides greater credence in the final conclusion than would the use of a single technique, no matter how precise the replicates." Coral reef scientists have often tried to establish their survey programs as the standard. But how can we be sure we are asking the right questions or measuring the right variables? If our goal is to understand the system or to get a true estimate of parameters, we should use a variety of techniques.

Standardized methods often define the questions we ask. Standardization can put blinders on us and prevent us from asking the right questions. For half a century or so, a major question in coral reef ecology was how so many planktivores could survive on so few plankton. Quantitative studies with drift nets indicated that drift zooplankton in waters surrounding coral reefs were not abundant enough to meet the metabolic requirements of animals feeding on them. The ledger didn't balance. Since the turn of the century, scientists defined plankton as drifters by the sampling technique, pulling a net. The standardized sampling program defined the question. Then Alldredge and King (1977) and Porter et al. (1977) invented a new sampling device to trap demersal plankton that were not drifting in, but were residing on the reef. They found that demersal plankton was a major source of food for coral reef animals that had not been previously included in models, hypotheses, or monitoring programs because the standardized towed net defined the question and did not allow consideration of demersal plankton or copepod swarms near the substratum.

To test specifically defined hypotheses, to monitor for changes in time of particular variables, or to do comparative regional surveys of definite variables, standardized sampling programs are required. However, if the objective is to determine causal relationships, developing models of reality, discovering latent resources, or estimating true parameters, then the achievement of objectives may be prevented by blind adherence to an arbitrary standardization of survey techniques.

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Chapter 8:

First Records of Coral Disease and Tumors on Hawaiian Reefs

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Abstract

A variety of methods were used to quantify the distribution and temporal progression of tissue necroses and tumors on coral colonies at three sites in Hawai'i. Distribution and abundance of diseased tissues were recorded as well as changes in overall coral and algal abundance through the analysis of permanent macrophotography stations and video transects of survey areas. The incidence and extent of necroses, as well as the amount of algal turf-covered surface area, increased with increasing colony size; the incidence of tumors on coral colonies was independent of colony size. Both coral necroses and tumors usually led to the loss of coral tissue, although recovery and regeneration sometimes occurred from the edges of healthy tissue with time.

Mean coral and algal abundance varied within and among sites, as well as through time. Coral cover declined by 10% during the seven-year period of this study (1991-1998) at Hanauma Bay, Oahu and Honolua, Maui, with a concomitant increase in turf algae. Coral cover increased during the same period at Puako, Hawai'i, while mean algal cover decreased. Incidence of diseased tissues initially declined between 1991 and 1993 censuses at Hanauma Bay and Honolua and has remained relatively constant at all sites from 1993 to 1998. These studies provide the first documentation of disease and tumors in Hawai'i.

Keywords

corals, colony size, disease, tumors, videography

Introduction

In November 1990, a complaint of "sick corals" beyond the fringing reef at Hanauma Bay was brought to the attention to University of Hawai'i scientists by an underwater photographer (D. Schricte) and the Hanauma Bay park manager (A. Hong). Initial field observations confirmed the presence of tissue lesions and tumors on numerous colonies of the dominant scleractinian coral at Hanauma Bay, *Porites lobata*. Although coral diseases have been previously reported in the Indo-Pacific (Antonius 1985), and major reef areas in Florida and the Caribbean have suffered marked losses in living coral over the last two decades as a result of disease epidemics (Gladfelter 1982; Peters and Santavy 1996), there had been no previous reports of disease in Hawaiian reef corals.

A series of studies, funded by the Hawai'i Department of Land and Natural Resources/ Division of Aquatic Resources and University of Hawai'i Sea Grant, were undertaken to ascertain the spatial distribution and temporal progression of these previously undescribed diseases and to develop a preliminary evaluation of the effects of coral disease and tumors on the overall "health" of reefs in Hawai'i.

Methods

General Study Design

Initial selection of study sites was based on areas with similarly high species diversity, range of depths and exposure, and high value as state natural resource areas: Hanauma Bay, Oahu; Honolua Bay, Maui; and Puako, on the Kona coast of the Big Island. Hanauma Bay and Honolua Bay are Marine Life Conservation Districts; Puako reef is a Hawai'i State Fisheries Management Area. Censuses have also been conducted at other reef sites on Maui (Puamana, Olowalu), Hawai'i (Kealakekua), and Molokini.

The impacts of disease and tumors on reef health were assessed at three levels:

1) Changes in overall coral abundance on reefs (benthic video transects),

2) Colony-level changes (visual surveys of distributions relative to colony size, depth, distance from shore; time-series macrophotography), and

3) Histological analysis of diseased tissue

Temporal Changes in Coral Community Structure

Site Stratification

Each of the study sites had slightly different reef community structure and topographic profile. Therefore, within each site, survey areas were stratified based on depth, exposure and reef community characteristics; positioning deliberately avoided large sandy areas and steep slopes.

Four reef areas within Hanauma Bay were surveyed on 22 January 1992; 3 March 1992; 12 July 1992; 7 October 1992; 1 January 1993; 14 April 1993; 4 May 1994; 24 May 1995; 21 May 1997; and 13 May 1998. The defined purpose of the present study was not to assess the direct impact of visitors on the shallow, nearshore reefs of Hanauma Bay, but rather to assess the impacts of disease on overall coral abundance and reef community structure. Selected survey areas were therefore sited beyond the shallow reef crest most utilized by Hanauma Bay visitors to minimize the complicating factors of potential impacts from trampling and other user contact (Fig. 1).

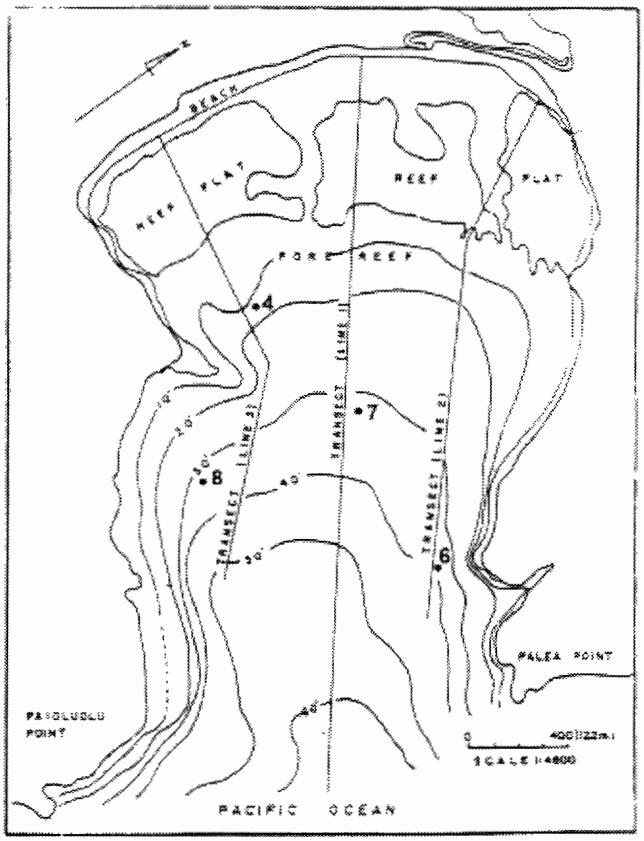


Figure 1. Map of Hanauma Bay, Oahu (from Anderson, 1978), showing study areas #H-4, 6, 7, and 8.

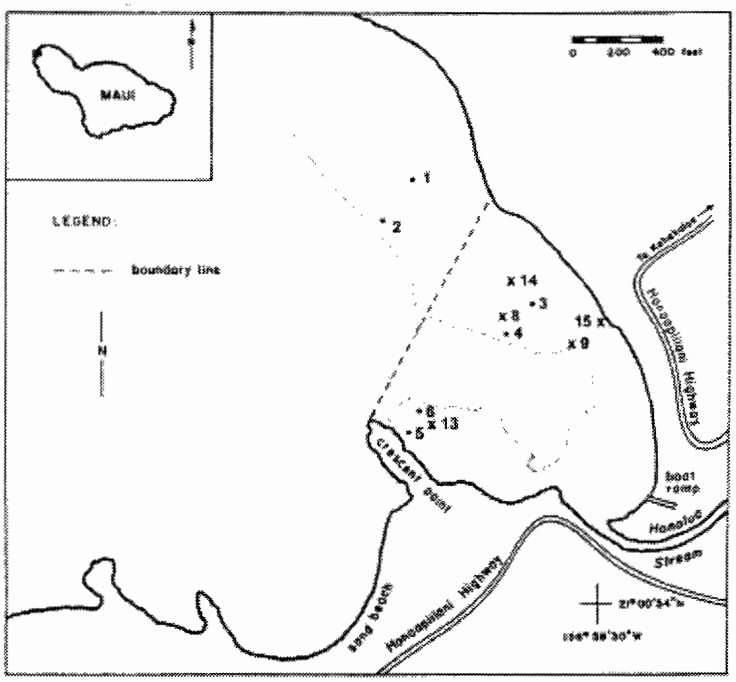


Figure 2. Map of Honolua Bay, Maui (from Torricer et al. 1979), showing study areas #M-1, 2, 3, 4, 5 and 6.

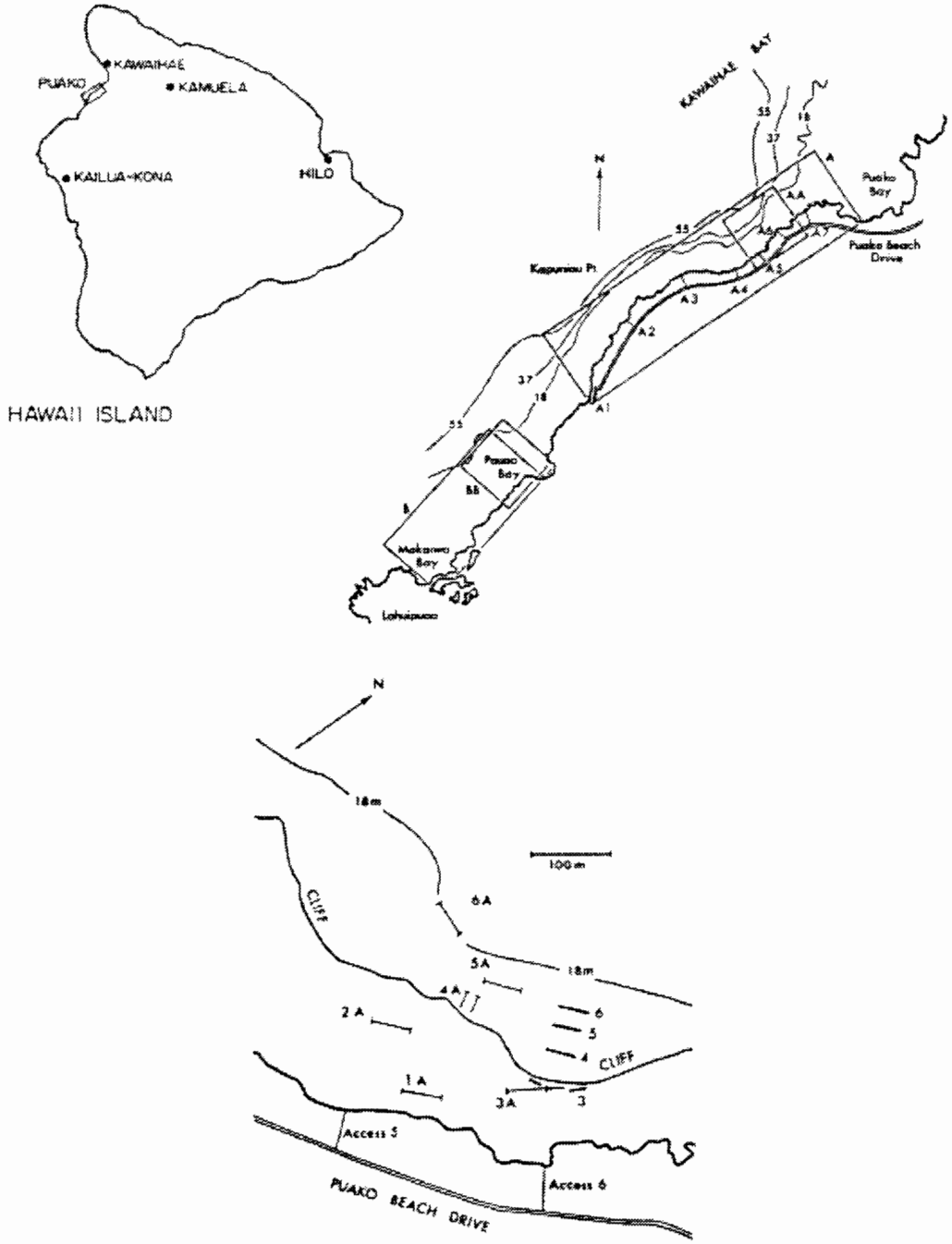


Figure 3. Map of Puako, Hawai‘i (from Hayes, et al. 1982), showing study areas #P-3, 4, 5, and 6.

Six areas were selected for monitoring at Honolua Bay (Fig. 2). The majority of coral cover at this site is distributed over a shallow fringing reef at 3 m depth that drops along a fairly steep contour to a shelf at about 6 m before sloping to the sandy bottom at the center of the bay. Therefore, three areas were located at 3 m depth (two "exposed" and one "protected") and three at 6 m (similarly, two were "exposed" and one "protected"). Surveys at the Honolua site were conducted on 2 October 1991; 16 June 1992; 30 October 1992; and 19 May 1997.

The Puako site differs from Hanauma and Honolua in being on a more open coastline. The Puako reef tract is unusual along the Kohala coast of the Big Island in having a fairly extensive subtidal reef flat shoreward of underwater sea cliffs that drop to a gently sloping zone of high coral cover. Four survey areas at this site were established parallel to the shoreline on the reef flat (2 m depth) and below the sea cliffs at 9, 11, and 13 m depth (Fig. 3). Surveys at the Puako site were conducted on 28 September 1991; 24 August 1992; 11 April 1993; 7 April 1995; 26 May 1997; and 30 May 1998.

Installation of Transects

Three parallel transects were marked within each study area (4-6 areas/site). Reconnaissance surveys identified areas predominated by corals, avoiding sand channels and vertical slopes. To randomly select the exact transect locations within each area, a lead weight was dropped blindly from the surface, marking the leeward end of the central of the three parallel transects. The center ("B") transect (10 m in length) was then deployed in a seaward direction along a pre-determined compass heading from that first random point. The "A" and "C" transects (also 10m in length) were aligned 2-4 m in either direction and parallel to the center ("B") transect. The endpoints of each transect were marked with stainless steel eyebolts securely screwed into pre-drilled holes (1/4") in the coral substratum. Eyebolts were collared with 4 x 4 cm plastic tags inscribed with the transect identification codes.

A measure of spatial heterogeneity within each area was obtained by laying a chain (each link=1.42 cm) under the middle transect line, taking care that the chain conformed to the bottom topography. The length of chain necessary to cover a horizontal distance of 10 m under the transect line was then recorded. The ratio of chain length to horizontal distance along the bottom provided an index of vertical relief and estimated spatial heterogeneity at each area (Rogers et al. 1983). Substratum type under each link of the chain was also recorded, providing an additional comparative index of benthic biota and substratum abundance.

Video Surveys

A Yashica TR-81 Hi-8 video camera in a VideoSea underwater housing (JayMar Engineering, San Pedro CA) was used to record benthic biota and substratum abundance at each census period. A diver swam slowly along the length of the transect, holding the camera vertically at a distance of 0.5 m above the transect. Stilled frames from video transects recorded from this height were approximately 20 x 30 cm in area.

Auto-focus and auto-exposure features of the camera provided high-resolution records of the substratum under the transects. The ability of the camera to maximally utilize these features was substantially enhanced by the use of a rainbow-colored nylon line rather than black or white surveyor tapes. The black or white tapes caused over- or under-exposure of the underlying reef features relative to the tapes themselves. The rainbow-colored line, marked and weighted at 10 cm intervals, also provided a color scale within each video frame.

Video Analysis

Hi-8 video records were transferred to sturdier VHS tapes prior to analysis. Original Hi-8 tapes were archived for permanent data storage (University of Hawai'i, Waikiki Aquarium).

Point locations in gridded arrays (10 points/array) were randomly generated and their locations transferred to an acetate sheet cut to fit the video monitor used for analysis of tapes. The positions of ten frames along each 10 m transect were randomly determined prior to viewing the tapes (generated from random numbers between 0-1000 cm at 10 cm increments; overlaps were discarded and regenerated).

Individual video frames were displayed on a VCR monitor. For each transect, the substratum type located under 60 points (6 sets of 10 points) in each of 10 frames was recorded. Percentage data were arcsine transformed prior to statistical analysis. Statistical analysis of results of the first census period at Hanauma Bay (with 6 transects/area) demonstrated that 30 points (3 sets of 10) in 5 frames/transect for 3 transects was sufficient to describe mean coral abundance within an acceptable level of experimental error (<5%). Increasing the number of points, frames, or transects/area did not result in a further decrease in standard error of the means. Subsequent analyses therefore consisted of 30 points x 5 frames x 3 transects/area x 4-6 areas/site resulting in 1800-2700 point-samples per site per survey.

Spatial Distribution of Diseased Corals

The numerical abundance and distribution of diseased colonies of *Porites lobata* in Hanauma Bay were assessed in three surveys on 16-17 April 1992; 23 February 1994; and 27 May 1998. Six areas were selected to represent a range of depths and distances from shore within the bay. A one-time survey of colonies outside of the bay (Paioluolu Point area, 13 m depth) was conducted on 6 March 1992.

At each area, the starting point of a 50 m linear transect was randomly determined by dropping a small lead weight from the surface. From this starting point, a 50 m transect line was deployed approximately parallel to the shoreline. Each colony of *P. lobata* intersecting the transect was measured along and perpendicular to its long axis. The relative proportions of healthy and diseased tissues (necroses and tumors) were estimated for each colony, along with the abundance of algal cover (macroalgae, turf, or coralline) and other features (bleaching, shrimp burrows, or fish bites) on the colony surface.

Temporal Progression of Disease

Diseased areas on *Porites* colonies were marked by drilling two 1/4" holes into the coral skeleton below the affected area. Expandable polypropylene anchor bolts were pushed through plastic tags (2"x2") inscribed with identifying numbers and tapped into one of the holes until flush with the colony surface. A 3/8" x 6" stainless steel peg with a square cross section was tapped into the other hole leaving 2-3 cm of the peg protruding. The top corner of each peg was filed down to indicate orientation of the camera framer. Tags and pegs fit snugly into the drilled holes and have remained in place for more than six years.

A Nikonos V camera with automatic exposure and flash was used for all macro-photography. A 3/8" square socket was affixed to the distal end of a Nikonos close-up kit frame extension bar, allowing an aligned fit with the stainless steel pegs. A label strip with corresponding identification numbers, date, location and a ruler scale was attached to the upper edge of the frame. At each survey, a diver aligned and fitted the socket on the close-up frame over the end of the stainless steel peg below each diseased

Table 1. Summary of chain transect data and indices of spatial heterogeneity measured at "B" transects for all sites.

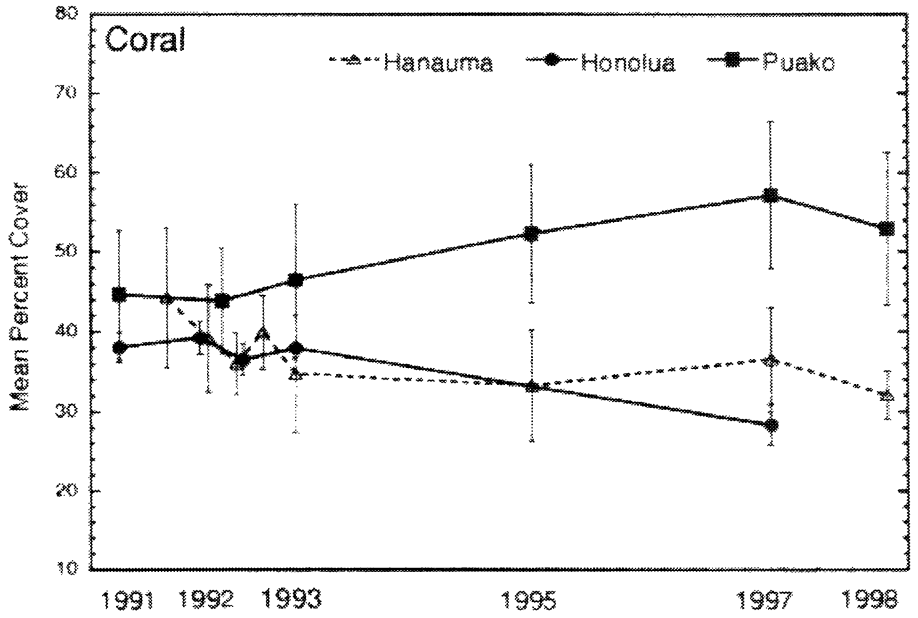
HANAUMA BAY		PL	PC	TURF	CA	PALY	PAV DUER	PAV VAR	POC MEAN	MON FLAV	MON PAT	MON VERR	MON DIL	ECHIN MATT	HET CENT	DEAD PM	SAND	TOTAL LINKS	TRANS LGTH (M)	SPATIAL HET.
SITE 4	SUM	325	30	512	15	6	18	0	0	0	0	0	0	0	2	0	0	908	10.1	1.26
	RATIO	0.36	0.03	0.56	0.02	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		721.0	
SITE 6	SUM	176	228	62	441	0	0	0	0	0	0	0	0	2	0	0	67	976	10.0	1.37
	RATIO	0.18	0.23	0.06	0.45	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07		714.0	
SITE 7	SUM	321	20	550	12	0	0	0	0	0	0	0	0	43	0	0	5	951	10.0	1.33
	RATIO	0.34	0.02	0.58	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.01		714.0	
SITE 8	SUM	136	195	355	329	0	0	0	0	0	0	0	0	0	0	0	0	1015	10.4	1.37
	RATIO	0.13	0.19	0.35	0.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		743.0	
HONOLUA																				
SITE 1	SUM	115	0	572	6	0	0	0	21	11	15	5	0	66	0	0	0	811	10.0	1.14
	RATIO	0.14	0.00	0.71	0.01	0.00	0.00	0.00	0.03	0.01	0.02	0.01	0.00	0.08	0.00	0.00	0.00		714.0	
SITE 2	SUM	89	11	426	0	0	0	1	47	0	138	40	5	4	0	40	20	821	10.0	1.15
	RATIO	0.11	0.01	0.52	0.00	0.00	0.00	0.00	0.06	0.00	0.17	0.05	0.01	0.00	0.00	0.05	0.02		714.0	
SITE 3	SUM	333	99	364	10	0	0	7	10	21	112	38	0	0	0	0	0	994	10.5	1.35
	RATIO	0.34	0.10	0.37	0.01	0.00	0.00	0.01	0.01	0.02	0.11	0.04	0.00	0.00	0.00	0.00	0.00		739.0	
SITE 4	SUM	173	312	234	119	0	1	31	0	0	3	64	0	0	0	40	81	1058	10.7	1.41
	RATIO	0.16	0.29	0.22	0.11	0.00	0.00	0.03	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.04	0.08		753.0	
SITE 5	SUM	73	9	465	25	0	0	6	0	38	75	25	0	11	0	0	75	802	10.0	1.12
	RATIO	0.09	0.01	0.58	0.03	0.00	0.00	0.01	0.00	0.05	0.09	0.03	0.00	0.01	0.00	0.00	0.09		714.0	
SITE 6	SUM	158	63	355	64	2	0	3	12	1	88	143	0	6	0	0	7	902	10.3	1.24
	RATIO	0.18	0.07	0.39	0.07	0.00	0.00	0.00	0.01	0.00	0.10	0.16	0.00	0.01	0.00	0.00	0.01		725.0	
PUAKO																				
SITE 4	SUM	295	93	206	311	0	0	0	0	0	0	3	0	2	4	0	0	914	10.3	1.26
	RATIO	0.32	0.10	0.23	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		725.0	
SITE 5	SUM	206	261	228	45	0	0	3	0	0	2	12	0	3	0	0	136	896	10.0	1.25
	RATIO	0.23	0.29	0.25	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.15		714.0	
SITE 6	SUM	155	503	68	105	0	0	2	0	0	0	8	0	3	0	0	64	908	10.0	1.27
	RATIO	0.17	0.55	0.07	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.07		714.0	

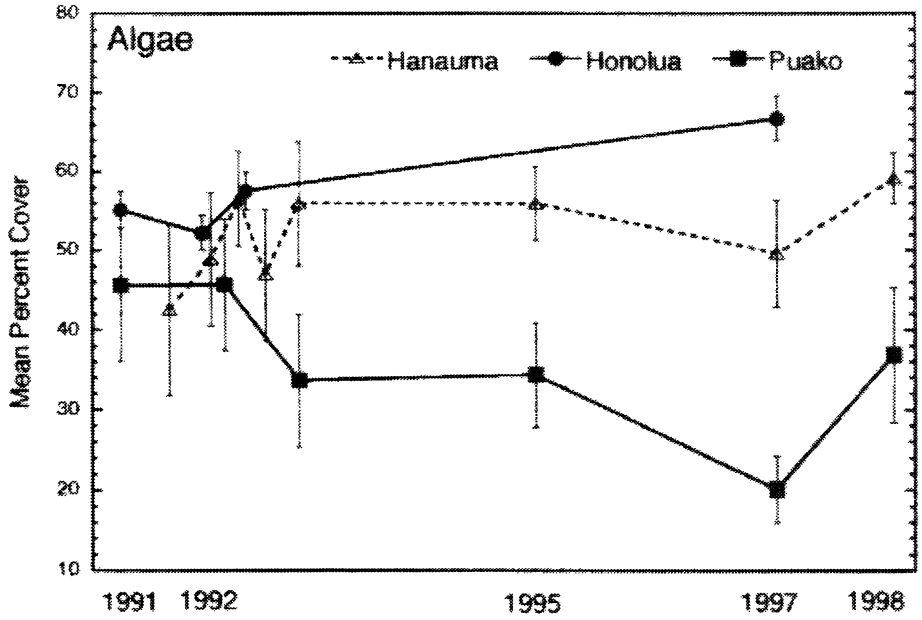
area and photographed the framed area using Ektachrome 100 slide film. Total area of photographs within the close-up frame (excluding the label strip) was 11 x 18.5 cm.

A total of thirty-six diseased colonies was photographed over a 6.5 year period at Hanauma Bay (October 1991-May 1998); 18 areas were photographed between October 1991-October 1992 at Honolua and between September 1991-April 1993 at Puako. Images from selected slides were transferred to Kodak compact disc for permanent storage and analysis.

Diagnosis and Pathology of Disease

Coral samples were collected from Hanauma Bay under permit from the State of Hawai'i, Department of Land and Natural Resources. Cores (5 cm diameter x 5 cm thick) were taken from representative diseased areas on four colonies of *Porites lobata* in 1-2.5 m depth at Hanauma Bay. Samples were placed in fresh Helly's fixative immediately upon return to the beach. After 24 h, the cores were thoroughly rinsed in running water and transferred to 70% ethanol for 72 h. Cores were then packed in ethanol-soaked diaper and shipped to the USEPA Environmental Research Laboratory at Narragansett, RI, where they were processed for histological analysis.


Coral
Hanauma
Honolua
Puako
Mean Percent Cover
80
70
60
50
40
30
20
10
1991
1992
1993
1995
1997
1998


Algae
Hanauma
Honolua
Puako
Mean Percent Cover
80
70
60
50
40
30
20
10
1991
1992
1995
1997
1998

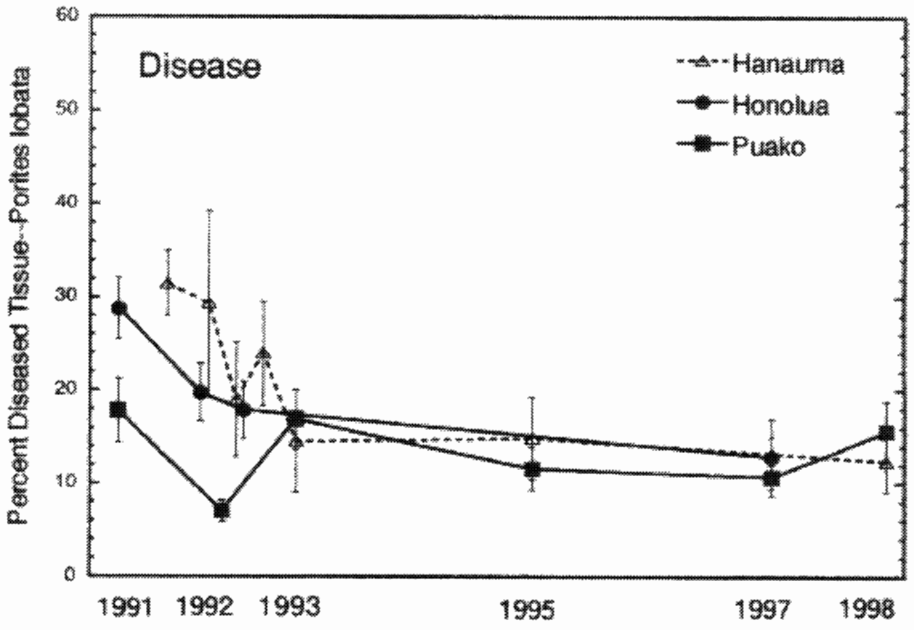
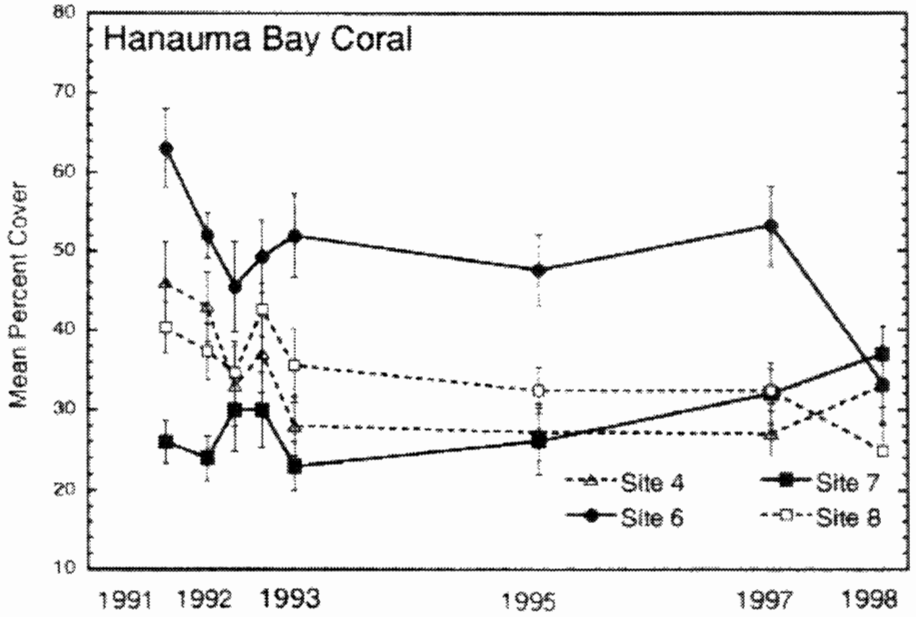
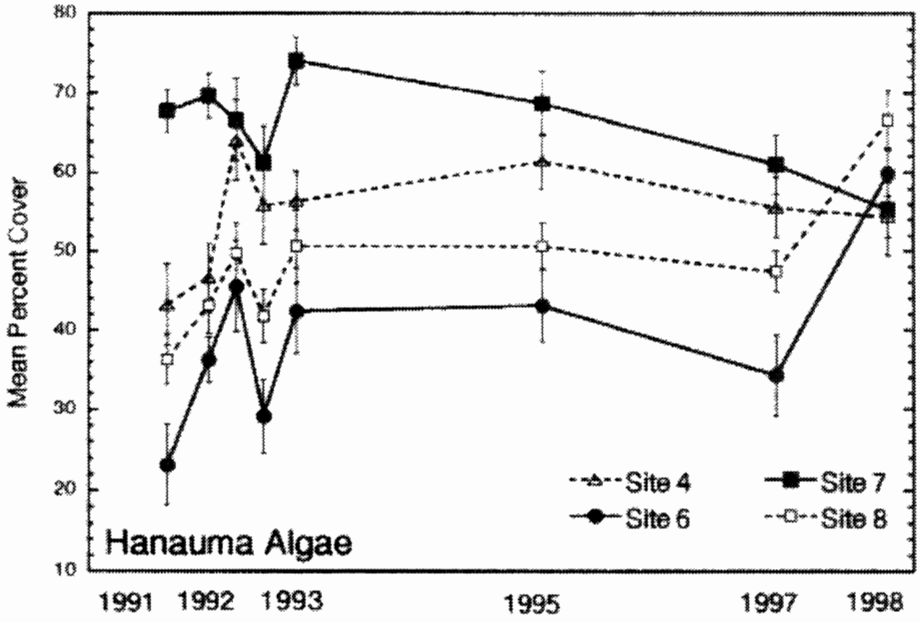


Figure 4. (Opposite page and above) Temporal changes in percent cover of coral, algae, and diseased *Porites* tissues averaged over all sites at Hanauma Bay, Honolua, and Puako, 1991-1998.


Hanauma Bay Coral
Mean Percent Cover
80
70
60
50
40
30
20
10
1991
1992
1993
1995
1997
1998
Site 4
Site 7
Site 6
Site 8


Mean Percent Cover
80
70
60
50
40
30
20
10
1991
1992
1993
1995
1997
1998
Site 4
Site 7
Site 6
Site 8
Hanauma Algae

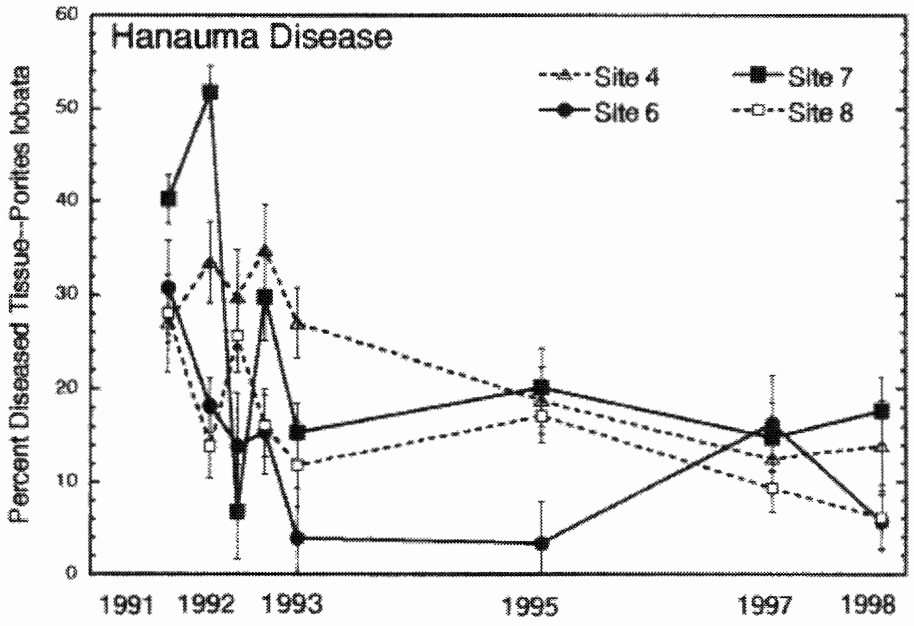
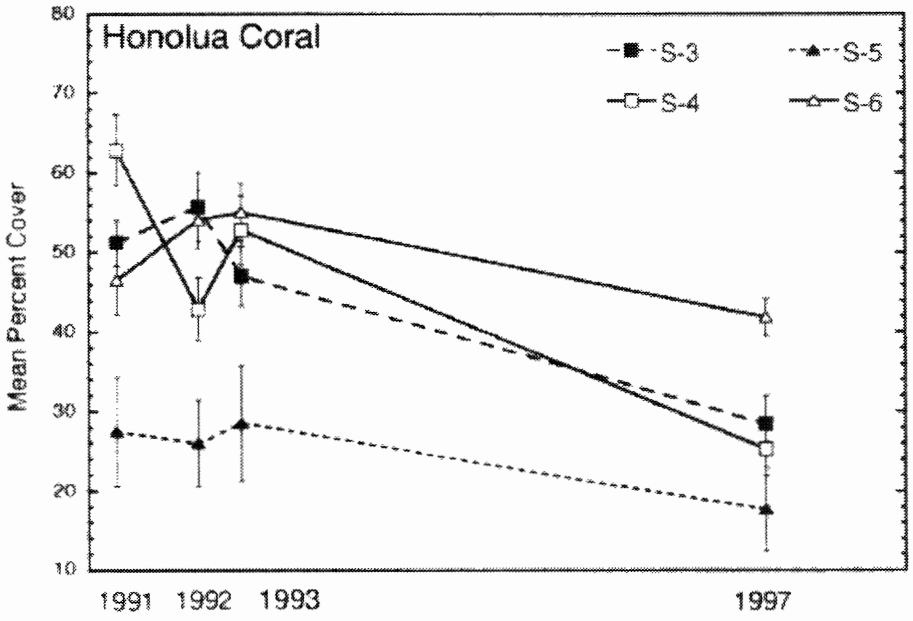
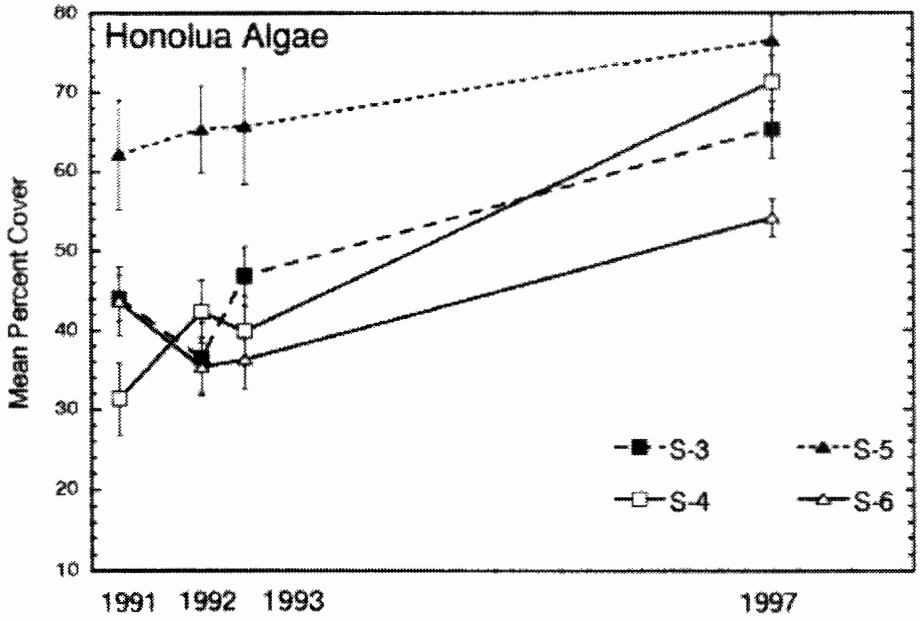


Figure 5. (Opposite page and above) Temporal changes in mean percent cover of coral, algae, and diseased *Porites* tissues for four sites at Hanauma Bay, Oahu, 1992-1998.


Honolua Coral
S-3
S-5
S-4
S-6
Mean Percent Cover
80
70
60
50
40
30
20
10
1991
1992
1993
1997


Honolua Algae
Mean Percent Cover
80
70
60
50
40
30
20
10
S-3
S-5
S-4
S-6
1991
1992
1993
1997

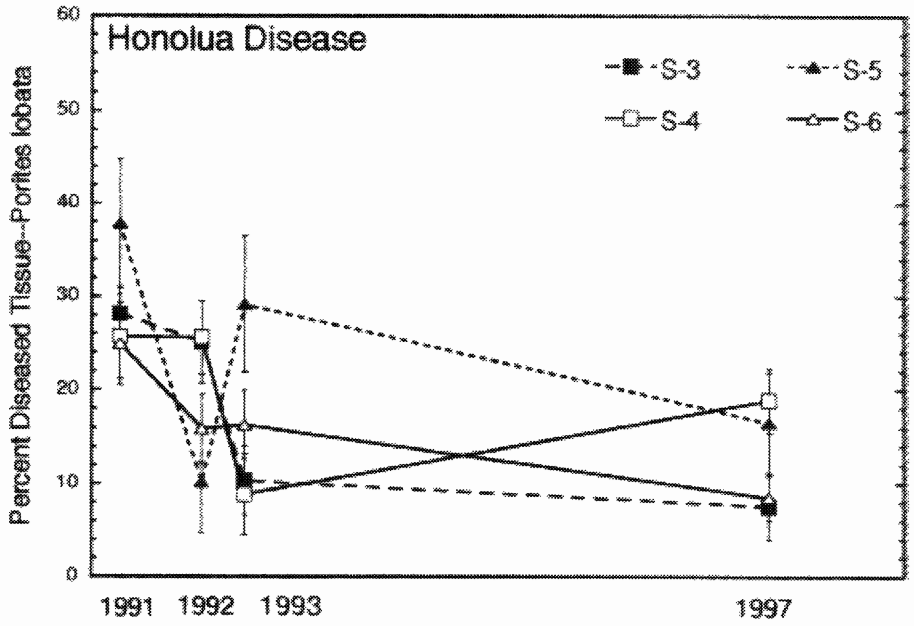
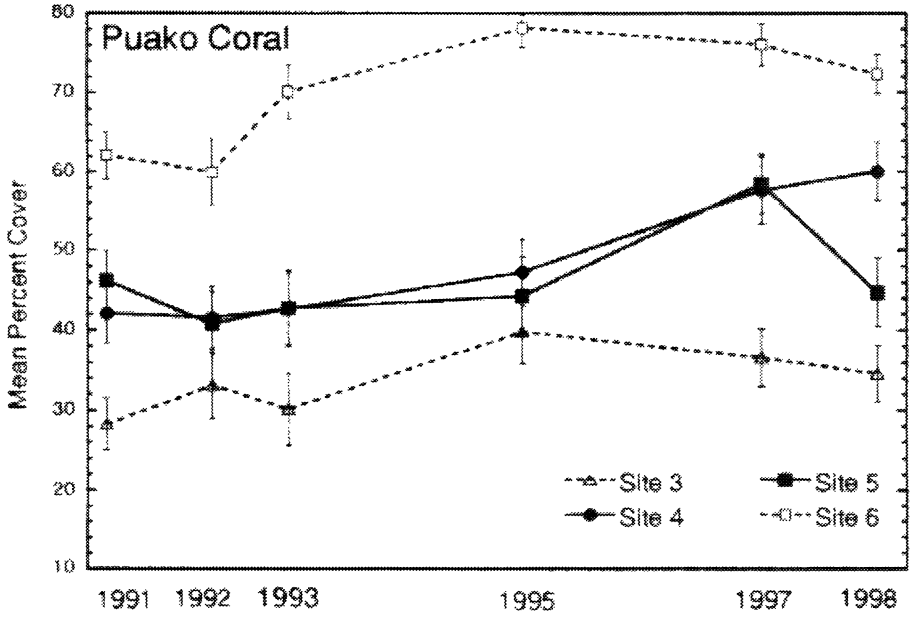
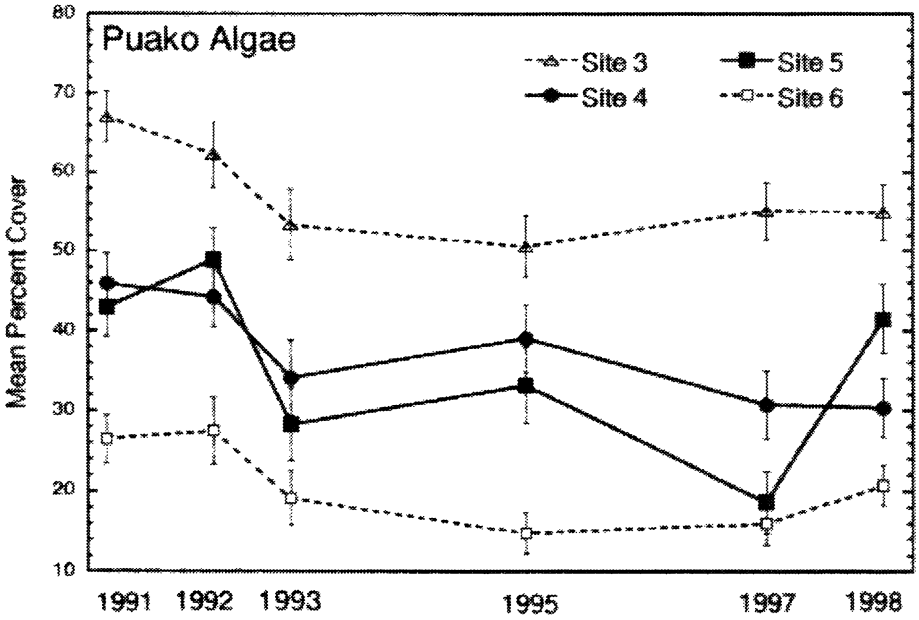


Figure 6. (Opposite page and above) Temporal changes in mean percent cover of coral, algae, and diseased *Porites* tissues for four sites at Honolua Bay, Maui, 1991-1997.


Puako Coral
Mean Percent Cover
80
70
60
50
40
30
20
10
1991
1992
1993
1995
1997
1998
Site 3
Site 5
Site 4
Site 6


Puako Algae
Mean Percent Cover
80
70
60
50
40
30
20
10
1991
1992
1993
1995
1997
1998
Site 3
Site 5
Site 4
Site 6

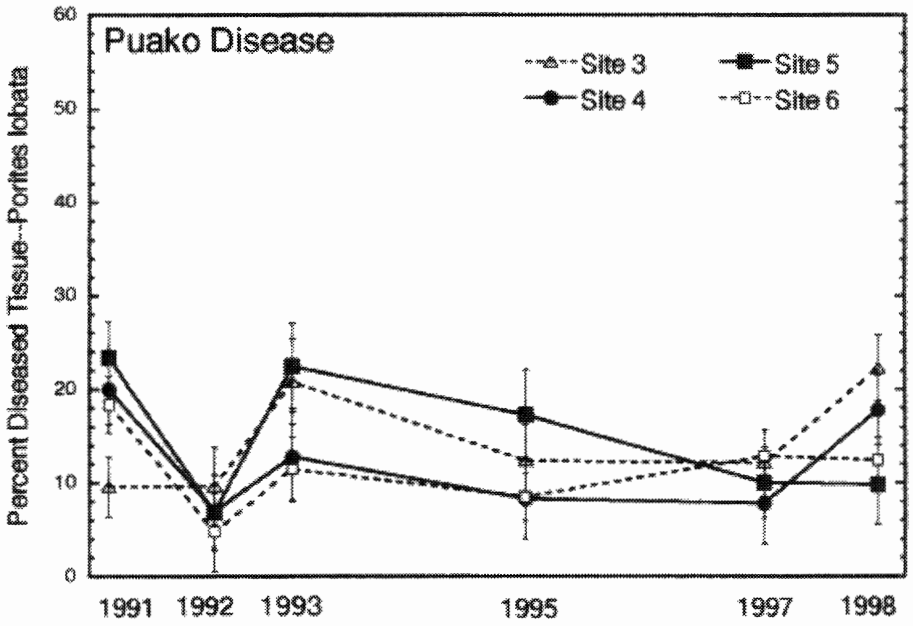


Figure 7. Temporal changes in mean percent cover of coral, algae, and diseased *Porites* tissues for four sites at Puako, Big Island, 1991-1998.

Trimmed pieces of decalcified tissues (oral, aboral, and longitudinal surfaces) were dehydrated and cleared, embedded in Paraplast, and sectioned at 6 mm. Slides from each sectioned block were stained with hematoxylin and eosin (to emphasize general tissue and cell structure), Masson's Trichrome (connective tissue), modified Movat's Pentachrome (mucus secretory cells), Gomori's Methenamine Silver (fungi), and Brown-Hopp's stain (gram +/- bacteria). Tissues and slides were sent to the Registry of Tumors in Lower Animals (RTLA) at the Smithsonian Institution, Washington, DC where they were examined by an expert on disease in Caribbean and Atlantic coral species (EC Peters, TetraTech, Fairfax, VA). RTLA accession numbers for these tissues are ERLN 51-54.

Results

Temporal Changes in Overall Coral Community Structure

Indices of spatial heterogeneity (Table 1) ranged from 1.13 (shallow reef flat, Honolua) to 1.37 (*Porites compressa*-dominated areas at Hanauma Bay). Overall mean percent coral cover as estimated from video analyses of 1997/1998 surveys were 32% for Hanauma Bay, 28% at Honolua Bay, and 53% at Puako (Fig. 4). Mean algal cover (macroalgae and turf) was 59% at Hanauma Bay, 67% at Honolua, and 34% at Puako. Incidence of diseased tissues in the most recent surveys (measured only for the massive Porites) was notably similar at all three sites in 1997/1998, ranging from 11% at Hanauma, 13% at Honolua, and 16% at Puako.

Coral abundance declined 10 to 12% overall from 1991 censuses at Hanauma and Honolua while increasing 8% at Puako (Fig. 4). Specific transect areas showed different responses within each site (Fig. 5; 6; 7). Most of the change in coral and algal cover at Hanauma Bay occurred between 1991 and 1993 censuses, and between 1993 and 1997 at Honolua and Puako. Significant declines in coral abundance occurred at Hanauma Bay for all areas combined between March 1992 and April 1993 (multivariate repeated measures ANOVA; $p=0.002$).

As estimated from video transects, the percent of total Porites tissues affected by disease varied considerably among colonies, areas, sites, and survey periods, but appeared to have been fairly constant at all three sites from 1993 to 1998 (Fig. 4; 5; 6; 7). Coral tissues affected by disease decreased by half (31.5% to 15%) over a one year period (1992-1993) at Hanauma Bay and by over a third (29% to 18%) at Honolua.

Spatial Distribution of Diseased Corals at Hanauma Bay

Mean size of massive *Porites* colonies varied from 0.004 m^2 (Puamana, 5 m depth) to 3.690 m^2 (Olowalu, 3 m depth) (Table 2). Colony size was significantly larger at shallower depths ($r^2=0.62$, $p<0.01$). However, distributions of diseased corals showed no apparent correlation with depth or distance from shore. The percent of necrotic tissue on colonies was significantly correlated with colony size, although the occurrence of tumors showed no such trend (Fig. 8). Turf algae abundance also increased as a function of colony size, although abundance of calcareous algae did not (Fig. 9).

Necroses and tumors were observed on all aspects of colonies (top, sides, and bottom) suggesting that exposure (e.g., to solar radiation (UV) plays little role in their development. The lack of clumping of diseased or tumorous colonies, along with the observation of distinct lines of demarcation between healthy and diseased tissues between adjacent colonies, suggests that these diseases are not infectious through proximity or direct colony contact.

Table 2. Mean proportion of *Porites* colony surfaces occupied by crustose coralline algae, turfs, alpheid shrimp burrows, fish bites, tumors and necroses for each colony intersecting a 25 m transect at each site (Kealakekua, Puako, Puamana, Olowalu, Molokini, Hanauma 1998) or the first 20 colonies intersecting a 50 m transect line (Hanauma Bay 1992, 1994).

											Tumors		
Trans #	Depth (m)	No. Cols.	Mean Col Size (m^2)	Total Col Area (m^2)	Crust	Turf	Shrimp Burrow	Fish Bites	Tumor	Necroses	% cols w/tumors	Mean No./col	Area (cm^2)
Kealakekua Bay, May-98													
1	3	17	0.211	3.59	5.6	5.9	5.3	1.5	3.09	0.00	12	1.12	435
2	3	20	0.886	17.72	8.3	14.8	7.0	4.0	0.05	0.00	5	0.20	13
4	3	42	0.752	31.57	0.5	13.3	3.1	3.2	0.05	0.12	2	0.19	32
Puako, May-97													
5	11	41	0.036	1.49	17.3	18.6	0.2	0.4	0.67	0.00	5	0.05	61
6	12	23	0.101	2.31	7.7	16.1	0.2	0.8	1.10	0.05	26	0.83	107
Puako, May-98													
5	11	38	0.022	0.83	0.0	2.2	2.2	0.3	0.53	0.79	3	0.03	25
Puamana, May-97													
1a	5	12	0.004	0.05	nd	nd	nd	nd	0.00	0.00	0	0.00	0
1b	5	26	0.122	3.18	nd	nd	nd	nd	2.12	0.10	27	1.10	787
Olowalu, May-97													
1a	3	9	0.052	0.47	nd	nd	nd	nd	4.20	0.00	33	0.89	37
1b	3	14	3.690	51.67	nd	nd	nd	nd	0.20	0.00	36	1.90	1116
Molokini,Mar-98													
W-15	5	40	0.037	1.46	0.0	4.3	2.1	0.0	0.03	0.13	3	0.08	6
W-40	13	44	0.010	0.43	0.0	1.3	0.0	0.0	0.00	0.23	2	0.02	1
C-15	5	32	0.081	2.58	0.5	2.3	0.6	0.0	0.18	0.31	19	0.41	43
C-40	13	11	0.068	0.75	0.0	0.5	0.0	0.0	7.27	0.00	36	0.73	27
Molokini, Oct-98													
C-15	5	23	0.082	1.89	7.4	7.4	4.6	0.9	0.00	1.52	0	0.00	0
C-40	13	28	0.066	1.85	0.5	1.3	2.1	1.4	0.43	0.71	14	1.04	281
E-15	5	25	0.015	0.38	0.4	2.6	1.2	0.2	1.00	0.60	4	0.04	12
E-40	13	9	0.006	0.05	0.0	2.8	0.0	3.3	0.00	0.56	0	0.00	0
Hanauma Bay Apr-92													
4	3	20	2.368	47.35	0.0	30.0	6.0	11.0	3.00	4.00	35	nd	19581
6	12	20	0.842	16.84	17.0	3.0	2.0	6.0	5.00	0.00	60	nd	6202
7	8	20	0.810	16.20	3.0	38.0	3.0	5.0	6.00	3.00	45	nd	11640
8	10	20	1.075	21.49	14.0	17.0	4.0	11.0	4.00	3.00	35	nd	7220
RR	9	20	1.072	21.43	11.0	15.0	1.0	7.0	2.00	5.00	30	nd	5560
Wall	4	20	2.388	47.76	0.0	28.0	4.0	9.0	4.00	5.00	45	nd	9180
Hanauma Bay Feb-94													
4	3	20	1.801	36.01	2.0	29.0	7.0	5.0	8.00	3.00	25	nd	28260
7	8	20	0.836	16.72	6.0	27.0	6.0	2.0	3.00	2.00	40	nd	4420
8	10	20	0.211	4.21	22.0	4.0	4.0	2.0	0.20	0.00	5	nd	30
RR	9	20	0.554	11.08	3.0	16.0	5.0	2.0	3.00	1.00	45	nd	4420
Hanauma Bay May-98													
Wall	4	25	0.242	6.05	2.0	30.0	3.2	7.0	1.17	0.61	32	1.24	424
RR	9	37	0.268	9.91	10.0	14.7	3.0	5.0	2.06	1.20	38	1.57	483
Mean over all sites		--	0.624	12.6	5.3	13.3	3.0	3.4	2.08	1.10	22.1	0.57	3347

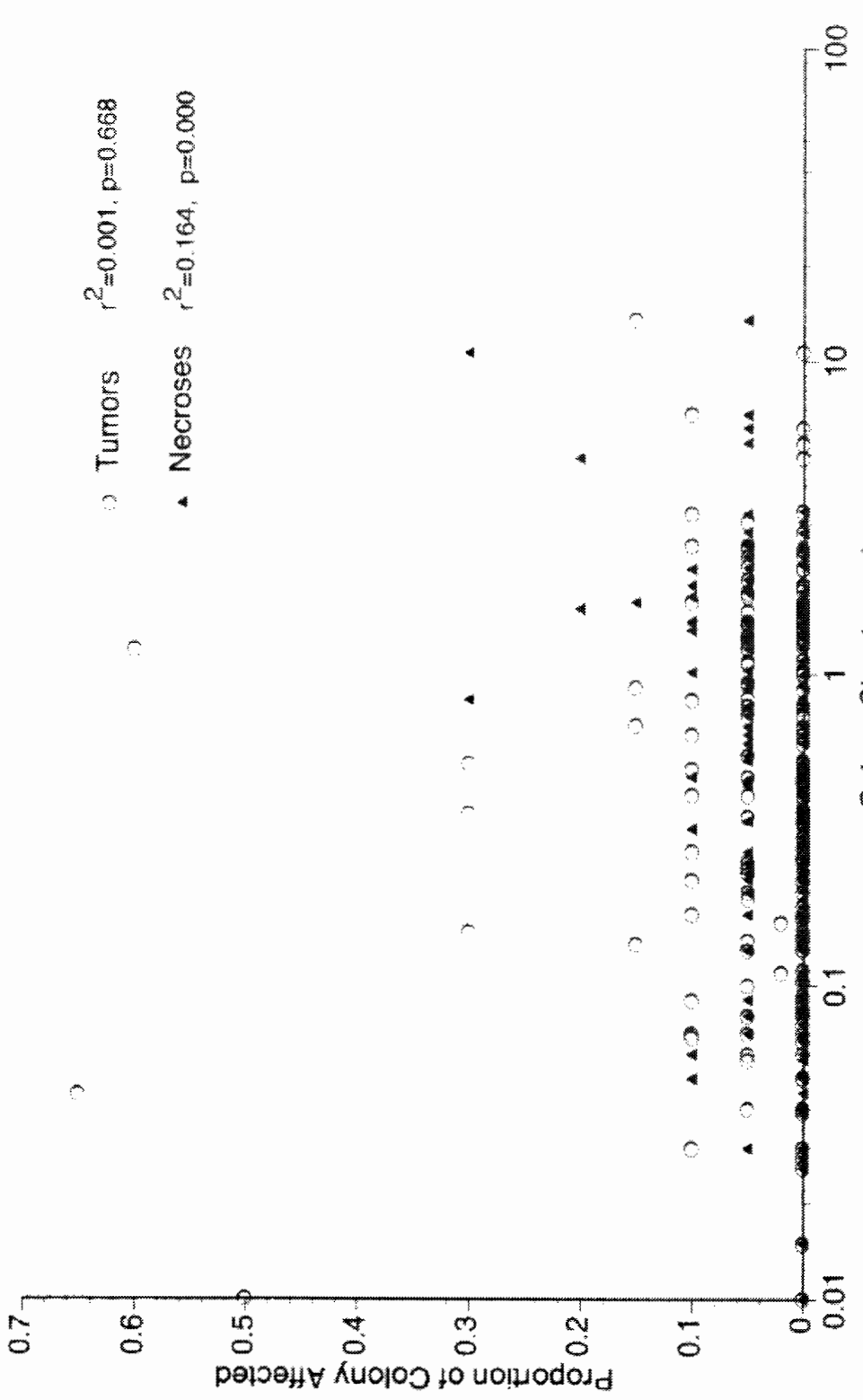


Figure 8. Relationship between colony size and proportion of colonies affected by tumors (open circles) and necroses (triangles) at Hanauma Bay, Oahu.

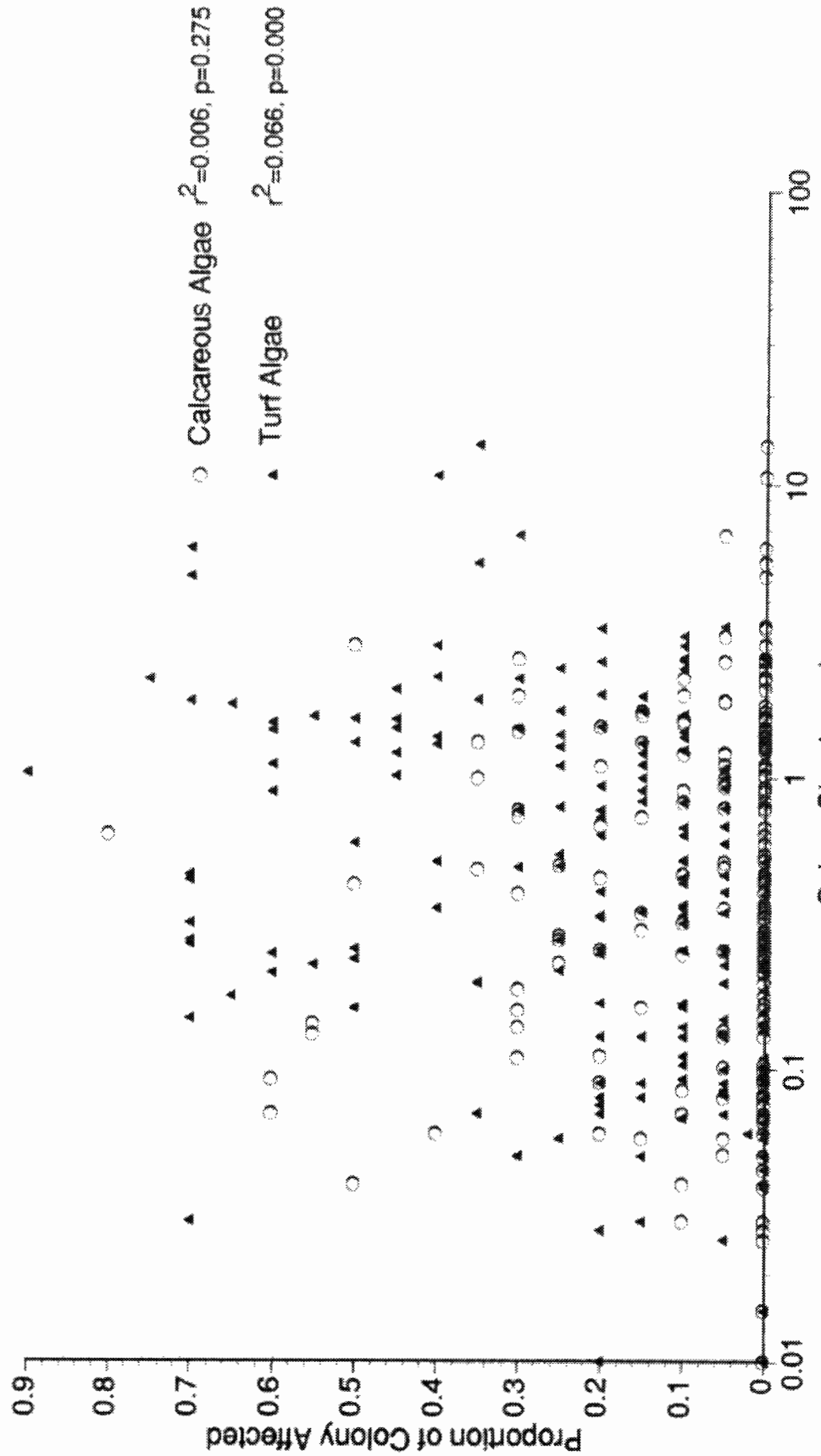


Figure 9. Relationship between colony size and proportion of colonies covered by calcareous algae (open circles) and turf algae (triangles) at Hanauma Bay, Oahu.

Table 3. Change in coral tissue health at macro-sites during initial 2 year study period. + coral tissue increased; - coral tissue decreased; = no measurable change.

Hanauma Bay Macro #	Status: Oct-91	Oct-92	Oct-93
7	tumor	=	=
8	necrosis	=	+
9	necrosis	-	=
10	necrosis	=	+
11	necrosis	-	=
12	tumor	=	=
13	necrosis	-	-
14	necrosis	-	=
15	tumor	-	-
16	necrosis	-	-
17	necrosis	-	-
18	necrosis	+	+
19	necrosis	+	+
20	necrosis	-	-
21	necrosis	+	+
22	necrosis	-	-
23	necrosis	+	-
24	necrosis	-	-
25	necrosis/tumor	-	-
26	tumor	-	=
27	necrosis	-	-
28	necrosis	-	+
29	necrosis	+	-
30	necrosis	-	-
31	necrosis	-	-
32	necrosis	-	-
33	tumor	+	+
34	tumor	+	=
35	tumor	-	-
36	tumor	-	-
Honolua Bay			
Macro #	Oct-91	Jun-92	Oct-92
3	sediment covered	-	=
4	necrosis	-	=
13	necrosis	-	-
14	necrosis	-	=
15	necrosis	-	+
16	sediment covered	-	-
17	bleaching	+	-
Puako			
Macro #	Oct-91	Jan-92	Aug-92
1	necrosis	-	=
2	necrosis	+	+
3	necrosis	=	-
4	necrosis	-	=
5	necrosis	-	=
6	necrosis	-	+
9	necrosis	-	-
10	necrosis	=	-
11	necrosis	-	-
12	necrosis	-	=
13	necrosis	=	-
14	necrosis	=	-
15	necrosis/tumor*	-	-
16	necrosis	+	+
17	necrosis	=	-
18	necrosis	=	-

*tumor decreased in size

Temporal Progression of Diseased Tissue

Of the 72 macro-photo areas of diseased tissues initially established, 53 (74%) were followed over the initial three year period of this study (Table 3). Tags were overgrown by algae or otherwise lost for 6 colonies at Hanauma Bay, 11 colonies at Honolua, and 2 colonies at Puako.

Generally, diseased colonies exhibited a progressive loss of tissue within and beyond the framed areas of observation. Necrotic areas were usually rapidly invaded (within days to weeks) by filamentous algae (*Ceramium* sp. and other unidentified spp). A blue-green bacterial mat was often observed to secondarily overgrow the affected areas, persisting from weeks to months before being replaced by filamentous turf. Areas covered by turf algae trapped fine sediments, particularly on the upper (horizontal) surfaces of colonies, perhaps accelerating further tissue loss.

At Puako, 88% (14/16) of diseased areas died (100% tissue loss within the study frames) within the first 18 months of the study period. Similarly, 86% (6/7) of diseased areas died at the Honolua site. During the initial 30 month observation period at Hanauma Bay, 70% (21/30) of diseased areas died. Recovering lesions (14 at Honolua, 12% at Puako, and 23% at Hanauma) replaced <5% of the lost coral tissues.

A "succession" of lesion recovery was observed to entail an initial invasion of fine filamentous or turf algae. This turf was often grazed by fish as exhibited by numerous feeding scars exposing bare coral skeleton. Grazed areas were then apparently invaded by calcareous algal crusts that spread to cover the area "opened" by coral tissue necrosis and death. Over the time of this study, no areas have yet shown signs of coral larval recruitment. Rather, any increases in coral tissue cover of initially diseased areas were the result of apparent remission of necroses followed by regrowth from adjacent healthy tissues.

Coral tumors exhibited a variety of temporal patterns over the period of the study. Of the ten tumors observed (9 at Hanauma, 1 at Puako), 7 increased

in size and exhibited partial (2) or total (5) mortality (Table 3). These tumors became necrotic and tissues were replaced by turf algae. One tumor showed no apparent change in size, shape or coloration over 2.5 years. Two tumors (one at Hanauma, one at Puako) gradually shrank in size and distinctness, and eventually regained normal appearance. Examination of fresh tissue samples from other tumors in July 1993 (not those under long-term observation), showed a lack of any evidence of gonadal development although surrounding "healthy" tissues were ripe with eggs or sperm. However, subsequent samples of tumors from *P. lobata* and *P. compressa* collected in July 1997, showed gonadal development, although number of oocytes per gonad and number of gonads per polyp were reduced up to 60% (Hunter and Field 1998).

Diagnosis and Pathology of Disease

The most common type of disease in Hawaiian *Porites lobata* [ERLN 51] was characterized by numerous gram negative bacterial aggregates in the gastrodermal cells of tentacles (sometimes deeper in the polyp) and was diagnosed as "stress-related necrosis" (Peters 1984). Necrotic tissues are evidenced by lysed nuclei and cell death. Similar bacterial aggregates have been identified in acroporid corals with "white band disease" in the Caribbean, but their association and/or pathogenicity have not been definitively confirmed.

Two other samples of necrotic tissues [ERLN 53 & 54] showed few or no bacterial aggregates, but secondary invasions of filamentous algae. One of the algal species was identified by K. McDermid (University of Hawai'i-Hilo) as *Ceramium* sp., a filamentous red alga with tiny rootlets that embed in coral epidermis. Such algal invasions may be caused by damselfish bites on the coral tissues resulting in cell lysis or by elevated nutrient loads in the water column increasing the potential for algal growth.

The fourth sample [ERLN 52] was a coral tumor characterized by larger polyps and lighter pigmentation than the surrounding normal tissues. Cell architecture of the tumor appeared normal, but there were numerous bacterial aggregates in the tentacles and coenosarc. In addition, *Nematopsis*-like cysts (gregarine sporozoans) were identified in this sample, the first record of the latter in coral tissues in the Pacific. There was no evidence of infection by fungi or the blue green bacterium *Phormidium corallyticum* in any of the samples. *P. corallyticum* has been shown to be associated with Black Band Disease in reef corals and gorgonians in the Caribbean.

Discussion

This study represents the first record and pathological diagnosis of diseased corals on Hawaiian reefs. Diseased and/or tumorous tissues in corals appeared to be chronic and relatively frequent on *Porites lobata* at all three sites in the present study, although they are particularly apparent at Hanauma Bay, both in number and proportion of coral tissue affected.

Although the potential causes of coral necroses are still conjectural, bacterial aggregates found in tissues of *Porites lobata* from Hanauma Bay were similar to those found in *Porites astreoides* in the Caribbean (Peters et al. 1983). Peters and Santavy (1996) suggest that these bacteria may be mutualistic in healthy corals but become opportunistic pathogens when corals have been environmentally stressed. Potential stresses may include elevated nutrient levels that encourage algal overgrowth; mechanical abrasion from shifting sand or fish predation; smothering of tissues from episodically high levels of sedimentation, and/or other factors that are currently unidentified. Rates of colonization and persistence of invading algae in the newly "opened" (dead) areas perhaps govern consequences of disease.

The cause and cellular basis of tumors in *Porites* corals is also unknown. Some exhibit a form of "gigantism" (Peters et al. 1986), similar to those reported in *Platygyra* in Australia by Loya et al. (1984), while others show uncontrolled or "hyperplastic" growth. A unique feature of the *Porites* tumors noted in the present study is the apparent ability of some to recover a normal appearance. Whether cells in these tumors are eventually replaced by normal cells or actually mutate back to a "normal" condition is an area of great interest. Of additional interest is the long-term impact of tissue loss on the coral community from tumors that become necrotic.

An outbreak of black band disease at Looe Key, Florida, in 1986 was found to be associated with high temperatures, eutrophication, and sedimentation (Peters and Santavy 1996). However, outbreaks of other diseases do not appear to be consistently associated with known anthropogenic stressors (e.g., pollutants, proximity to urban centers). Background rates of tissue loss as a result of disease (i.e., non-outbreak occurrence levels) and the relationship to subsequent algal overgrowth are currently unknown for any reef ecosystem. Nutrient and toxic contaminants input levels are unknown for most Hawaiian reef sites. Estimates of shoreline water column concentrations of nitrates, ammonium, phosphates, chl a, and enterococcus are monitored biweekly at Hanauma Bay by the Hawai'i State Department of Health, but may not reflect absolute rates of input nor amounts of uptake of nutrients or toxic contaminants by the ecosystems.

Previous studies provided a basis for comparison of present coral and algal abundance with past conditions as Hanauma Bay (Anderson 1978), Honolua Bay (Torricer et al. 1979), and Puako (Hayes et al. 1982). The maps provided from the previous Puako survey (and the fairly homogeneous nature of the reef areas at this site) allowed reasonably accurate comparisons. Exact locations of study areas within each site were not available from the previous studies at Hanauma Bay and Honolua Bay. Coral abundance at three of the four Hanauma Bay reef areas was remarkably similar over a 16-year period (Table 4). The fourth area (Area 6 in the present study) showed a substantial decline in coral abundance between 1976 and 1992. Of note is the apparent increase in algal abundance in 1992 when compared to 1976 levels at all areas surveyed. Whether this increase in algal cover is real or the result of differences in scoring by individual investigators cannot be resolved. Such questions emphasize the need for permanent visual records of reef sites.

The importance of long-term, quantitative studies of temporal and spatial variability of coral reef ecosystems has become increasingly apparent over the past few years (Knowlton 1992; Hughes 1993). Symposia and workshops at national and international meetings have reiterated the present lack of information necessary to make scientifically sound decisions as to whether potential environmental changes (e.g., increased pollution, seawater temperatures, sea level, UV radiation) are causing or will cause significant changes in coral reef vitality. It is hoped that the data presented here will provide the basis and framework by which long-term changes on Hawaiian reefs might be evaluated and managed.

At least three general types of coral disease commonly occur on Hawaiian reefs (necroses, band diseases, tumors; Hunter pers. obs.). Diseased coral tissue is rapidly invaded by unidentified fine filamentous (turf) algae and cyanobacteria. There appear to be three resultant outcomes of this invasion: 1) a successional change from turf to crustose coralline algae on which new coral recruits may become established, 2) recovery and overgrowth by healthy, adjacent coral tissues, or 3) persistence of the turf community and a change from coral to algal domination of the affected coral colony.

The change from an ecosystem dominated by reef-building corals to one dominated by non-reef-building algae is usually gradual and difficult to detect with statistical rigor using traditional repeated measures assessments of macro-scale surveys of reefs. In future studies, diseased tissues will be photographed

Table 4. Comparison of percent coral and algal abundance data from present study to data from previous studies. * refers to site designations of previous studies

Hanauma Bay

Mean Coral Abundance	Anderson, 1978* Sep-76	present study Oct-92
Area:		
10', p.293-94*, 4	28.10	33.90
25', p.254-55*, 7	36.30	30.22
30', p.301-02*, 8	37.65	34.65
35', p.277-78*, 6	64.76	45.51
Mean Algal Abundance		
Area:		
10', p.293-94*, 4	29.00	64.07
25', p.254-55*, 7	31.50	66.67
30', p.301-02*, 8	29.00	49.78
35', p.277-78*, 6	17.00	45.58

Puako

Mean Coral Abundance	Hayes, et al., 1982* Spring. 1990	present study Apr-93
Area:		
3A*, 3	16.00	30.13
4A*, 4	65.00	42.72
5A*, 5/6	63.00	42.78/70.11
Mean Algal Abundance		
Area:		
3A*, 3	73.00	53.34
4A*, 4	29.00	34.05
5A*, 5/6	31.00	28.22/19.11

Honolua Bay

Mean Coral Abundance	Torricer, et al., 1979* Jun-76	present study Jun-92
Area:		
9* (6m), 4 (6m)	30.80	42.93
13* (2-8m), 6 (3m)	36.40	54.16
15* (1-6m), 3 (3m)	15.90	55.78
Mean Algal Abundance		
Area:		
9* (6m), 4 (6m)	5.50	42.40
13* (2-8m), 6 (3m)	43.40	35.40
15* (1-6m), 3 (3m)	16.00	36.56

using a higher magnification close-up lens. Time-series photographs will be digitized and the process of change recorded at each survey interval. Algal taxa involved in each successional stage will be characterized within and among study sites, and principal component analyses will be used to assess any taxonomic and/or geographic trends in the outcomes of disease on Hawaiian coral reefs.

Acknowledgments

Numerous people have assisted in various aspects of the development and implementation of the research objectives of this project since its inception: G. Aeby, R. Ames, M. Awai, K.O. Beckman, E. Brown, B. Carlson, B. Carman, E. Chornesky, S. Coles, M. Cottone, N. Craig, J. Culp, F. Coopersmith, G. DeCouet, C. Evans, S. Field, A. Friedlander, M. Gassel, D. Gochfeld, R. Grigg, D. Gulko, B. Hannum, J. Hardy, S. Hau, P. Hendrix, J. Holland, A. Hong, L. King, R. Kosaki, B. Larson, M. Lesser, K. Lowe, J. Maragos, K. McDermid, O. McMillan, K. Meier, R. Nishimoto, D. Pence, E. Peters, D. Potts, B. Rinkevich, S. Romano, S. Russell, D. Schricte, S. Scott, C.M. Smith, J.E. Smith, N. Stambler, D. Stellar, K. and U. Stender, M. Stephenson, R. Trench, L. Udall, R. Vago, P. Vroom, S. Wells. Particular thanks are due to S. Folsom for the majority of the video analyses; A. Hong for providing research access to Hanauma Bay; E. Brown for diving and logistical support on Maui; B. Carman and the DAR crew for diving and logistical support on the Big Island; D. Schricte for identifying the initial question and providing great photographs; M. Stephenson for enormous assistance in every aspect of the study; and Hawai'i DAR and UH Sea Grant for funding support. While little would have been accomplished without the contributions of those listed above, all errors and omissions reside with the author.

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Chapter 9:

Topographical Relief as a Proxy for the Friction Factors of Reefs: Estimates of Nutrient Uptake into Coral Reef Benthos

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Abstract

The energy dissipated by a reef community is correlated to the topographical relief and thus is an important parameter for calculations of mass exchange of metabolites between coral reef benthos and the water column. Such calculations can provide estimates for impacts of point and non-point source pollutants to coral reef communities. I recommend obtaining baseline topographical relief for all monitoring sites in the Hawaiian Islands.

Keywords

topographical relief, benthic monitoring, coral, habitat heterogeneity

Introduction

Management of coral reefs requires some basic understanding of the rate processes that control community metabolism. Exchange of compounds from water to benthos is essential to estimate impacts of point or non-point source inputs of dissolved compounds into coastal seawater. We have been working on understanding nutrient uptake (uptake of ammonia, nitrate and phosphate) into coral reef benthos, and we have developed a straightforward method for estimating the maximum uptake rate of nutrients into coral reefs. Maximum uptake rates have provided good predictions of actual uptake in experimental flumes; however we have not yet confirmed this for natural reef communities. Nevertheless, our equations should be useful to managers and consultants in estimating anthropogenic impacts from stream or sewage discharge, etc.

This paper is presented as one argument for making topographical relief (TR) a standard parameter for monitoring sites, where TR is the actual length of a chain laid across the benthos divided by the projected length (Fig. 1; Loya 1978). I believe TR can be used as a proxy for another parameter, the friction factor, which is difficult to measure in coral reef communities. By having an estimate of TR, water velocity and the concentration of the compound of interest, predictions of uptake can be made relatively easily with a calculator.

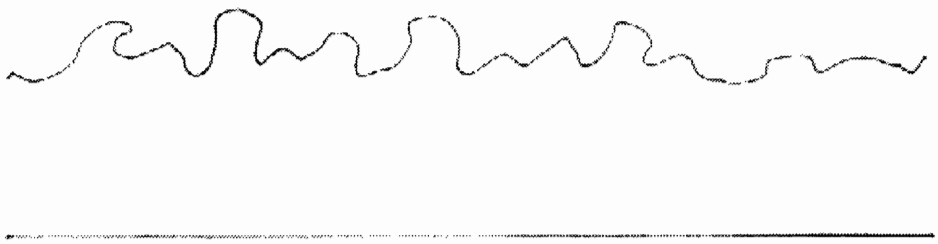

Fig. 1: Topographical Relief is the length of over and around the benthos divided by the projected linear length.

Background on mass transfer

When organisms have a very high affinity for dissolved compounds in water, they have a high uptake rate. The uptake rate into the organisms can be faster than the diffusion of molecules in the water adjacent to the organisms. Under these conditions, a depleted boundary layer in the concentration of these molecules develops between organisms and the water adjacent to their surfaces. The concentration-depleted boundary layer grows until the movement of water determines the outer edge of the boundary layer. The transport or diffusion of molecules through this boundary layer, from the outer edge to the surface of the organism, sets the upper limit for the rate of uptake. This condition is called mass transfer limitation (see Bilger and Atkinson 1992 for a review, Fig. 2). Nutrients are in great demand by benthic autotrophs of coral reefs, including all macroalgae, filamentous algae, corals and coralline algae, so it is not surprising that nutrients are mass transfer limited. It is likely that many other substances are also mass transfer limited.

Nutrient uptake into coral reef benthos (m_s) is proportional to the concentration of the substance in the water (C_b) and water velocity (U_b) by a dimensionless rate constant, termed the Stanton number (St_m):

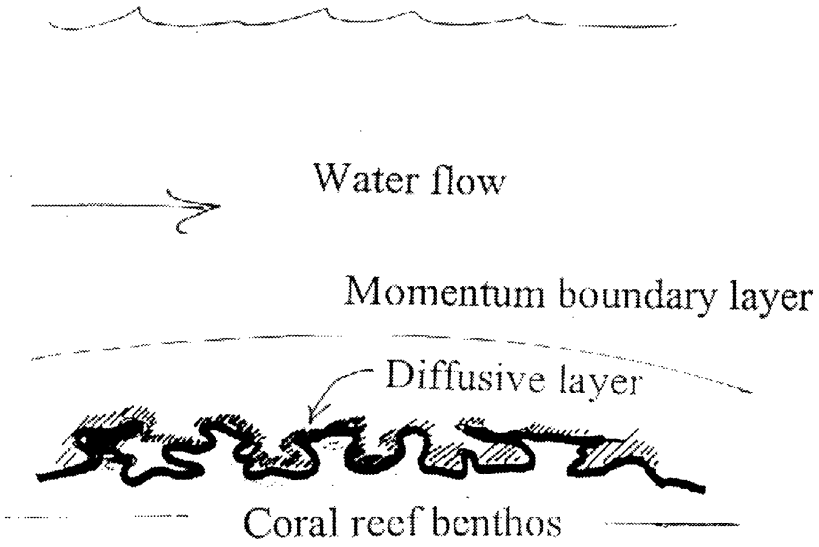

Fig. 2: Diagram showing a depleted boundary layer within a momentum boundary. Diffusion of nutrient compounds through the diffusive boundary layers is the rate limiting step for nutrient uptake. This sets a maximum limit for nutrient uptake at any given concentration and water velocity.

$m_s = St_m U_b(C_b-C_w)$, (Table 1, Eq. 1). C_w is the average concentration of the metabolite at the surfaces of benthic organisms which has been shown to be close to zero for experimental reef communities growing in low nutrient water (Baird and Atkinson 1997). As communities of organisms experience elevated concentrations of nutrients, C_w increases and $St_m U_b$ decreases (Bilger and Atkinson 1995). Thus when C_w is assumed to be zero, Eq. 1 is a maximum uptake rate. St_m is on the order of 10^{-4} to 10^{-3}.

The Stanton number is positively correlated to a coefficient called the friction factor (c_f, Fig. 3, Table 1 Eq.2). The friction factor is another dimensionless number that describes the amount of energy lost by friction to the bottom at a given water velocity. Each benthic community has characteristic friction factors, and these have been measured to a limited extent in experimental flumes at Coconut Island, Hawai'i Institute of Marine Biology (Thomas and Atkinson 1997; Baird and Atkinson 1997). The friction factor is best thought of as a physical characteristic of the bottom community. It would be ideal to have friction factors for all communities of reef benthos around Hawai'i, but they are difficult to measure.

Table 1: Equations to calculate uptake rates from topographical relief.

Uptake Rate:	$M_s = St_m U_b(C_b-C_w)\ C_w \sim 0.$	Eq. 1
Stanton Number:	$St_m = (c_f/2)/[0.9 + (c_f/2)^{1/2}/St_k]$	Eq. 2
Friction Factor:	$c_f = 0.194\ TR - 0.186$	Eq. 3
St of Roughness Elements:	$St_k^{-1} = 5.19 Re_k^{0.2} Sc^{0.44} - 8.48$	Eq. 4
Roughness Reynolds Number:	$Re_k = U_b(c_f/2)^{1/2} k'/\nu$	Eq. 5
Schmidt Number:	$Sc = \nu/D = 475$ for NH_4; 495 for NO_3 and 1340 for PO_4 @ 25 °C	Eq. 6

TR is topographical relief; D is the diffusivity of a compound ($m^2\ s^{-1}$); U_b is the velocity of the bulk water ($m\ s^{-1}$); V is kinematic viscosity of seawater at 25 °C = $0.94 \times 10^{-6}\ m^2\ s^{-1}$; k' is mean height of the bottom assemblages and only needs to be an estimate (0.05 m for example).

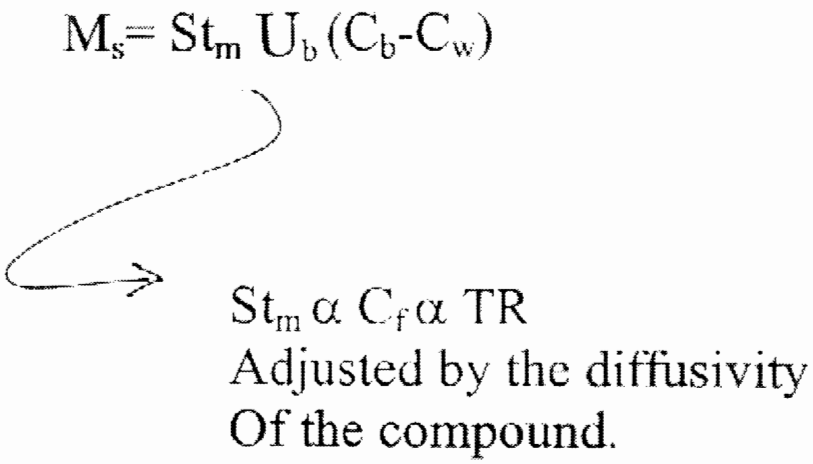

Fig. 3: Diagram of the equation for mass transfer, showing the dependence of Stanton number on topographical relief.

We have shown, in experimental flumes at Coconut Island, that TR is positively correlated to friction factor (Fig. 4; Table 1 Eq. 3). Thus as a bottom community becomes rough in gross features, TR increases and the community creates more drag on the water, enhancing the water-surface contacts and increasing mass exchange. Therefore the Stanton number is a physical descriptor of reef benthos (Baird and Atkinson 1997), and describes how well the reef can exchange metabolites with the water column.

For example, suppose a monitoring site near a stream mouth or groundwater source has a TR of 1.5. By inspection of the regression in Fig. 4 (Table 1, Eq. 3), a TR of 1.5 gives a friction factor or c_f of about 0.1 (dimensionless). Using Equations in Table 1, a c_f of 0.1 corresponds to a St_m of 6.9 x 10^{-4} for NO_3 at

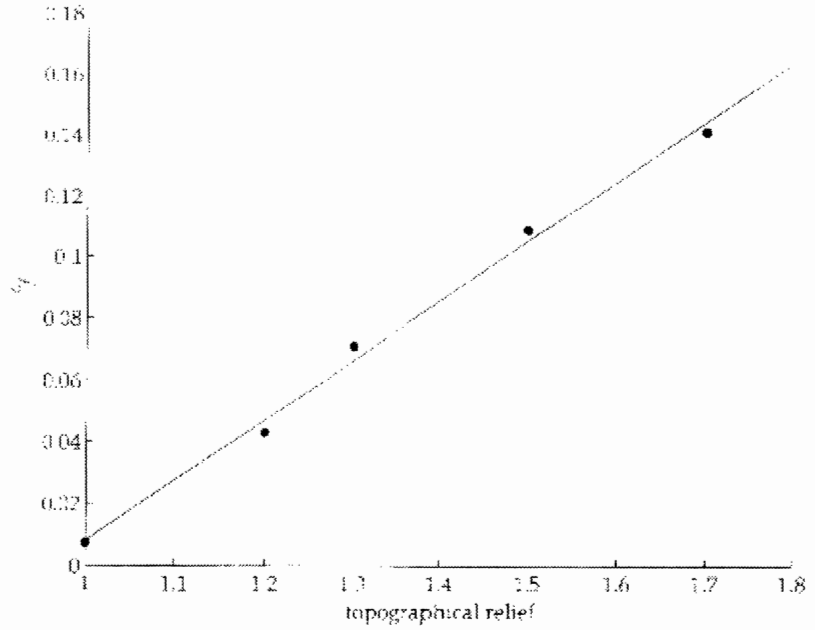

Fig. 4: Correlation of topographical relief with friction factor for five reef communities in a flume at Coconut Island.

25°C (diffusivities of molecules are different, giving different values for Schmidt numbers, see Table 1 Eq. 4 & 6). As explained above, nutrient uptake is directly proportional to St_m (Table 1 Eq. 1). Given a relatively high NO_3 concentration of 1μM (mmol m^{-3}), a water velocity of 0.10 m s^{-1}, the estimated uptake rate of NO_3 is 6.0 mmol m^{-2} d^{-1} ($6.0 = 6.9 \times 10^{-4} \times 0.10 \times 1.0 \times 60 \times 60 \times 24$). Usually concentrations of nutrients in water near stream mouths and groundwater sources are varying spatially, so some numerical integration of the uptake rates is expected. Nevertheless these calculations give good estimates of nutrient uptake. We expect that uptake or release rates of other metabolites can be estimated using this approach, by varying the hypothetical "wall" concentration or C_w in Eq. 1. I believe every monitoring site in Hawai'i should have at least a baseline measurement of TR, as a proxy to the friction factor, so we may calculate Stanton numbers and make estimates of nutrient uptake or exchange. It is probably only necessary to get a topographical relief every year or so to characterize a site, or after any major storm that restructures the bottom.

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Chapter 10:

Monitoring Coral Reefs and Fishes in the Florida Keys

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Abstract

The Florida Keys reef system extends over 370 km and is under different levels of protection by several state and federal agencies. Reef monitoring of the benthos and fishes in the Florida Keys is intended to detect resource changes, evaluate the success of restoration efforts, and increase the knowledge and understanding of marine ecosystems. A hierarchal monitoring approach seeks to answer management questions across disciplines with a focus on the effects of 19 no-take zones established by the Florida Keys National Marine Sanctuary in 1997. Level I focuses on critical factors including processes as well as structure at specific sites. Level II monitoring covers a broader area at lower intensity. Level III focus on assemblage structure changes using volunteer monitoring. Sanctuary Program coordination ensures that: the parameters measured are relevant for both scientific and management purposes and that the same sites are being monitored so that benthic and fish monitoring compliment each other. The initial 5-yr phase will test the hypothesis that the intensity of extractive effort (including commercial and recreational fishing, spearfishing, marine life collecting, and recreational collecting) has no effect on the composition, abundance, and size of reef fishes, lobsters, or sessile benthic organisms or on certain population and trophic processes. The research design is set up to test specific hypotheses involving the effects of no-take marine reserves and trophic interactions between fishery species and the benthos. Fish surveys are conducted throughout the reef tract using primarily stationary plot sampling

by scientists and roving diver surveys by volunteers. Advantages and disadvantages of four visual fish monitoring methods are compared: stationary plot, belt transect, timed search, and roving diver.

Keywords

reef fish, marine reserves, benthic monitoring, visual census, belt transect

Introduction

The Florida Keys reef tract extends 370 km from Miami to the Dry Tortugas, Florida (Jaap 1984). The reef ecosystem supports extensive commercial, subsistence, recreational, and aesthetic fisheries, as well as other activities (Bohnsack et al. 1994), but is vulnerable to damage from exploitation, pollution, and habitat alteration associated with coastal development and rapid regional population growth, as well as changes in water quality and flow from Florida Bay (Smith-Vaniz et al. 1995). Because of the importance and vulnerability of Florida coral reefs, the U.S. Congress created the 9,500 km^2 Florida Keys National Marine Sanctuary (FKNMS) in 1990. After 6 years of planning, the management plan became effective on July 1, 1997 as a joint effort between Florida and the federal government (U.S. Department of Commerce, 1996). A critical part of the plan was the creation of 19 no-take areas to protect Sanctuary resources (Suman 1997; Bohnsack 1997). These include 14 small Sanctuary Preservation Areas (SPAs); four research only Special Use Zones (SUZs) that protect coral reefs; and one large (30 km^2) Ecological Reserve (ER) near Key West that protects multiple habitats from shore out to 20 m depth (Fig. 1). Four additional SPAs allow only catch and release trolling as an experiment.

Reef monitoring in the FKNMS is intended to improve management by detecting resource changes, evaluating the success of restoration efforts, and increasing the knowledge and understanding of marine ecosystems. Monitoring coral reefs is a challenge because of their structural complexity, high variability, rich biological diversity, and complex ecosystem interactions. Because of the controversy over establishing no-take zones, special emphasis was put on monitoring and assessing the impacts of marine reserves on resources and human activities. We describe the monitoring strategy and methods used to monitor reef benthos and fishes. Three specific goals of the FKNMS monitoring program are: (1) to evaluate the effectiveness of "no-take" zones for protecting and understanding marine biodiversity, enhancing non-consumptive activities, promoting conservation, and increasing fisheries knowledge; (2) to measure biological changes inside and outside no-take marine zones; and (3) to monitor trends in resource use and reef conditions throughout the FKNMS.

Methods

Site Description

Because the Keys cover a large geographical area, they are divided into regions to test for geographical differences: Upper, Middle and Lower Keys, and Dry Tortugas. The Florida Keys are also managed under different levels of protection in terms of allowed activities by Florida and two federal agencies. The FKNMS is managed by the National Oceanic and Atmospheric Administration (NOAA) and does not include Biscayne, Everglades, and Dry Tortugas National Parks which are managed by the National Park Service.

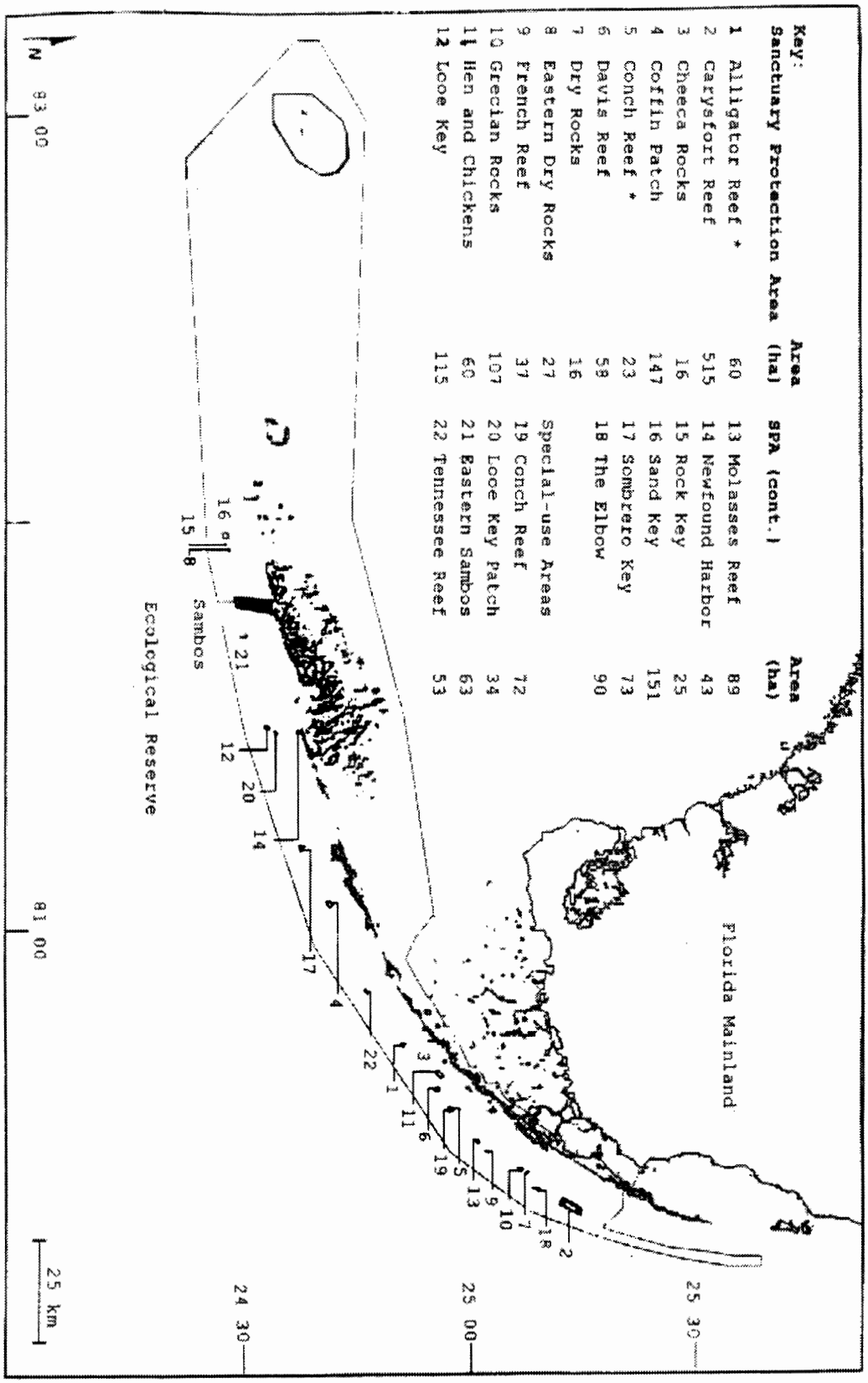

Figure 1. Map of the Florida Keys showing Florida Keys National Marine Sanctuary preservation areas and the Sambos ecological reserve.

Monitoring Strategy

The FKNMS uses a three level monitoring strategy. Level I focuses on changes in ecosystem structure and function in three no-take zones: the large Western Sambo Ecological Reserve and a tiny research SPA at Eastern Sambo near Key West, and the largest SPA, Carysfort, at the northern end of the Sanctuary. Level II focuses on changes in ecosystem structure (species abundance) and human activities and attitudes in the smaller no-take zones. Level III uses volunteers to focus on changes in ecosystem conditions in all of the zones. The initial 5-yr phase tests the hypothesis that the different levels of extractive effort (including commercial and recreational fishing, spearfishing, marine life collecting, and recreational collecting) have no effect on the composition, abundance, and sizes of reef fishes, lobsters, or sessile benthic organisms nor on selected population and trophic processes.

Benthos

Predictions of response of fish communities to no-take management are relatively straightforward because they involve primarily direct interactions (people killing fishes). However, the predicted response of benthic communities is ambiguous because of the complex trophic cascade of indirect interactions. Unfortunately, there are few studies that have explicitly evaluated responses of benthic communities in no-take reserves. In those cases where benthic responses have been demonstrated, the indirect interactions are most often mediated by herbivorous sea urchins (e.g., McClanahan and Muthiga 1988; Sala et al. 1998). Since an epizootic event in 1983, herbivorous urchins on Caribbean reefs have been quite rare and so are not expected to mediate trophic cascade effects in the Caribbean. Figure 2 illustrates the hypothesized trophic cascade operating in Florida Keys reefs. The hypothesized net response of corals and macroalgae to no-take management depends on the relative strength of each link in the cascade.

For example, via one path the replenishment of piscivorous fishes in reserves may decrease damselfish populations, decreases in damselfish induced coral mortality, and thereby increase the standing stock of coral. Alternatively, via a separate trophic cascade path, the replenishment of piscivorous fishes in reserves may decrease grazing fish populations, decrease grazing intensity on macroalgae, increase macroalgal stocks and competitive pressure on coral recruitment and growth, and thereby *decrease* coral (see Fig. 2). To further complicate the picture, grazing parrotfishes in the Florida Keys have been found to directly consume corals in experimental studies (Grottoli-Everett and Wellington 1997; Miller and Hay 1998). Thus, a decrease in grazing fish populations may also decrease direct predation on coral. In order to evaluate these hypothesized trophic cascade pathways, various aspects of the benthic monitoring program are assessing the relative intensity or impact of the links between grazing fishes and macroalgae, and between coral "predators" (snails and damselfishes) and corals as well as standing stocks of territorial damselfishes (*Stegastes planifrons*), corallivorous snails (*Coralliophila abbreviata*), macroalgae, and corals in no-take reserve and reference sites.

Level I benthic monitoring addresses changes in the structure (corals, sponges, macroalgae, mobile invertebrates) and function (coral recruitment and grazing) of the benthic community in the three reserves and three fished reference sites. Three major studies are being conducted with monitoring stratified into forereef depth zones at 7 m and 18 m. First, non-fixed video transect monitoring (Aronson et al. 1994) is being used to monitor changes in benthic community structure using parameters of percent cover of live coral, other sessile invertebrates, and macroalgae, as well as density of hard and soft coral colonies and sponges. Ten 20m x 0.4m transects are video-recorded annually at each site and depth strata. Non-overlapping frames are analyzed by random point counts. Second, repeated annual photographs in conjunction with visual census of randomly situated permanent plots are used to quantify rates of scleractinian coral recruitment and survival. Third, macroalgae standing biomass is being estimated

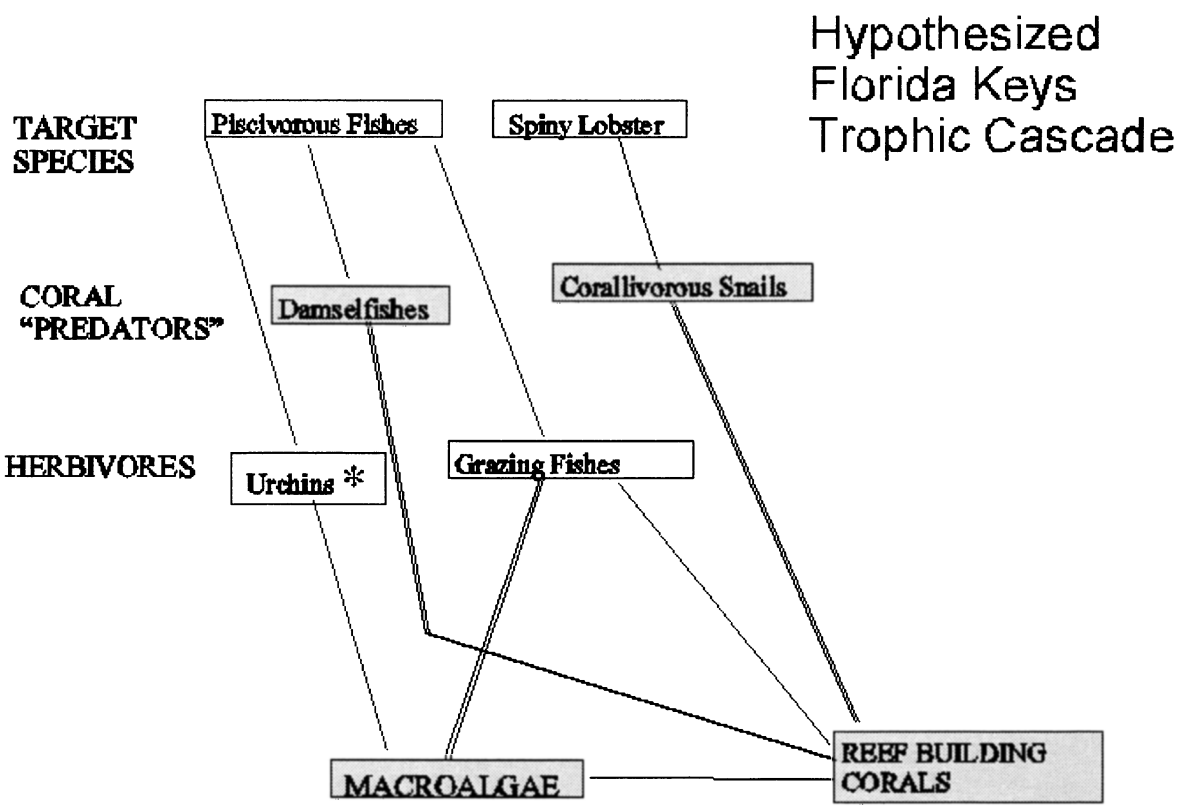

Figure 2. Schematic illustration of the hypothesized trophic cascades structuring Florida Keys coral reefs. The dominant structuring benthic process is competition between macroalgae and corals, depicted by solid arrow. Dashed arrows depict standard predation (negative impact on the lower component). Dotted arrow indicates mortality imposed on corals by territorial damselfishes in the establishment of their territories on coral heads. Note (*) that herbivorous urchins are rare on Caribbean coral reefs since the 1983 die-off. Shadowed boxes and links are being explicitly assessed in the Level I monitoring program and supplemental studies in order to elucidate the dominant trophic pathways whereby benthic communities respond to target stock replenishment in FKNMS reserves.

quarterly by removing macroalgae from haphazard 40 x 40 cm plots placed along the video transects. These samples are sorted to genus and dry weights determined. Also, assays of relative grazing rates are performed quarterly at the shallow depth of each site by quantifying the loss of outplanted sprigs of a variety of macroalgae and seagrass during a fixed time of exposure to ambient grazers.

Supplemental studies to the Level I effort are assessing the standing stocks and impacts of damselfish and corallivorous snails at reserve and reference sites in the northern region of the Sanctuary. Snail abundance, size structure, and impact on coral prey are assessed twice per year by careful scrutiny of 100-150 colonies of *Acropora palmata* (the snails' preferred prey) and 10-20 colonies of *Montastrea* spp. at each site for the cryptic snails. Snail feeding scars are photographed and the snails are counted, measured, and returned to the colony from which they came. Also, permanent 5m x 5m plots at two paired reserve and reference sites are being used to map and monitor changes in abundance and territory size of coral-killing damselfishes (primarily *Stegastes planifrons*) as well as the total area of coral killed (primarly *Montastrea* spp.). These plots are also used to quantify other parameters hypothesized as links between predatory fishes and invertebrates and the structure of reef benthic communities, such as

density of herbivorous sea urchins (*Diadema antillarum*) and bite scars on corals made by grazing parrotfishes.

Fishes

Fishes are an essential and conspicuous component of coral reef ecosystems. Factors that increase mortality, such as pollution, fishing, and loss of habitat are eventually reflected in species composition, population abundance, and average size. Because destructive sampling was not appropriate, fishery-independent, non-destructive, diver-based visual methods are used to assess reef habitats. This monitoring has documented changes in fish populations in the Keys (Bohnsack 1982) and shown that several species are over-fished (Ault et al. 1998).

Monitoring is intended to assess the impacts of zone differences on fish communities. Zones provide different treatments (protected, partially protected, and unprotected areas) at replicated sites. Falsifiable hypotheses and alternatives (Table 1) were developed to examine the following six central themes:

1. **Abundance, average individual size, and spawning potential of exploited species will increase within reserves.** Alternative: There will be no difference over time or between areas under different levels of protection. Prediction: the greatest proportional population increases will be in protected areas among species more heavily exploited and with more sedentary adults.
2. **The biological diversity will be correlated with the level of protection.** Alternative: there is no relationship between biological diversity and levels of protection.
3. **Reserve size is important.** Alternative: Reserve effectiveness has no relationship to its size. Prediction: larger species and species with greater areas of activity will require larger protected areas. Monitoring results will help predict what minimum reserve size is necessary to be effective.
4. **Reserve location is important.** Alternative: the area selected has no effect on response. Prediction: zone effectiveness will be influenced by habitat quality, distribution, quantity, and proximity of nursery habitats.
5. **Reserves will serve as scientific controls to assess fishery impacts on ecosystems.** Alternative: Reserve areas will be ineffective by being no different from exploited areas.
6. **Florida Bay restoration will result in greater responses in areas adjacent to Florida Bay.** Alternative: Areas adjacent to Florida Bay and reefs exposed to Florida Bay water plumes will be no different than more distant reefs. Prediction: increased juvenile recruitment and changed species composition at reefs near Florida Bay.

Table 1. Marine reserve hypotheses being tested from reef fish monitoring data.

(a). **Exploitation response.** Abundance of exploited populations will increase within reserves.
(b). **Unexploited population response.** Abundance of unexploited species will remain the same or decrease in response to an increased abundance of exploited species.
(c). **Quantitative exploitation response.** The greatest proportional population increases will be among the more heavily over fished reef fish species.
(d). **Catchability.** Marine reserves will be more effective for species with higher catchabilities.
(e). **Size structure.** Mean size and age will increase for species benefiting from reserves.
(f). **Spawning potential.** Spawning potential of exploited populations will increase.
(g). **Home range size.** Population increases in marine reserves will be greatest for species with sedentary adults.
(h). **Species size.** Larger species will require larger reserves because of larger areas of activity.

(i). **Dispersal distance.** Increased fishery landings due to postsettlement dispersal will be directly correlated with distance from reserve sites.

(j). **Total area effect.** Total fishery landings for exploited species will increase with the amount of area under protection up to a point and then decrease.

(k). **Edge effect.** Dispersal from reserves to surrounding areas will be proportional to the ratio of reserve boundary length to area enclosed.

(l). **Local area effects.** Population responses in a reserve will be proportional to the protected area of that reserve.

(m). **Threshold response.** Reserves must be of a minimum size to protect a sufficient population to be effective.

(n). **Fecundity.** Reserves will improve regional recruitment on impacted species by increasing recruit abundance.

Sampling Methods

Fish surveys are conducted throughout the reef tract using three methods. The primary method uses a stationary diver centered in a circular plot to collect data on all observed fishes. The second uses a 15 min swimming search to determine presence, abundance and size of exploited species. The third method uses volunteers to independently assess presence, frequency-of-occurrence, and abundance of all observable fishes.

Stationary Plot Sampling (SPS) is a plot technique described by Bohnsack and Bannerot (1986). At random points, SCUBA divers settle to the bottom and list all species observed in 5 min within an imaginary 7.5 m radius cylinder around the diver. Species are listed while rotating in one direction and scanning the field of view. The radius can be estimated or measured either by trailing a 7.5 m fiberglass tape behind a diver or by using a meter stick carried for size estimation. Near the end of the 5 min period, special attention is paid to detecting small, cryptic species within a couple meters of the diver. After the initial 5 min listing, data on abundance, minimum, mean, and maximum size are recorded for each listed species. An exception is made for a few highly mobile species that experience indicates must be counted when first observed in the sampling cylinder because they are unlikely to remain in the area for later counting. Divers systematically work up the list from the bottom to avoid overlooking a species and to avoid bias caused by a tendency to count a species when it is particularly conspicuous or abundant. Species observed after the 5 min listing period are merely listed. This procedure forces actual counts for each species to be made at random times although delayed after the initial observation. When large schools are present, fish are counted by 10's, 20's, 50's, or even 100's. Fish fork lengths (cm) are estimated by using a 30 cm ruler held out perpendicularly at the end of a 1 m rod. Length data are converted to biomass using empirically derived conversion formulae (Bohnsack and Harper 1988).

Habitats are characterized within each sample area by estimating the maximum vertical relief and the percentage cover for various bottom types from the observer's polar-centric viewpoint. Water temperature, underwater visibility, and sea and weather conditions are also recorded. The number of observed fishing items, such as leaders, hooks, and sinkers, is used as an index of fishing effort and normally ranges from 0 to 4 items per plot.

Most study reefs are being sampled annually except for a few being sampled seasonally. A total of 33 reefs are in marine reserves, 39 are nearby reference reefs with fishing, and additional reefs are in the Tortugas region. The minimum daily goal is twelve plots per reef. Adequate sample size for estimating mean sample abundance and mean number of species was determined according to Bros and Cowell (1987) in which a Monte Carlo procedure is used to draw multiple samples for a range of sample sizes

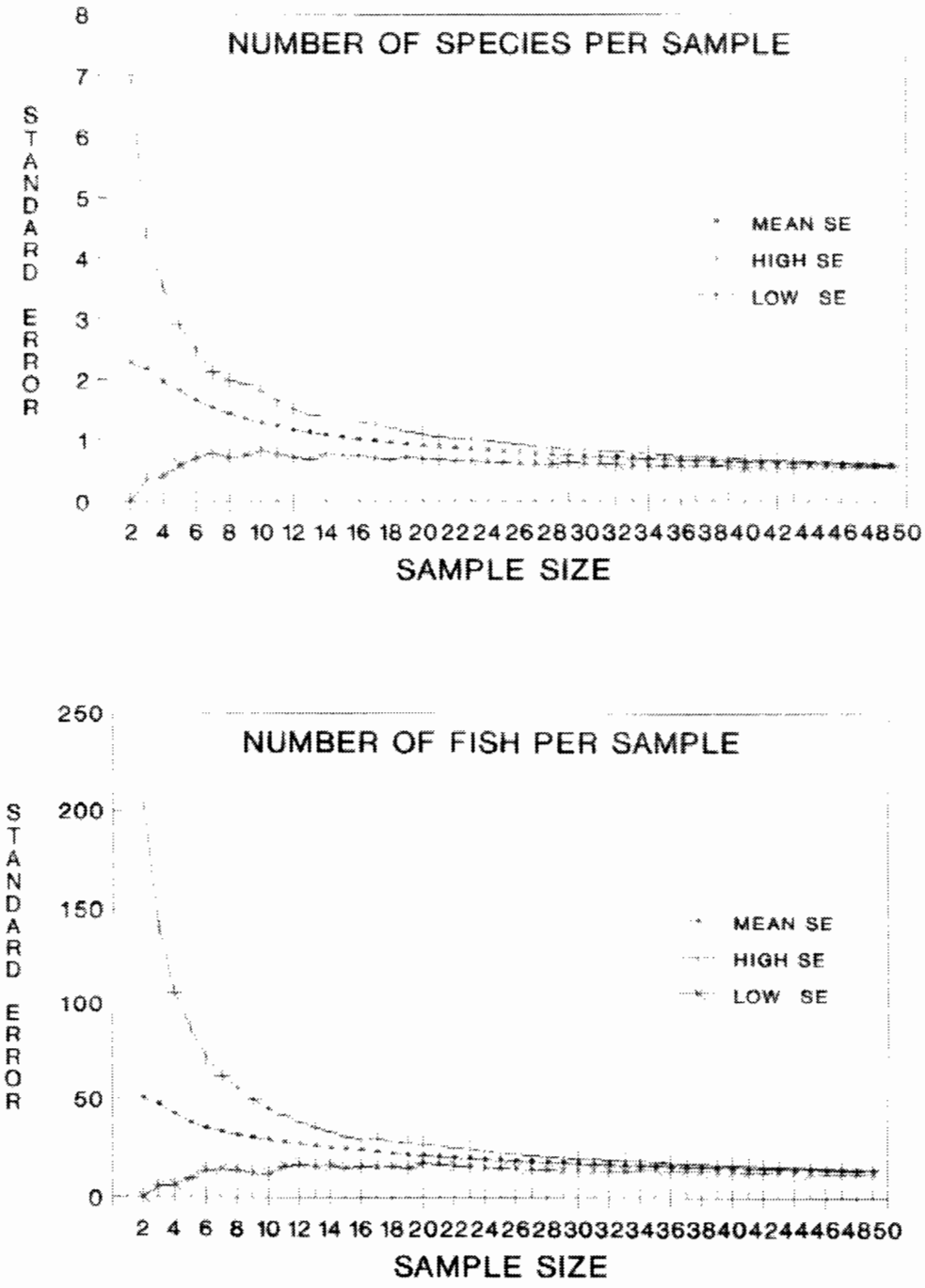

Figure 3. Power analysis of the estimated number of samples needed (alpha = 0.05) to detect changes in mean fish abundance and number of species per sample over a range of accuracies from 10% to 60% of the reported mean.

Table 2. Comparison of data collection and derived parameters possible from different visual fish assessment methods.

	Stationary Plots	Belt Transects	15 min Search	Roving Diver
DATA COLLECTED:				
• Species Presence	All	Some	Piscivores	All
• Abundance	✓	✓	✓	log 10 score
• Frequency-of-Occurrence	✓	depends on area	✓	✓
• Fish Length	✓	✓	✓	NO
• Benthos (% Cover)	✓	*2	NO	NO
CALCULATED PARAMETERS:				
• Species Richness	✓	NO	Limited	✓
• Diversity (H', S, etc.)	✓	NO	NO	?
• Density	Relative / Plot	Absolute	NO	NO
• Total Biomass	Index / Plot	NO	NO	NO
• Comprehensive Sp. List	Good	Poor	Poor	Excellent
SAMPLE CHARACTERS:				
• Area	Fixed by radius	Fixed (1 * w)	Variable	Variable
• Time Searched	5 min	Variable	15 min	Variable
• Avg Time / Sample	20 min (5-30)	Variable	15 min	60 min
• Benthic Composition	% Viewed Area	*2	NO	NO
COVERAGE ADEQUACY:				
• Poster Species	✓	✓		✓
• Cryptic Species	✓			✓
• Shy / Secretive Species	✓			✓
• Highly Mobile Species	✓			✓
• Juvenile Recruits	✓	✓		✓
• Rare Species	✓			✓
• Exploited Species	✓		✓	✓
EFFICIENCY FACTORS:				
• Air Consumption	Excellent	Average	Average	Good
• Equipment Needed	Minimum	More	Minimum	Minimum
• Equip. Handling Time	Minimum	More	Minimum	Minimum
• Total Data Collection	Excellent	Average	Average	Low
• Samples / Dive (20 ft)	4 - 6	*1	1	1
POTENTIAL BIAS:				
• Observer Movement	Limited	✓	✓	✓
• Height Above Bottom	Restricted	✓	✓	✓
• Swimming Speed	na	✓	✓	✓
• Search Width	✓	✓	✓	✓
• Search Methods	Limited	Limited	✓	✓
• Habitat Variation in Sample	Variable	*1	✓	✓
• Edge Effects	Minimum	✓	✓	✓

Footnotes: *1 Depends on length and width., *Optional

from a preliminary data set. The standard error of the mean was used to measure resolving power by graphing the mean standard error along with estimated maximum and minimum standard errors for each sample size (Fig. 3). As sample size increases, the range between the maximum, minimum, and mean standard errors decreases and levels off.

The **15 min Swim Search (15 SS)** targets exploited species and is intended to detect the more secretive and rarer species, such as grouper (Bohnsack 1982). Divers have 15 minutes to tally all the piscivores that they can find, as well as spiny lobster (*Panulirus argus*) and the hogfish (*Lachnolaimus maximus*). Divers can move freely and search as they wish. Caves, ledges and other structures can be closely examined. Abundance and the mean, maximum, and minimum sizes are recorded for each species. Swimming counts are usually conducted between the second and third stationary samples on a dive, providing a ratio of 1 swimming sample per 3 SPS samples.

The **Roving Diver** method, described by Schmitt and Sullivan (1996), uses data collected by trained Reef Environmental Education Foundation, Inc. (REEF) volunteers as a Level III monitoring activity. On each dive, divers list all species that they can find and provide an abundance score using log10 categories: 1, 2-10, 11-100, or >100. Total dive time, depth, temperature and other environmental information is recorded. Data provide measures of geographical and temporal species distributions, frequency of occurrence, and an abundance score. Relative abundance for each species at a site is calculated by multiplying its average abundance score by percent frequency of occurrence.

Discussion

The hierarchal monitoring approach for benthos and fishes in the FKNMS is designed to detect and assess changes for management purposes across disciplines. Sanctuary coordination ensures that: 1) the parameters measured are relevant for both scientific and management purposes and 2) the same sites are being monitored so that benthic and fish monitoring compliment each other. In addition an extensive water quality monitoring program is being conducted. Spiny lobster and conch monitoring is being done by Florida's Department of Environmental Protection. Finfish monitoring is being done by NOAA Fisheries and extends beyond the boundaries of the FKNMS to include all the Keys.

The current experimental design for the FKNMS was established to detect change in time and space and to test for effects of increased protection since July 1, 1997. To examine effects of marine reserves, protected reefs are compared to similar nearby fished reference sites. In the Sambo ecological reserve, for example, replicated bank reefs and inshore and offshore patch reefs are each compared to similar fished reefs to the east and west. We consider the no-take SPAs and the ER "controls," even though we expect them to change initially, because they have minimum direct extractive exploitation. Areas with different levels of fishing are considered the "treatment" sites and will be compared to the control sites over time. The number of replicated sites adds considerable power to the research. Depth and associated habitat data recorded for each site allow depth and habitat effects to be analyzed.

Advantages and Disadvantages of Fish Sampling Methods

Attributes, advantages, and disadvantages of the different fish sampling methods are compared (Table 2). As with all visual methods, sampling is restricted to days with adequate water visibility and safe sea conditions. Each method is ideal for certain questions or limitations. The three fish monitoring methods are intended to compliment each other although the stationary plot method is the core of the monitoring effort.

Stationary Plots NMFS divers using this method have characterized fish communities over 20 yr in different habitats (Bohnsack et al. 1987). They have collected data on over 255 fish species in approximately 6,000 stationary samples from the Florida Keys (Ault et al. 1998). The method is being used widely around the world. The main assumptions are that diver observations objectively reflect the relative abundance of individual fishes and that the presence (or absence) of species tells something about the environment.

ADVANTAGES: All observable species are included. Species richness is rapidly accumulated since over 20 species per sample are usually observed in coral reefs. Data are collected for the three important ecological parameters needed to characterize community structure: abundance, frequency, and size. The method is rigorous and provides an objective index of abundance, frequency-of-occurrence, size, and biomass for all observable fishes. The simultaneous collection of fish and habitat information can be used to describe microhabitat influence on community structure and ecological factors that operate on the scale of the plot, such as ecotone effects. The method offers a standard approach for assessing any marine habitat. Large sample sizes are quickly and easily collected adding considerable statistical value and power, especially for highly variable systems (Goodall 1970). Goodall (1970) noted that in high variance systems, efforts to increase sample size are more likely to reduce variance than efforts to increase precision. The plot size makes frequency-of-occurrence an especially good statistical parameter because values range continuously from 0 to near 100%. Confidence intervals for each species can easily be applied based on sample size. The method is well suited for comparisons of different habitats, the same habitat over time and space, and between species with similar visual signatures (e.g., grunts, wrasses, etc.).

Observer bias is minimized by being stationary. Instead of the observer swimming and counting fishes, the plot method lets just the fish swim. Our experience indicates that by staying still an observer sees more than by moving around. Fish quickly habituate to the diver, become more apparent, and easily tallied. After a few minutes jawfish come out of burrows, squirrelfish come out of caves, and other fishes return to normal activity. The circular plot also minimizes edge effects for the amount of area sampled and empirical experiments have shown that visibility over a wide range has relatively little impact on the data (Bohnsack and Bannerot 1986).

Data collection efficiency is high. By segregating different tasks, a considerable amount of data and a large number of samples are collected in a short time. Time is not wasted in handling gear or waiting for fish to habituate. Because the diver spends relatively little time moving, air consumption is low.

PROBLEMS: The 7.5 m radius was selected to be sensitive to detecting exploited species. Some investigators have reduced this radius slightly for particular reasons. Bortone et al. (1989), for example, used a 5.67 m radius to provide a sample area of 100 m and Polunin and Roberts (1993) used a 5 m radius. Although the data collected for most species are somewhat insensitive to changes in radius (Bohnsack and Bannerot 1986), comparisons between these different data sets should be done cautiously and ideally with appropriate adjustments or calibration. Also, before comparing samples collected under extremely different visibility, the data should be calibrated or adjusted, perhaps through rarefaction techniques.

Extending the species listing time is strongly discouraged because it greatly inflates the number of highly mobile species. Eventually, every mobile species with a large home range will be detected distorting their importance. The 5 min search was selected as a compromise between having sufficient time to do an adequate visual search of the surrounding area and allowing fishes to habituate to the diver, versus inflating the sample with more mobile species.

DISADVANTAGES: Because the observed abundance for many species is likely to be underestimated (Bohnsack and Bannerot 1986), raw data provide an index of abundance and biomass; not absolute density. Thus, density comparisons should be done on a relative basis per sample. Calculating absolute density requires correction and calibration for each habitat. Appropriate calibration studies can be done, but are difficult, require destructive sampling, and must be done for each habitat. Fortunately, absolute abundances or densities are not required for most purposes. Comparing abundances between families with very different visual signatures makes little sense (e.g., gobies with snapper). Abundances of small cryptic species likely have the most error. Other disadvantages are that divers must be able to identify all fish species observed and the method is mentally demanding and too tedious for most volunteers.

Belt Transects Although considered, belt transects were not chosen for monitoring fishes in the FKNMS. A comparison of their advantages and disadvantages is provided because they are perhaps the most widely used visual survey method (Sale 1997).

ADVANTAGES: The method estimates absolute abundance and density for species covered. Size can be estimated if a limited number of species are sampled. It is well suited for small fishes, such as damselfishes, and young recruits, if narrow transects are used (Sale and Sharp 1983). It is well suited for homogeneous habitats and sedentary species.

PROBLEMS: There is no standardization of transect length, width, and orientation. It is recognized that accuracy drops with increased width, although problems with the viewing distance along the transect tend to be ignored. Frequency data from long transects are not very useful because many species will have near 100% occurrence. Data variability is averaged out for transects that cross heterogeneous habitats and relating fish data to microhabitat features along the transect can be difficult. Some patch reefs are too small for transects.

DISADVANTAGES: Perhaps the biggest problem with belt transects is a lack of standardized methodology which makes comparing different studies problematic. Sale and Sharp (1983) describe problems including their variable belt length and width, height of the observer above the transect, distance ahead being viewed, speed of observer, edge effects, and protocols for recording fishes crossing the transect. How transect lines are laid can add considerable bias, particularly when habitats are highly heterogeneous or structured.

In rich environments, only a limited number of species can be effectively sampled per swim and that number drops if size data are collected. Thus, total community structure can not be easily examined. Searching simultaneously for small and large fishes is difficult because of their different search images. The method tends to focus on bottom fishes and misses midwater species.

Considerable time is used to lay, retrieve, and wait for fish to habituate to transect lines; time not available to collect data when underwater time is limited. Air consumption is high because the diver keeps moving and must work to lay, pick up, and move transect lines.

The moving diver and transect line add sampling bias since some species are attracted to the line and diver, while others are repelled. Yellowtail (*Ocurus crysurus*), for example, tend to swarm around a moving diver and some butterflyfishes (Chaetodon) tend to concentrate around a transect line, apparently feeding on disturbed mucous. In contrast, grouper and some secretive species hide in the presence of a moving diver. Finally in heterogeneous habitats, long transects tend to average out effects of

different habitats and thus fail to adequately capture the importance of habitat variability (e.g., reef ecotones, structural complexity, etc.).

Timed (15 min) swimming counts

ADVANTAGES: The method provides reasonable statistical data for top predators, including secretive and less frequently encountered fishes, such as grouper. Little equipment is required. Statistically, the data collected are desirable because they tend to have relatively few zero occurrences and tend to be parametrically distributed compared to other methods. Good size estimates and size distributions can be provided for rarer species. The swimming search provides three secondary functions: it allows observers to have a mental rest between demanding SPS samples; it allows divers to warm up in cool water; and it allows observers to gain a better knowledge of the reef structure and layout.

DISADVANTAGES: Data collection is severely biased by differences between observers in swimming speed, search modes, knowledge of reef layout, visibility, search area, and search patterns. Only a few species are sampled. Differences in visibility and currents can greatly reduce or increase the area searched. It is virtually impossible to judge distance while moving, especially in open water. Divers become familiar with the area and tend to optimize search patterns for a particular site: there is no such thing as a random swim. Different swimming speeds, different heights above the bottom, and different search methods provide unknown biases. Estimated abundance for the most common predators (usually snapper) are at best crude.

Roving Diver Searches

ADVANTAGES: Volunteers using the REEF protocol collect important supplementary data on fish frequency, abundance, and distributions. These data provide an independent, although less precise, assessment of population trends. It is ideal for volunteers because it is not tedious and keeps them interested. The method provides the quickest and most comprehensive species listing. Cluster analyses based on presence-absence data have been shown to distinguish reefs (Schmitt and Sullivan 1996). The advantage of the REEF program is the wide spatial coverage and intensity of sampling. Rare and inconspicuous species are much more likely to be detected and the large number of samples provides excellent estimates of frequency of occurrence with narrow confidence limits. REEF data are accessible on the Web (www.reef.org) and will be used to independently verify trends noted in other more formal data collected by scientists.

Volunteer monitoring is an essential part of the FKNMS public education and awareness effort and helps communicate trends to the public and to managers. Many volunteers are well trained and experienced and have proven helpful to scientists and managers by alerting them to changes, problems and observations that otherwise would go unnoticed. As volunteers become more knowledgeable, they are more likely to support management efforts.

DISADVANTAGES: Search methods and total search area vary considerably between divers. Data collected are sensitive to different skill levels and only well trained and experienced volunteers can be reliably used. Biomass and size data are not collected. Divers tend to learn habitat on a particular reef becoming more efficient. Non-experts can collect significantly biased data through misidentifications. Abundance scores are crude.

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Chapter 11:

The Use of Indicator Species for Coral Reef Monitoring

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Abstract

Indicator species are used successfully in the assessment of ecological conditions in both aquatic and terrestrial environments. Only recently, however, has the indicator species method been applied to coral reef ecosystems. This is unfortunate as the method has many advantages over conventional monitoring methods that have dominated coral reef research.

Conventional monitoring programs are often expensive, time consuming, require scientifically trained personnel, involve complex sampling schedules, and require the processing, storage and analysis of samples. Photographic sampling methods, which at first glance appear to be an excellent choice, are expensive and analysis is extremely time consuming. Furthermore, permanent transecting lines or pins may not be permitted in preserves and sanctuaries.

In contrast, the use of indicator species has many advantages. It is less expensive and does not require scientifically trained personnel. It eliminates the costs in time and money of establishing a conventional sampling plan, including the archiving, storage and analysis of samples. It is not necessary to leave permanent markers in the field.

The use of indicator species to monitor coral reefs may be the method of choice when gradual change is suspected due to low but chronic input of pollutants into a reef area, where previous baseline data do not exist, when funds are limited and scientifically trained personnel are not available. These conditions exist in many developing nations where coral reefs are an important resource and there is a need for conservation and management.

Choosing the best one or two species to serve as indicators is extremely important. It necessitates knowledge of the life history and behavioral ecology of species which are abundant, conspicuous, diurnal, long lived, site attached, and dependent on the ecosystem to be monitored for their food and shelter. Fortunately this information often exists for one area where the species was studied and can be applied effectively elsewhere. Thus, for example, it is possible to recommend species of butterflyfishes as potential indicator species for many coral reefs in the Indian and Pacific oceans based on studies in Hawai'i, Guam, Australia, the Marshall Islands, and French Polynesia.

Keywords

reef fish, indicator species, butterflyfish, coral cover, Hawai'i

Introduction

The concept of using one or a small group of species to indicate the condition of a larger community or ecosystem is not new. Hunting and gathering peoples, both past and present, intuitively apply the idea (Lee and Devore 1968), and similarly early naturalist/conservationists like Aldo Leopold recognized the usefulness of the concept (Leopold 1966). A recent example of the utility of the concept in the terrestrial environment is the finding that the decline of migratory songbird populations is an indication of degraded and hostile forests across the continental United States (Askins 1995).

To our knowledge, the concept of indicator species first was applied formally to marine organisms by the celebrated British marine biologist F.S. Russell in the 1930s (Fraser 1962; Raymont 1963). In a series of papers, Russell showed that certain species of zooplanktors were indicative of water masses which, in turn, contained commercially important species of fish. Russell emphasized that a good indicator species must be easily recognized and its distribution must characterize a particular mass of seawater.

Recently two edited volumes address the use of marine organisms as indicators for detecting ecological impacts, usually from human activities, on marine ecosystems (Soule and Kleppel 1988; Schmitt and Osenberg 1996). Each contains chapters on coral reefs (Hourigan et al. 1988; Jones and Kaly 1996), which are discussed below.

Chaetodontid Fishes and Coral Reefs

The seminal paper by Odum and Odum (1953) clearly established coral reefs as communities and with the addition of energy flow the concept of the coral reef ecosystem was born. Surprisingly the idea of coevolution, in the broad sense, between corals and coral feeding fishes was not widely recognized (Reese 1977) nor was the possibility recognized that this relationship might prove useful for the conservation and management of coral reef ecosystems (Reese 1981). The idea that the distribution, abundance and behavior of coral feeding fishes of the Family Chaetodontidae, the butterflyfishes, could be used as indicators of environmental stress on coral reefs was established by Hourigan et al. in 1988.

There are a number of studies where entire assemblages of chaetodontid fishes are correlated with coral cover (Sano et al. 1984; 1987; Bell, Harmelin-Vivien and Galzin 1985; Bouchon-Navaro, Bouchon and Harmelin-Vivien 1985; Findley and Findley 1989; Williams 1986; Roberts and Ormond 1987; Roberts, Ormond and Shepherd 1988; White 1988; Cadoret et al. 1998). Although generally there is a good correlation between chaetodon abundance and coral cover, the results are sometimes mixed and do not always support the thesis that chaetodontid fishes can be used as indicators of coral cover. There are two

primary reasons for these mixed results. First, entire assemblages of butterflyfishes were counted rather than just the corallivorous species. Second, between reef comparisons were not consistent because the stochastic processes relating to both coral and fish recruitment and survival vary independently of one another both within sites and between sites. Thus, although between-site comparisons may be instructive when used judiciously, the indicator species concept is most useful, sensitive and accurate when within-site comparisons are made over time.

Refinement of the Method to Corallivore Species

When coral feeding species are studied at one location over time, the results become clearer. For example, at Moorea, French Polynesia, the obligate corallivores *Chaetodon trifascialis* and *C. trifasciatus* were abundant on the reefs in 1979, but disappeared completely after intense predation by the crown-of-thorns starfish *Acanthaster planci* in 1981. They were replaced by the algal browser *Chaetodon citrinellus*. This example clearly illustrates the danger of including non-corallivore species as indicators of corals (Bouchon-Navaro, Bouchon and Harmelin-Vivien 1985). In fact, this case demonstrates an inverse relationship between algal feeders and condition of coral reefs as a function of abundance of species.

Although Roberts et. al (1988) are critical of the methodology, when they plotted the abundance of three species of obligate corallivores in the Red Sea against hard coral cover, they obtain a correlation of 0.8. They agree, however, the repeated censuses at the same sites over time would be useful in assessing changes in coral reef communities. White (1988) provides further support. A total of 34 species of Chaetodontid fishes were counted at eight locations. Only about a third of these species can be classed as coral feeders, both facultative and obligate, but nevertheless when the number of species was plotted against hard (scleractinian) coral cover, the correlation was 0.71. More recently Jones and Kaly (1996) showed graphically a high correlation between the obligate corallivores *Chaetodon trifascialis* and *C. trifasciatus* and total percent coral cover in the Funafuti lagoon, Tuvalu. Interestingly, they found the same good correlation between algal cover and *Chaetodon citrinellus* and *C. auriga*, which again illustrates the danger in counting all species of chaetodontids. Nevertheless these authors miss the point of using a mobile indicator species and suggest simply measuring coral cover. Finally, Erdmann (1997) offers a thoughtful critique of the butterflyfish indicator method as developed by Crosby and Reese (1996) and points out a number of areas where caution is needed. We agree but believe we have addressed these criticisms in our research at Kaho'olawe Island, Hawai'i, and in a series of forthcoming publications. As an alternative to butterflyfishes, stomatopods are suggested as bioindicators of marine pollution stress on coral reefs (Erdmann and Caldwell 1997) but most of the criticisms of our approach are applicable to theirs as well, and going back to F.S. Russell's comment, it is easier to count brightly colored relatively large butterflyfishes than cryptic, cavity-dwelling stomatopods. The choice is up to the researcher, public agency, environmental group or individual volunteer.

There are always some unknowns and problems of interpretation. Although recruitment processes may vary from reef to reef, the long life of chaetodontid fishes (Reese 1991) and of corals leads us to conclude that there is a causal, synergistic relationship between the butterflyfish and the corals on which they feed. Our own findings as well as those reported in the literature (see above) support this assumption.

The Kaho'olawe, Hawai'i, Study

The island of Kaho'olawe (156° 35'W Longitude and 20° 35'N Latitude) is approximately 18 km long and 10 km wide comprising an area of 116.5 km^2. Although it is uninhabited and is the smallest of the eight main Hawaiian Islands, it is of cultural importance to the native Hawaiian people. For many years

it was used as a bombing target by the military. This activity along with a population of feral goats severely damaged the vegetation and caused extensive soil erosion. The resulting sedimentation damaged the shallow coral reefs especially along the northwest shore of the island. In 1990 the bombing was stopped and by 1992 the feral goats were eliminated. The State of Hawai'i took over jurisdiction of Kaho'olawe in 1993 and a program of restoration of the island was begun.

During the five year period 1992-1997, we were fortunate to have the opportunity to field test the use of a single species of butterflyfish, *Chaetodon multicinctus*, as an indicator of ecological conditions at two reef sites at Kaho'olawe. One site, Hakioawa, is pristine and beautiful with high coral cover, a good assemblage of butterflyfishes and relatively little sedimentation. We consider this our control or reference site. The other site, Kuheia, is heavily impacted by terrigenous sediments due to soil erosion. We consider Kuheia as our experimental disturbed site. We are testing the methodology described in a training manual and video (Crosby and Reese 1996) which are available on request from the authors and the manual is on our Website at http://coral.aoml.noaa.gov/themes/.

There are two unique aspects to the Kaho'olawe study. First, during the nearly 50 years that the island was under the jurisdiction of the U.S. military, the nearshore areas were closed to boating, diving, fishing and related activities. This means that reef areas such as our site at Hakioawa are undisturbed, while the source of the disturbance at impacted reef areas such as our Kuheia site is known to be sedimentation. It is rare that an opportunity for research such as this presents itself, and we feel very fortunate that we received permission from the Kaho'olawe Island Reserve Commission, the Hawai'i State agency with jurisdiction for the island, to conduct our research at these two coral reef sites at Kaho'olawe.

The second aspect is more complex. In concept most monitoring programs start by looking at areas where damage to coral reefs is anticipated or suspected usually due to human activities such as channeling streams, building resorts and golf courses, or constructing boat harbors. Thus, the monitoring program is put in place to determine whether or not, and to what extent, degradation of the reef area is occurring.

At the Kuheia site the opposite situation pertains. The Kuheia coral reefs are already degraded from sedimentation, so our monitoring program aims to record the gradual restoration of the reefs as the vegetation on the island slowly recovers, the sediment is flushed away, and the biota of the reef area is replenished through the process of larval recruitment.

This is an important difference and it necessitates some readjustments in our thinking about the application of the indicator method. For example, it means that fish size becomes more important. Why? Because in the more conventional scenario, fish sizes, territory sizes, and agonistic encounter rates as well as the numbers of coral feeding butterflyfishes relative to the percent coral cover are postulated to change in such a manner as to indicate a deteriorating condition, or alternatively, not to change and be indicative of non-significant impacts from the perceived perturbation(s) at the experimental site. If, for instance, an area is impacted adversely, the sizes of fish do not change, but agonistic encounter rates and aggressive chasing increase as the fish attempt to enlarge their territories due to the decrease in the quality and abundance of their coral food resource. Similar changes should not occur at the control site.

At the Kuheia site, however, we postulate that fish abundance and size should increase due to recruitment and growth, and agonistic encounter rates should be higher than at the stable Hakioawa site as the fish seek to establish territories and enlarge them as they grow in size. And territory sizes should increase. But the stochastic nature of the recruitment, survival and growth of both the fish and corals must be considered carefully. These processes vary independently of one another. Thus, for example, there

may be good recruitment of juvenile *Chaetodon multicinctus*, our indicator species, in one year at the Kuheia site and there is an increase in their numbers, while the recruitment, survivorship and growth of corals, their food source, may be poor in the same year. For this reason, we stress the importance of change occurring within sites as opposed to making between site comparisons.

For these reasons, it is clear that the situation at Kaho'olawe provides a wonderful opportunity to field test the indicator species method on the one hand, but at the same time presents some interesting complexities of interpretation.

Evaluation of Indicator Species Monitoring

The task for each participant in this workshop is to discuss the advantages and disadvantages of the monitoring method with which they have had the most experience. We are not to present our data.

Advantages

Learning to recognize one or two brightly colored butterflyfishes and to observe their behavior, and to learn to recognize growth forms of corals are easy tasks. It is not necessary to learn scientific names or to have scientifically trained personnel. It is easy to train volunteers in one day to learn to collect the necessary data. In fact we encourage the use of volunteers because the training requirements are minimal, the data collection is easy, and therefore there is less chance for error or inexperienced volunteer bias than in more complicated and therefore demanding and learning intensive monitoring techniques.

Although it is convenient to work from a boat and SCUBA is necessary for deeper reefs, it is possible to apply the method to shallow nearshore reefs by snorkeling. Costs are minimal. The necessary equipment, as described in the manual (Crosby and Reese 1996), include one or more 30m by 1cm brightly colored polypropylene transect lines with colored surveyor's tape tied through the weave at 1 m intervals, inexpensive clipboards or writing slates with regular soft lead No. 1 pencils. Writing slates or plastic overlays of the data sheets may be copied on any copier, erased and reused, or underwater writing paper, such as Xerox Never-Tear plastic paper, may be used, but this will add to the cost (a pack of 100 sheets costs about $50). This plastic paper is very durable, however, and can be erased and reused and it goes through most copying machines. Simple carpenter's nails and surveyor's tape are needed to mark the territory boundaries, and fiberglass measuring tapes of 30-50m (cost about $35-$45), needed to measure the territory size, are available at most hardware stores. It is essential to have an underwater watch to time the various activities and brands like Casio are reliable and inexpensive (about $35).

Sample data sheets are shown in the manual and can be modified to meet the requirements of the particular monitoring site to depict the indicator species being used and other species of corallivorous butterflyfishes as well as the most common coral growth forms. It is necessary to have access to a computer and appropriate statistical packages. We use data spread sheets created using the Microsoft Excel program. Details of the data management procedures we use are available in our manual (Crosby and Reese 1996).

Another advantage of the method is that it can be used in an incremental, stepwise manner with each step providing more data and increasing accuracy and sensitivity to the predictive strength of the method. In the simplest first step, only the fish and coral counts are made along the transects. The second step is marking the territory boundaries and recording agonistic chasing events (it is convenient to do these two parameters at the same time as described in the manual). The third step is measuring the sizes of the territories. The final fourth step is measuring feeding rates. The complete data set provides a wealth of

quantitative data about the behavioral ecology of the indicator species and the overall condition of the coral reef.

It goes without saying that operational costs increase with the use of SCUBA and boats, but these costs are going to be the same for all monitoring procedures.

Further strengths or advantages of the method are that it directly addresses management issues and concerns where a previous monitoring data base does not exist but gradual, low level, chronic disturbances to the reef area are expected. The method is immediately available for use and it is sensitive to changes occurring on the reef before the corals become obviously moribund and begin to die. We are pleased with the accuracy, quality and replicability of the data we have collected at Kaho'olawe, Molokini and other locations in Hawai'i. It is especially sensitive, we believe, to small-scale spatial variability, which is often high on coral reefs.

In our judgement, the indicator method should be the method of choice, or at least be a component of any monitoring program, where there are management issues and concerns about suspected gradual changes in the ecological conditions of the coral reef biota.

Disadvantages

Clearly the indicator method is not useful or appropriate where large episodic events like storms, oil spills, crown-of-thorns, or poor land development practices damage coral reefs. In addition, for the method to be applied effectively coral cover should be relatively high, about 40% in areas where corals are abundant. For example, at the Kuheia site at Kaho'olawe, coral growth is good on the ridges or spurs, ridges or spurs, but there are no corals whatsoever in the sediment clogged depressions or grooves. As a result the transect lines are established on the ridges where both corals and fish are present. There should also be a good assemblage of coral feeding butterflyfishes, and at least one of these species should be relatively common and territorial. Since behavioral observations play an important role in the method, water clarity should be reasonably good, perhaps not less than 10m, and it must not be too rough. Fishes stop "behaving" when it is rough and visibility is poor.

Another disadvantage to the indicator species method is that it only provides data on the abundance, distribution and diversity of coral feeding species of butterflyfishes and of the more common species of corals, if species are being recorded, or the coral growth forms, if they are being used. Therefore, only within these limitations can diversity estimates be made. Furthermore, it is necessary to know to what extent collecting of reef fishes is occurring for the aquarium trade and whether cyanide and other chemicals are in use to collect target species for the fine dining market.

Although it is well established that after a reef is severely degraded or destroyed, populations of coral feeding butterflyfishes are reduced or eliminated altogether (Sano et al. 1984; 1987; Bell, Harmelin-Vivien and Galzin 1985; William 1986), the process of this event remains unknown. As the corals become moribund and eventually die, what do the fishes do? Do the larger more aggressive pairs of butterflyfish enlarge their territories to compensate for the decreasing amount of food at the expense of the smaller less aggressive pairs of fish, which then must either starve or emigrate? Do these strongly territorial and site-attached fish stay and die as their food supply is depleted or do they emigrate from the reef when some food threshold value is reached? Our Kaho'olawe research did not enable us to track these events or provide answers to these questions. To date, experiments both in the field and in the laboratory designed to simulate the gradual decline in the nutrient value of corals as a food resource and to record the resultant behavior of the fish are unsuccessful. This effort, however, is continuing and we

are alert to any opportunity to observe the behavior *in situ* of an indicator species on a reef undergoing the process of gradual degradation.

Conclusions

1. The indicator species method for monitoring the occurrence of change on coral reefs may be the method of choice; (a) when change is suspected due to slow but chronic low levels of pollution, whatever the source, (b) when no prior data base exists, (c) when funds and trained personnel are limited, (d) when volunteers are available, and, of course, (e) when corals are sufficiently abundant to support an existing population of one or more potential indicator species of coral feeding butterflyfish.
2. The indicator species monitoring method is inexpensive, easy to learn, does not involve complicated, expensive, and time consuming sampling protocols with the concomitant problems of sample storage, analysis, and data reduction. With the indicator species method, quantitative data are ready for entry into the computer spread sheet literally as soon as the diver is out of the water.
3. The method can be used in a step-wise incremental application with each step adding increasingly valuable information. The analysis of behavior of the indicator species provides valuable insights into the dynamics of the reef community and may provide a useful "early warning" of the degradation of the coral reef. An added bonus of the method is that volunteers learn to appreciate the subtle workings and interactions of the coral reef ecosystem. This is a benefit of great value in terms of public awareness, involvement and support of conservation and management programs for coral reefs.

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