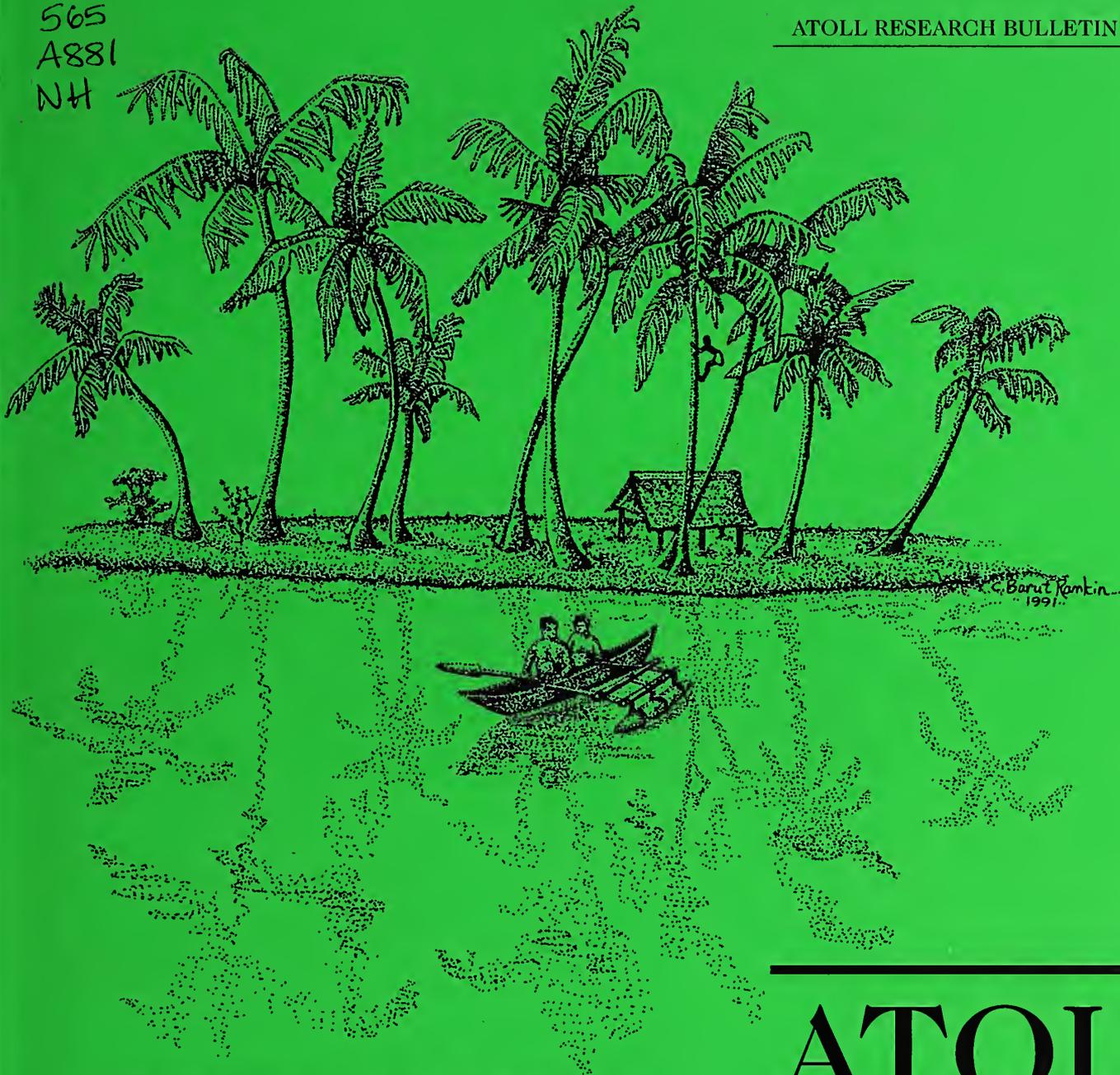


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ATOLL RESEARCH BULLETIN NO. 496



**STATUS OF CORAL REEFS IN  
THE WESTERN ATLANTIC:  
RESULTS OF INITIAL SURVEYS,  
ATLANTIC AND GULF RAPID  
REEF ASSESSMENT (AGRRA) PROGRAM**

Edited by  
**Judith C. Lang**

*Issued by*

**NATIONAL MUSEUM OF NATURAL HISTORY  
SMITHSONIAN INSTITUTION  
WASHINGTON, D.C., U.S.A.  
JULY 2003**

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# ATOLL RESEARCH BULLETIN

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# Atlantic and Gulf Rapid Reef Assessment (AGRRA)

A Joint Program of  
The Rosenstiel School of Marine and Atmospheric Science, University of Miami and  
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**EDITED BY**

**JUDITH C. LANG**

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ATLANTIC AND GULF RAPID REEF ASSESSMENT (AGRRA) PROGRAM**

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Patricia Richards Kramer, Robert N. Ginsburg, and Judith C. Lang

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## FOREWORD

BY

ROBERT N. GINSBURG<sup>1</sup> and JUDITH C. LANG<sup>2</sup>

## INTRODUCTION

The global decline in coral reefs during the last decades has provoked the most serious concerns about these remarkable ecosystems. Were they owing to a single worldwide cause, like influenza, plague or HIV in humans, the focus of efforts to understand and remedy the situation would be clear. Instead, the causes of declines as well as the nature of reefs vary significantly from region to region and within regions. Thus the urgent need is to assess the condition of reefs regionally with directly comparable quantitative observations rather than anecdotal reports.

This volume contains the initial reports and their synthesis of a new approach to assessing the regional condition of coral reefs in the Western Atlantic Ocean developed under the Atlantic and Gulf Rapid Reef Assessment (AGRRA) program. It features rapid, multiscale assessments by teams of five-six trained observers for reefs of the Greater Caribbean, Gulf of Mexico and South Atlantic with the same method. Thus it becomes possible to assess many reefs spread over the entire region.

The AGRRA protocols are focused on three key functional and structural elements of reef ecosystems: stony corals, fish and algae. The long-term goal of this region-wide effort is to provide, for the first time, a database suitable for comparative evaluation of current reef condition. This approach is similar to that which public health officials would use to make a rapid health assessment of villagers in remote areas. Analysis of the assessments provides norms of condition for some 30 key parameters of reef condition, such as the species identity, sizes and partial mortality of reef-building corals and the biomass of ecologically and commercially significant reef fishes. As demonstrated in the Synthesis of this volume, these initial norms, which are like those of human health assessments (blood pressure, pulse, reflexes), can be used to compare individual reefs at different spatial scales: between reefs; groups of reefs; subregions; or for most of the 20 reef areas in this volume. Even these initial results offer valuable background information for selecting protected areas or perturbed reefs in need of monitoring.

While this volume was in preparation, additional AGRRA surveys have been carried out in several other major reef areas: Jamaica's northern and western coasts; Cuba's southern coast; the Caribbean reefs of Panamá and the Florida Keys. The

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completed regional assessment will expand and refine the initial regional interpretations presented in this volume, and produce an essential baseline with which to gauge future major changes disclosed by revisits.

## SPECIAL VALUE OF THESE INITIAL RESULTS

Each of the assessments in this volume is based on data from multiple sites. The areas assessed extend from the northern Bahamas to the southernmost reefs of the Caribbean and from the windward Netherlands Antilles to the Gulf of Mexico. Some areas are of special interest because of significant anthropogenic impacts (e.g., Veracruz, México, Costa Rica and some Virgin Islands sites). Others are notable for the relative scarcity (apart from fishing) of direct human impacts (e.g., Andros in the Bahamas, Flower Garden Banks in the NW Gulf of Mexico and Los Roques, Venezuela); all are affected by adverse regional- and global-scale changes. The inclusion of the southernmost reefs in the Atlantic off Brazil extends the range of areas examined and adds a coral community and reef structure quite different from that of the Greater Caribbean. These assessments were accomplished owing to the major efforts of about a hundred reef observers and team leaders from eight different countries who volunteered to spend long days of arduous diving to collect the basic data.

The results of these first 20 AGRRA surveys demonstrate the promise of this approach for characterizing and comparing reef condition, for distinguishing between regional and local impacts, for recognizing the differences between acute and chronic stressor effects and for identifying candidate reef areas for protection or remediation. Some unanticipated but valuable dividends are the ancillary observations on the distribution of reefs, their general community structure, their geomorphology, and identification of localized threats to reefs. For example, the reef off Andros Island, some 217 km long, was so little known that our general characterization provides the most comprehensive information available on this second longest reef complex in the Western Atlantic.

Kramer (this volume) has contributed an extensive Synthesis of the separate reports. The regional means cited below are based on this standardized analysis of the initial AGRRA dataset.

### Corals

Our results provide quantitative verification of the decline of elkhorn coral (*Acropora palmata*) on reefs crests in Los Roques and the Tobago Cays (St. Vincent). Furthermore, what had been dense arrays of *A. palmata* on reef crests off Providencia in the Turks and Caicos Islands and Grand Cayman (Ginsburg, personal observations) and off Abaco (Bahamas) are now largely standing dead skeletons. Living staghorn corals (*A. cervicornis*) were rare in all areas, including those in which they are known to have occurred historically. That some of the die-offs of sensitive acroporid corals occurred in relatively remote areas reinforces earlier demonstrations that the declines are regional not local (e.g., Aronson and Precht, 2001). Surprisingly, healthy thickets of living elkhorn predominated on the shallow (1-3 m) crest along the Andros (Bahamas) reef tract,

however, and signs of recovery and recruitment in localized areas elsewhere in the region are encouraging.

The averages for living stony coral cover of  $18 \pm 10\%$  in shallow reefs ( $\leq 5$  m) and  $26 \pm 13\%$  in deeper reefs ( $10 \pm 3$  m), which are based on 17 of the assessments, cannot be used as representative of the whole region for at least three reasons: a wide range of percent cover was found within and among reef areas; the means are not adjusted for the relative sizes of the different reef areas; and shallow reefs are under-represented in most areas.

Partial mortality in stony corals represents the cumulative effects of diseases, overgrowth by algae and other epibionts, predation, bleaching, physical abrasion, etc. It can be subdivided by assessors on the basis of skeletal appearance into “recent” ( $< \sim 1$  year) and “old” ( $> \sim 1$  year) mortality. As with coral cover, the regional mean of recent mortality of 4% of colony surfaces for both shallow and deep sites masks significant subregional variations. Recent mortality was well above the regional mean off Andros and in the western Caribbean (Belize, Yucatán) as a result of the cumulative effects of bleaching and diseases associated with the 1998 ENSO-related warming. The most severely affected taxa were *Agaricia tenuifolia* and species of the *Montastraea annularis* complex. Outbreaks of diseases in *M. annularis* and *M. faveolata* were additionally noted off Curaçao, the Cayman Islands, Costa Rica and some of the Virgin Islands. Of special concern is the prevalence of diseases and bleaching-related mortality in the *M. annularis* complex, a major contributor to reef framework throughout the Greater Caribbean.

## Algae and Fishes

Although much attention has been given to the current prominence of macroalgae on many wider Caribbean reefs, in terms of relative abundance, turf algae generally predominated in the AGRRA assessments. In deeper sites, however, elevated values for both macroalgal relative abundance and macroalgal index (a proxy for its biomass) were found throughout the Bahamas (Abaco, Andros and San Salvador) and in María la Gorda (Cuba).

Herbivores affect the types, abundance and biomass of algae found on reefs. Herbivorous fishes [surgeonfishes (acanthurids), parrotfishes (scarids)  $\geq 5$  cm, and the damselfish *Microspathodon chrysurus*] averaged  $\sim 30/100$  m<sup>2</sup> overall on deeper sites in the 17 areas having comparable data. Their density was not related to the relative abundance and index values of macroalgae for the region as a whole. Moreover, the long-spined urchin (*Diadema antillarum*), formerly a key herbivore, was too scarce (regionally  $< 3/100$  m<sup>2</sup>) to have any significant effect on algal distribution patterns.

The total density of reef-associated AGRRA fishes (primarily the “AGRRA herbivores” plus commercially important predators) was nearly twice as high in shallow ( $85/100$  m<sup>2</sup>) as in deeper ( $49/100$  m<sup>2</sup>) sites for the 17 assessments mentioned above. Large-sized parrotfishes, seen mostly in the southern Caribbean, were rare. The overall mean density of large-sized groupers (serranids) and snappers (lutjanids) averaged  $< 1/100$  m<sup>2</sup>. The scarcity of large fishes is an indication that, regardless of location, legal designation, or local fishing regulations, the entire region has been overharvested at least for these species.

## Synopsis

Quantitative historical data with which to compare the present results are lacking for reefs in most of the assessed areas. Where prior information of some form exists, it is clear that only the relatively remote Flower Garden Banks have remained essentially unchanged in recent decades. Everywhere else (San Salvador in the Bahamas, Cayman Islands, Costa Rica, Los Roques, U.S. Virgin Islands, Yucatán) their condition has deteriorated.

Kramer (this volume) shows how 13 of the 30 individual norms of condition can be used to establish a preliminary biotic health index for the 17 deeper assessments having comparable data. The four considered “better” (above average) are two in offshore locations (Flower Garden Banks and Los Roques), and two that are adjacent to small human populations (Bonaire and the windward Netherlands Antilles). In contrast, the six that were grouped in the “worse” (below average) category include sites in two sparsely populated biosphere reserves (Sian Ka’an, México, and Guanahacabibes, Cuba), two of the Bahamian islands (Andros, Abaco), unprotected areas of the Yucatán, México, and Costa Rica’s Cahuita National Park. Given these spatial patterns, no single type of threat, at any scale from localized anthropogenic inputs through regional overfishing and diseases to ENSO events and climate change, seems sufficient to explain the details of presumed or documented declines on reefs that have been assessed to date by the AGRRA protocols.

## DEVELOPMENT OF THE AGRRA PROGRAM

The AGRRA program developed from insight afforded during the 1993 Colloquium and Forum on Global Aspects of Coral Reefs. This meeting, attended by some 120 scientists from 20 different countries, was an early attempt to consider the condition of reefs on a global scale. A major conclusion of the meeting highlighted the insufficiency of available information: “The database for evaluating the condition of the world’s reefs is quite inadequate on all counts,” (Ginsburg and Glynn, 1994).

This conclusion was especially relevant to reefs of the Western Atlantic and Gulf of Mexico. Despite the extensive research on this region’s reefs beginning in the early 1900s, large areas had received little or no attention from reef scientists. [An exception is the ongoing international Caribbean Coastal Marine Productivity (CARICOMP) program, a pioneering effort to monitor reefs, sea grass communities and mangroves at a series of fixed localities around the region (Kjerfve, 1998).] However, it had already been clear for about a decade that reef-building corals, most notably the branching elkhorn and staghorn acroporid corals, were in serious decline at numerous sites in the Greater Caribbean. What was not clear in many areas was which other stony corals were affected.

An initial effort to develop a standard method of assessing reef condition was focused solely on the reef-building coral community and followed the approach of Juan Manuel Díaz and his colleagues (Díaz et al., 1995) to record partial mortality of stony corals. The results demonstrated that a census of corals by species, size, and partial mortality could be done rapidly and provide useful comparisons of the condition of patch reef corals in south Florida (Ginsburg et al., 1996; Ginsburg et al., 2001).

The idea of a region-wide survey of Caribbean reefs was first discussed informally at the 1996 Reef Symposium in Panamá. A major refocusing came later in 1996-1997 when Robert Steneck and Judy Lang proposed adding observations on algal functional groups, fish densities, herbivory, recruitment and the distinction between recent and old partial mortality, and then field-tested the protocols in three different geographic locations. Philip Kramer and Patricia Richards Kramer organized the first extensive field test of this expanded method in the Bahamas along the Andros reef tract in August 1997. It included quantitative fish assessments, following suggestions from Peter Sale, and roving diver surveys that were both conducted by Ken Marks.

The positive results of all these trial applications encouraged us to post the protocols on the Internet and organize an international workshop that was held in Miami in June 1998. Eighty-one scientists from 19 different countries of the Greater Caribbean, Brazil, Canada, the United Kingdom, Austria and the Philippines participated in the five-day session to review and refine the prototype AGRRA protocols and plan their region-wide application. All the participants contributed to the development of the final product through lengthy discussions of the methods and a field trip to test the proposed revisions. The product of this workshop was Version 2 of the AGRRA protocols (see Appendix One, this volume) and most of the reports in this volume were based on that version.

## **TRAINING AND APPLICATION OF THE PROTOCOL**

It was evident from the field trials before and during the 1998 workshop that training was necessary to ensure the consistent application of the AGRRA protocols. Accordingly, Philip Kramer and Patricia Richards Kramer with Christy Pattengill-Semmens and Andrew Bruckner organized the first training workshop in Bonaire. Held in February 1999, for 11 participants, its success encouraged us to conduct a second workshop for reef scientists from Central America in Akumal, Quintana Roo in May, 1999. Philip Kramer, Patricia Richards Kramer, Andrew Bruckner and Elizabeth Fisher helped conduct this five-day bilingual session. The 25 participants came from México (13), Belize (5), Honduras (3), Cuba (2), Costa Rica (1) and Colombia (1).

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## Contributors

Most the individual assessments reported in this volume required many long days of scuba diving and/or snorkeling, usually from small boats operating from shoreside bases, by participants who were all volunteers. Each assessment team consisted normally of six divers, four of whom made transects to evaluate corals and algae, and two who collected information on fish through belt transects and a roving diver census (for a description, see Appendix One, this volume). At the end of each day, team members then had to spend further hours transferring their results to a standard spreadsheet. Their commitment and hard work provided the basic data on which each report is based and for the Synthesis chapter. The leaders of these assessments naturally became the authors and coauthors of these reports. Their patience with, and responses to, the prolonged editorial and database construction process is much appreciated.

The Synthesis chapter was made possible only through the use of an Access database. A major effort was required to ensure the consistency of the data entered in the database. Philip Kramer spent days correcting the initial entries through correspondence with observers. We are fortunate indeed that Kenneth Marks set up and maintained the database in such a way that queries could be answered in short order. Assembling this regional database was only possible owing to the generous sharing of basic data by the team leaders.

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## Advisors

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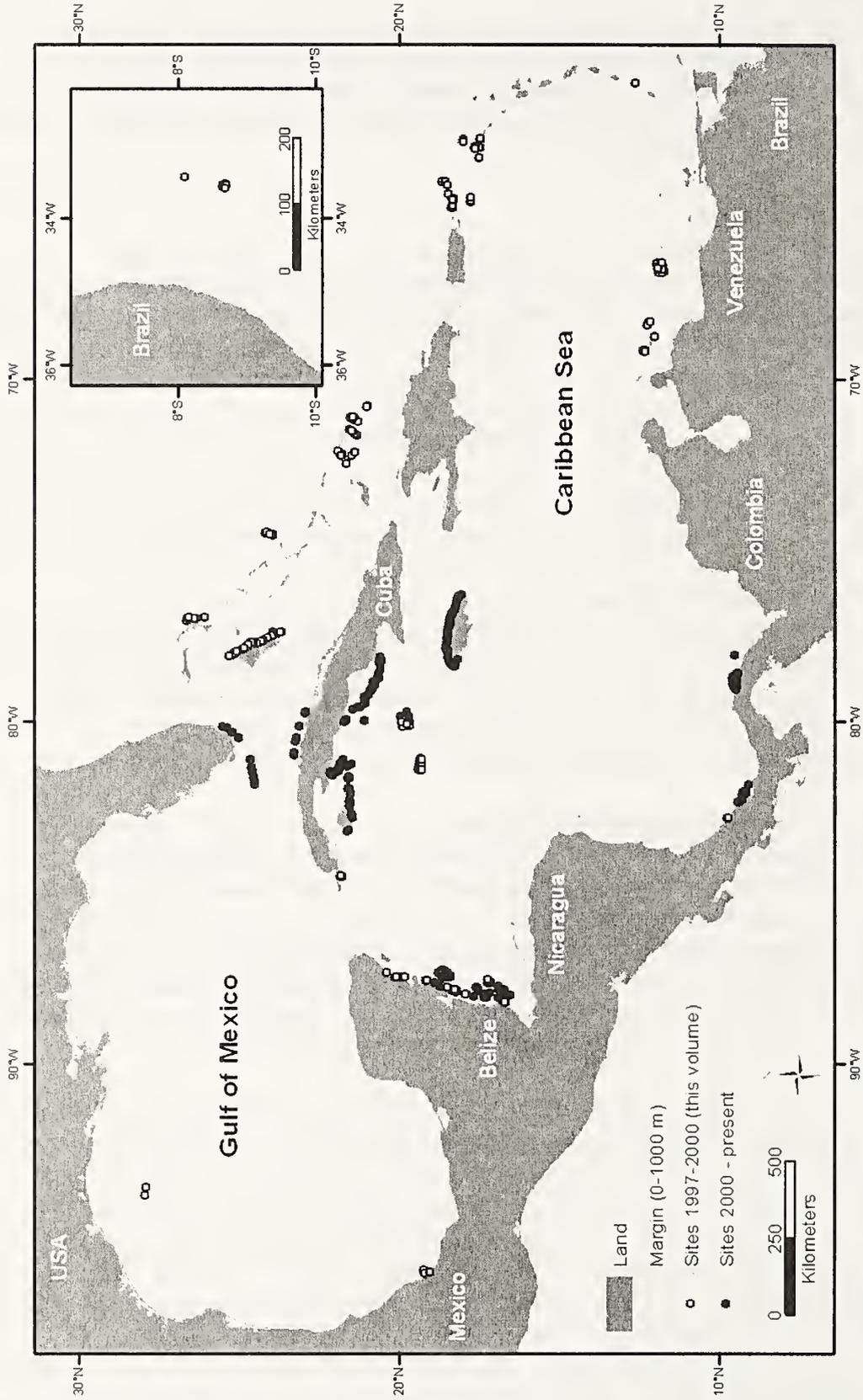


Plate 1. Location of all AGRRA sites assessed as of mid 2003.

## CAVEATS FOR THE AGRRA “INITIAL RESULTS” VOLUME

BY

JUDITH C. LANG<sup>1</sup>

### INTRODUCTION

The Atlantic and Gulf Rapid Reef Assessment (AGRRA) collaboration is designed for small teams of trained observers to quickly collect relatively simple quantitative indicators of the condition and/or abundance of stony corals, benthic algal groups, and reef-associated fishes at specific depth intervals in certain zones of maximum reef development. Results of the early (August 1997 to mid-2000) AGRRA assessments provide the focus of this volume. Coral reef ecosystems are so diverse, and their inhabitants engage in such intricate ecological relationships, that no rapid visual assessment technique can possibly provide in an unbiased manner all the information desired by scientists and resource managers for any given location. Comparisons among reefs are inherently constrained by numerous differences in physical environment, geomorphology, species composition, and proximity to direct human influences. Nevertheless, standardized application of the AGRRA methodology is facilitating multiscale spatial and temporal comparisons of key species, functional groups or guilds in the wider Caribbean (e.g., Ginsburg et al., 2000; Kramer, this volume). The purpose of this section is to alert readers to some of the special attributes of the AGRRA approach and some limitations in its initial application.

### QUALIFICATIONS

#### General Considerations

*Versions.* The AGRRA protocols have undergone several changes since their original posting in 1997 (see <http://coral.aoml.noaa.gov/agra/method/methodhome.htm> for the current version). Version 2, which is the basis for most of the research reported herein, is summarized in Appendix One (this volume). Given in the Methods section of each assessment paper are the particular version of the protocol that was used and any changes made in response to field conditions (or for any other reason).

*Sites.* Site selection criteria, and the rationale employed when any sites were chosen for “strategic” purposes, are specified in the Methods.

*Nomenclature.* The generic and specific names in the Methodology that were posted on the Internet and found in most of the papers in this volume are based on Foster (1987) for *Stephanocoenia*, Weil and Knowlton (1994) for the *Montastraea annularis*

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species complex, and two publications of the American Fisheries Society (AFS): Cairns et al. (1991) for the remaining stony corals and Robbins et al. (1991) for fishes. The nomenclature and/or spelling of certain species differ from those given in the several editions of Paul Humann's exemplary field guides that are widely used in the field by the AGRRA observers and/or from Eschmeyer's (1998) revised *Catalog of Fishes*.

*Consistency.* A degree of subjectivity is inherent in many of the decisions made when executing these protocols and subtle distinctions will occur as a function of the observer's knowledge and level of experience. Consistency training to standardize the visual assessments followed by periodic reviews are two important components of the AGRRA methodology. Specific efforts to reduce observer bias among team members are described in the individual assessment papers. It must be admitted, however, that when divers and/or time become limited, in-water reviews are likely to be sacrificed. Inter-observer variability undoubtedly contributes to some of the larger variance values, especially in the means for individual assessment sites.

*Spatial coverage.* During the time interval covered by the papers in this volume, several of the AGRRA geographic subregions were either poorly represented (e.g., Cuba, Lesser Antilles) or missing (e.g., Panamá, Hispaniola) as was the entire Florida region. Due to various circumstances (e.g., funding, geography), some papers are limited in spatial coverage and/or in the number of assessed sites. Furthermore, the two high-relief habitats of particular interest (*Acropora palmata* in ~1-5 m, and fore reef or equivalent in ~8-15 m) are not present in all coral reef areas or, if present, sometimes could not be assessed for other reasons such as remoteness or weather.

*Temporal Coverage.* Some of the individual assessments predate the 1998 ENSO while others either overlap with or postdate this extreme event, the effects of which were not experienced uniformly across the western Atlantic. Hurricanes (Georges, Mitch, Lenny) and outbreaks of disease also had nonrandom spatial and temporal distribution patterns profoundly affecting some of the assessed reef areas without influencing others. Particularly when recent partial mortality estimates of stony corals from different areas are compared, it is important to note the dates of the various assessments.

*Synthesis.* In order to include all the data in the initial Synthesis, Kramer (this volume) has provisionally classified each site as either shallow ( $\leq 5$  m) or deep ( $>5$  m) on the basis of its mean depth under the benthic transect lines with some resulting mixing of habitats and reef types. Each assessment has been treated as a separate unit and given equal weight in his analysis (Kramer, this volume). Hence its contribution is independent of the areal extent of the local reef system and of the numbers of assessed sites (or habitat types). In other words, the small (Costa Rica, Flower Garden Banks) and large (Cuba) areas, each with few assessed sites as of mid 2000, have been treated the same as the small (windward Netherlands Antilles) and large (Andros) areas for which a larger number of sites had been assessed (The corresponding numbers of habitats are three, one, two, numerous, and two, respectively.)

As explained by Kramer (this volume), the specific datasets and methodology used for calculating site means in the Synthesis chapter differ from those employed in most of the individual assessment papers.

## Stony Corals

*Condition.* The prevalence of bleaching, disease, predation, overgrowth, etc. are each expressed as a percentage of the surveyed population, i.e., all colonies of  $\geq 10$  cm (or  $\geq 25$  cm) maximum diameter that underlie the haphazardly placed 10-m transect lines. Observers vary in how much information they record as a function of time available and/or by their familiarity with these disturbances. Photographs and descriptions of the perturbations that commonly affect the wider Caribbean's stony corals are now available at several web sites, in sets of laminated field cards, and in Bruckner (2002), yet there is no substitute for good, in-situ training. Given the variability of signs displayed by disturbed corals, however, even the most experienced observers are presently unable to reliably distinguish between the effects of certain diseases and certain predators, particularly during rapid "snapshots" like the AGRRA assessments. Therefore "absence of evidence" in some locations cannot be taken as necessarily indicating "evidence of absence." For example, the AGRRA geologists have a tendency to report a higher proportion of stony corals that are "standing dead" (= completely dead with the colony still in growth position and recognizable at least to genus) than have the AGRRA biologists (P. Kramer, personal communication).

*Mortality.* "Recent" and "old" mortality of stony corals is estimated as the percentages of their outward-facing surfaces that are dead when seen from above the colonies. Hence, average "partial-colony mortality" (or partial mortality) refers to the mean percent of tissue loss/colony and not to the percent of colonies with any (necessarily unspecified amount of) tissue loss. "Recent partial mortality" (after Díaz et al., 1995) in the AGRRA benthos protocol encompasses that percentage of the colony surface in which the skeleton is white and covered by a (necessarily thin) layer of algae or fine mud. "Old partial mortality" is used to describe the corresponding percentage in which the skeletal structures are no longer white and have either been lost or are covered by epibenthic organisms that are not easily removed. Standing dead corals are included in the calculations of mean values for old and total (= recent + old) partial mortality in all but one of the individual assessment papers; in the Synthesis they are excluded from the mean values of old partial mortality (Kramer, this volume).

Attempts to "flesh out" these definitions and, in the absence of published data, to add putative temporal ranges to the definition of recent mortality, are given in the Methodology section of the AGRRA web site and by Bruckner and Bruckner (this volume), Kramer (this volume), and Steneck and Lang (this volume). Recently occurring mortality will be underestimated when: (a) exposed skeletal surfaces are quickly covered with sediment and/or algae (Fonseca, this volume); (b) turf algal-sediment mats expand at the expense of stony corals without creating any noticeable "recently dead" areas at their interfaces (Roy, personal communication); and (c) in the presence of superior spatial competitors (Deschamp et al., this volume) like the rapidly growing *Trididemun solidum* (Bak et al., 1981) since the skeleton that is being overgrown is never exposed to view.

## Algae

Functional algal groups are characterized by their abundance in 25 cm x 25 cm (= 0.0625 m<sup>2</sup>) quadrats with at least 80% coverage by any kind of benthic algae. The location of the quadrats is spatially limited to a 1-m radius of the 2-m marks on the transect lines. Although not a measure of algal cover, the abundance of each group on exposed substrata that are available to herbivores is provided by these data. As the identity of the functional groups that are assessed was changed between Versions 2 and 3 of the protocol, the usage of “relative abundance” has been restricted in this volume to the groups that were estimated in Versions 1 and 2 (i.e., macroalgae, turf algae and crustose coralline algae).

## Fishes

The AGRRA benthos protocol is a novel collaborative creation (see the Forward and Appendix One, this volume) that is still being fine-tuned as we gain experience with its application in diverse geographical areas. In contrast, only minor adjustments have been implemented thus far with the fish belt transects (here restricted to ecologically important herbivores and commercially significant carnivores) and Roving Diver Technique (Schmitt and Sullivan, 1996). Both had been thoroughly tested for some years prior to their adoption for the AGRRA fish protocol. Hence their relative strengths and limitations are better understood (e.g., Brock, 1954; Sale, 1980; Thresher and Gunn, 1986; Fowler, 1987; Schmitt et al., 2002). For example, serranids (e.g., Pattengill-Semmens and Semmens, this volume) are generally underreported in the belt transects, especially on reefs with high structural complexity (Kramer, Marks and Turnbull, this volume). Also underestimated in belt transects are roving schools of scarids or acanthurids (Nemeth et al., this volume). While the Roving Diver Technique provides a relatively rapid quantification of reef fish assemblage, longer search times or a larger number of searches than are appropriate for rapid assessments would be needed to fully estimate species richness (Nemeth et al., this volume; Marks, personal communication; Semmens, personal communication).

## AFFIRMATION

Caveats notwithstanding, our understanding of reef condition in the western Atlantic is enhanced as a result of the early (August 1997 to mid 2000) AGRRA efforts reported in this volume. Some important geographical gaps have been filled during subsequent assessments: northern and western Jamaica; southwestern and south-central Cuba; Bocas del Toro and western Kuna Yala, Panamá; and Upper and Lower Keys, Florida. The data added from these (and remaining as-yet unvisited) areas is certain to modify some of the initial conclusions presented in this volume. Pending the outcome of their analysis we anticipate being able to provide a more complete accounting of the overall status of the coral reefs in the Intra-Americas Seas and Brazil.

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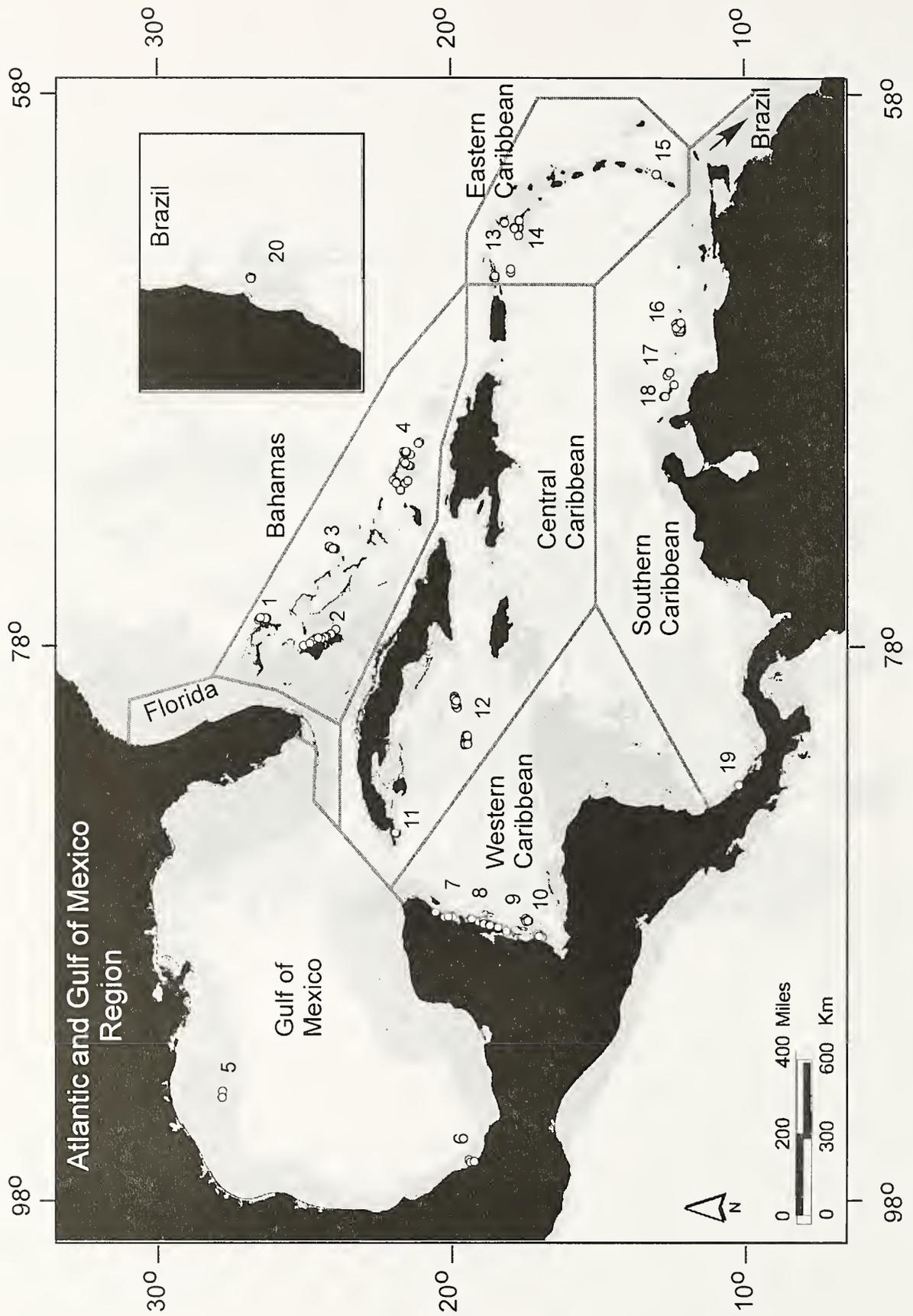


Figure 1. Map of western Atlantic showing AGRRA regional division boundaries and site locations for surveys reported on in this volume.

# SYNTHESIS OF CORAL REEF HEALTH INDICATORS FOR THE WESTERN ATLANTIC: RESULTS OF THE AGRRA PROGRAM (1997-2000)

BY

PHILIP A. KRAMER<sup>1</sup>

(On behalf of the AGRRA contributors to this volume)

## ABSTRACT

The Atlantic and Gulf Rapid Reef Assessment (AGRRA) sampling strategy is designed to collect both descriptive and quantitative information for a large number of reef vitality indicators over large spatial scales. AGRRA assessments conducted between 1998 and 2000 across a spectrum of western Atlantic reefs with different histories of disturbance, environmental conditions, and fishing pressure were examined to reveal means and variances for 15 indicators. Twenty surveys were compiled into a database containing a total of 302 benthic sites (249 deep, 53 shallow), 2,337 benthic transects, 14,000 quadrats, 22,553 stony corals. Seventeen surveys contained comparable fish data for a total of 247 fish sites (206 deep, 41 shallow), 2,488 fish transects, and 71,102 fishes. Shallow ( $\leq 5$  m) reefs were dominated by *A. palmata*, a good proportion of which was standing dead, while deep ( $>5$ m) reefs were nearly always dominated by the *Montastraea annularis* species complex. Fish communities were dominated by acanthurids and scarids with seranids making up less than 1% of the fish seen on shallow reefs and 4% on deep reefs.

AGRRA benthic and fish indicators on deep reefs showed the highest variation at the smallest spatial scale ( $\sim <0.1$  km), with recent mortality and macroalgal canopy height displaying the largest area and subregional scale ( $\sim 1$ -100 km) variation. A mean live coral cover of 26% for the 20 survey areas was determined for the deep sites. Significant bleaching and disease-induced mortality of stony corals associated with the 1998 (El Niño-Southern Oscillation) ENSO event were most apparent in the western Caribbean and Bahamas subregions and the *Montastraea annularis* complex was the most heavily impacted.

The overall low number of sightings for larger-bodied groupers and snappers ( $\sim < 1/100$  m<sup>2</sup>) as a whole suggest that the entire region is overfished for many of these more heavily targeted species. More remote reefs showed as much evidence of reef degradation as reefs more proximal to human coastal development. Characterizing present-day reef condition across the region is a complex problem since there are likely multiple sources of stress operating over several spatial and temporal scales. Notwithstanding the many

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limitations of this analysis, the value of making multiple observations across multiple spatial scales that can approximate the “normal” state for the region today is still very high.

## INTRODUCTION

Coral reefs in the western Atlantic have undergone massive changes over the past several decades from coral-dominated to algal-dominated states. Widespread impacts such as the 1983 die-off of *Diadema antillarum* and severe overfishing have disrupted herbivory processes. Coral diseases, like the white-band disease epizootic (Green and Bruckner, 2000) and bleaching (Wilkinson, 2000) have decimated previously healthy coral populations. Our understanding of these and other decadal-scale changes comes largely from a limited number of descriptive studies where long-term monitoring has been conducted. Some of the most frequently cited examples are those from Jamaica (Hughes, 1994), Florida Keys (Dustan and Halas, 1987; Porter and Meir, 1992), Curaçao (Bak and Nieuwland, 1995); and Belize (Aronson and Precht, 1997). With the exception of the Caribbean Coastal Marine Productivity (CARICOMP) program (Kjerfve et al., 1998) and the fish surveys of the Reef Environmental and Education Foundation (REEF) ([www.reef.org](http://www.reef.org)), few studies in the western Atlantic (Caribbean, Gulf of Mexico, Florida, Bahamas, and Brazil) have examined the condition of coral reefs over large spatial scales (100s-1000s km). Moreover, most studies have focused on localized impacts and used diverse sampling methods making statistical comparisons on larger spatial scales impossible. Establishing regional patterns of reef condition is essential for characterizing the extent and severity of decline and developing hypotheses about the causes of decline (Ginsburg and Glynn, 1994).

AGRRA was conceived to provide a “snapshot” characterization of a number of structurally or functionally important benthic and fish indicators on western Atlantic coral reefs. When applied synoptically to the entire region, results can be used to develop a biotic index of relative health or condition. The concept of evaluating “ecosystem health” is a rapidly emerging science for which a number of different definitions and approaches have been suggested (e.g., Costanza, 1992; Rapport et al., 1998, 1999). Because health involves the response of structural and functional components of the ecosystem, most approaches have used more than a single indicator. Some of these include naturalness, normality, productivity, organization (species diversity and complexity of interactions), and resilience (Coates et al., 2002). Normality in itself has been shown to be particularly useful since it provides guidelines for the range of system states. The AGRRA approach relies on normality as its principal measure of condition and these norms are meant to represent a baseline for the region. The methodology and indices, which were developed with the advice of specialists and based on current scientific understanding of coral reef dynamics as well as standard monitoring methodologies, are summarized below.

## Coral Condition

The condition of the principal scleractinian and hydrozoan corals that contribute most to the construction and maintenance of the three-dimensional framework is critical for determining the long-term integrity of the reef ecosystems (Dustan and Halas, 1987; Done, 1997). Species composition, colony size, mortality, recruitment, disease, bleaching, predation, coral cover, etc., are examined over large spatial scales in the AGRRA assessment. Estimates of colony size provide information on rugosity, architectural complexity, and an approximation of colony age (Hughes and Jackson, 1980, 1985). Visual estimates of the partial mortality of colonies (hereafter partial mortality) are used to distinguish between the amount of moribund tissue considered recently dead (about 1 year) and long dead (more than 1 year). The amount of “recent mortality” indicates current impacts while “old mortality” is primarily an integration of mortality over longer time scales. The AGRRA data, within certain limits, can be used to calculate size frequency distributions (Bak and Meesters, 1998) as well as identify mortality patterns related to size and/or species identity. Given current knowledge it is hypothesized that large percentages of recent partial mortality are a signal of decline yet some level of old mortality is expected, at least in the larger corals (Hughes and Connell, 1999). By examining coral condition indices on many types of reefs throughout the region, patterns should emerge to establish and help formulate hypotheses on causes.

## Algae

There has been a noticeable change on many reefs around the wider Caribbean from coral-dominated communities to those dominated by macroalgae (e.g., Done, 1992; Hughes, 1994; Hallock et al., 1993; Dustan and Halas, 1987; Lewis, 1986; Steneck and Detheir, 1994; McClanahan and Muthiga, 1998). The causes for these shifts have been attributed, in part, to a loss of key herbivorous fishes and sea urchins, particularly *Diadema antillarum*. *Diadema* affects coral reef structure and composition, including algal composition and abundance, by competition with other grazers, particularly certain fishes, and by erosion of coral skeletons (e.g., Steneck and Dethier, 1994; Roberts, 1995; McClanahan and Muthiga, 1998). The dramatic decline of *Diadema* that occurred after its 1983 die-off (Lessios et al., 1984) led to the dominance of many reefs in Jamaica by fleshy and calcareous macro algae, increased coral mortality and decrease in coral recruitment (Hughes, 1994). Recent reports have suggested the abundance of *Diadema* has been increasing in localized areas but little regional information is available on its recovery and subsequent influence on the condition of reefs (but see Edmunds and Carpenter, 2001).

The objective of assessing algae is to quantify the relative abundance of several key functional groups (crustose corallines, macroalgae and, initially, turf algae), and to relate these abundances with herbivorous fish biomass and coral condition. Given that reefs in decline often have high fleshy (noncalcified) macroalgal biomass, sometimes accompanied by a high biomass of *Halimeda*, it is expected that reefs with a low macroalgal:crustose coralline ratio, or a low macroalgal index (= macroalgal abundance x

macroalgal height) are more biologically intact than those with a high ratio (after Steneck and Dethier, 1994).

## Fishes

Reef fish, as predators or grazers, play important roles in the community dynamics of coral reefs through their interactions with corals, algae and other herbivores (Roberts, 1995). Fish communities respond to disturbance in various ways depending on the type and degree of perturbation. Various combinations of commercial, subsistence and recreational fishing, particularly of herbivores, constitute some of the most widespread and greatest anthropogenic impacts on coral reefs (Roberts, 1995). In particular, the loss of certain indicator species and guilds causes both direct and indirect shifts in fish community structure as well as in other components of reef ecosystems (Munro and Williams, 1985; McClanahan and Muthiga, 1988; Hughes, 1993, 1994). Disruption in the balance of reef fish assemblages can result in decreased coral cover and increased algal abundance (Roberts, 1995, 1997; McClanahan et al., 1996). Yet only a few studies have examined the response of fish communities to degradation or changes in habitat structure and composition (Jones and Syms, 1998). Whereas visual fish censuses have been conducted throughout the Caribbean and Gulf of Mexico, relatively few are comparable interregionally because of the various methodologies used (Sale, 1998).

The AGRRA approach includes two distinct assessment methods that provide complementary “snapshots” of fishes at a given site although they do not fully account for the daily, tidal, and seasonal changes known to occur in reef habitats (Ault and Johnson, 1998; Bellwood, 1988; Willis, 2001). While subtle differences among fish assemblages can be difficult to detect with this level of sampling, robust patterns across large number of assessments can be revealed. It is expected that areas near human populations will have lower abundance of fishes, particularly commercially significant species.

In this paper I have synthesized the AGRRA data on coral reef condition collected between 1998 and 2000 by a network of scientists from 20 distinct locations in the western Atlantic extending over 1,000 km (Table 1). The areas presented in this synthesis include a wide spectrum of situations with respect to history of disturbance, environmental conditions, and fishing pressure from humans. The principal goals of this synthesis are to examine over various spatial scales (0.1-100 km):

- 1) variability of coral condition (mortality, recruitment, disease, damage),
- 2) the relative abundances of major algal functional groups and factors contributing to any differences in macroalgal index,
- 3) patterns of spatial variability of the abundance and size of targeted fish species or key guilds such as herbivores to evaluate the degree of overfishing, and
- 4) the integration of indices into a biotic “reef health index” that can be used in an exploratory way to examine patterns and form hypotheses for future experimental testing.

This synthesis is unique in providing the first regional perspective on coral reef condition in the western Atlantic that is based on multiple indicators and in its

examination of spatial variation and trends of reef condition on multiple spatial scales. The results summarized here originate entirely from the data generously contributed by the authors of this volume to the AGRRA database and may differ slightly from that reported in their papers because of differences in the way the data have been analyzed. The reader is referred to individual papers in this volume for more specific information about each of the assessment areas.

## METHODS

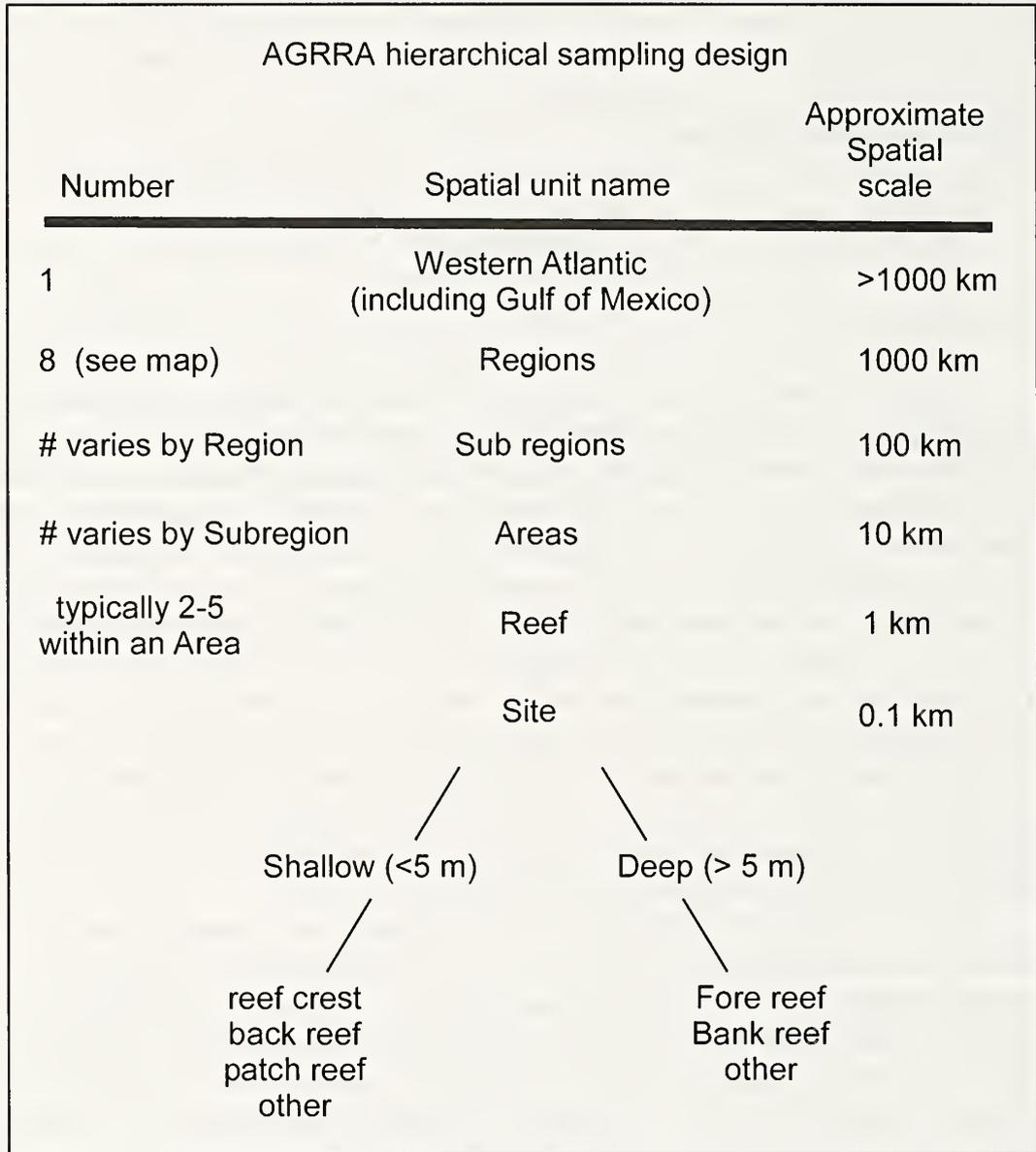
### Sampling Design

A map of the western Atlantic showing AGRRA regional division boundaries and geographic location (indicated by identification number) for the 20 assessments reported on in this volume is shown in Figure 1. All data submitted by individual AGRRA teams in the form of preformatted spreadsheets were checked for errors and standardized to uniform species codes. The timing of the assessments spans a three-year period, many of which (~12) were conducted during the wet-season months (June-August) in much of the wider Caribbean. To synthesize results from the many assessments, an Access database was developed that allows users to manage large quantities of data, make statistical comparisons across various spatial scales, and provide this information to other AGRRA scientists, resource managers and interested researchers. A version of the database (Version 1.1) covering the period from 1997-2000 is currently available and newer versions will be released in the future. Fish data collected using the Roving Diver Technique are not discussed here as they are not part of the AGRRA database but instead are housed in the REEF database ([www.reef.org](http://www.reef.org)).

A synopsis of the AGRRA methodology is provided in Appendix One (this volume). The majority of the AGRRA teams used Version 2 of the protocol. Conceptually, the AGRRA sampling design is based on principles of stratified two-stage sampling (Cochrane, 1977). A hierarchical multiscale sampling approach and associated spatial scales are shown in Figure 2. The sampling domain, defined as the western Atlantic *region*, is subdivided spatially into *subregions*, *areas*, *reefs*, and *sites*. Eight subregions are recognized in the western Atlantic: Gulf of Mexico, Bahamas, western Caribbean, central Caribbean, eastern Caribbean, southern Caribbean, Brazil, and Florida. AGRRA data have been collected from each of these subregions except Florida, for which an assessment is scheduled in summer 2003.

Further stratification of reef types can be accomplished using a combination of depth, cross-shelf position, and geomorphology to delineate areas of similarity over any specified geographic area. One aim of AGRRA sampling is to select a series of “typical” sites that are representative of the geographic area. For the purpose of this synthesis, however, *sites* were only distinguished based on the mean benthic transect depth into “shallow” ( $\leq 5$  m, mostly reef crests and patch reefs) and “deep” ( $> 5$  m, mostly fore-reef slopes) categories.

The appropriate sampling effort needed to characterize sites, areas, and subregions can vary from location to location depending on the spatial variation of coral,



**Figure 2.** Hierarchical sampling scheme and associated spatial scales for AGRRA surveys. Sites are also classified hierarchically based on depth, geomorphology, relief, and additional modifiers. For the purpose of this synthesis, sites are only distinguished based on depth into shallow ( $\leq 5$  m) and deep ( $> 5$  m).

algal and fish indicators. Often AGRRA surveys are the first to be conducted in an area and/or little or no suitable baseline data exist to determine the appropriate level of sampling that would be needed to characterize the area fully. Furthermore, each indicator will have a separate distribution and desired degree of precision. A power analysis was conducted for several of the AGRRA indicators to examine the tradeoff between sampling effort and precision at both the site and subregional spatial scales. To assess how many transects were needed to adequately estimate *site* means for each indicator, within-site variance as a function of sampling effort was examined at one site in each of four areas.

## Data Analysis

The database used for this synthesis contained a total of 302 benthic sites (249 deep, 53 shallow), 2,337 benthic transects, 14,000 quadrats, 22,553 corals, 247 (206 deep, 41 shallow) fish sites, 2,488 fish transects, and 71,102 fishes. For this initial analysis, I divided the sites into either shallow ( $\leq 5$  m) or deep ( $> 5$  m) depth categories based on the mean depth of the benthic transects, whereas sites were categorized by depth ranges or by mean depth of the habitat in some of the individual assessments in this volume. In several cases (e.g., Curaçao and the Virgin Islands) the same geographic site was assessed at different times. I treated these sites as independent samples whereas in the individual assessment papers they are grouped together as a single site. Not all results described in the individual assessments were included in the synthesis either due to inconsistencies in the way the methodology was implemented or because of missing data for individual transects or entire sites. The cylindrical (volumetric) fish counts used in Belize and San Salvador, Bahamas (see both papers by Peckol et al., this volume) could not be directly compared to the belt transect data. The 1997 Andros, Bahamas survey followed an early version of the protocol and differed enough from later versions that only data collected during 1998 were used.

To equalize sampling effort, results in the synthesis have only incorporated a maximum of the first 10 transects/site in calculated averages for either fish or benthic parameters, despite larger sample numbers for some sites. Statistical analysis was performed using STATISTICA software, Version 6.0 (StatSoft, Inc., 2002). The percent live coral cover, percent partial-colony mortality (recent, old, total), standing dead (i.e., completely dead and still in original growth position, and calculated as a percentage of the total), mean colony size, prevalence of disease and bleaching, and relative algal abundance were calculated and summarized. Only data from corals that were  $\geq 25$  cm in maximum diameter was included. I chose to remove standing dead colonies from old partial mortality averages; thus, averages in the synthesis should be lower than those in the individual papers for sites with significant amounts of standing dead coral. The macroalgal index, a proxy for macroalgal biomass, was calculated as % *relative* macroalgal abundance (as approximated in Versions 1 and 2 of the AGRRA protocol) x macroalgal canopy height. Of the 75 species of fishes in the AGRRA belt transect list (see Appendix One, this volume), a total of 18 species of herbivores, both territorial and non-territorial, were analyzed (all acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus* (yellowtail damselfish) and black durgon (*Melichthys niger*). Free-swimming predators

and sedentary predators were classified as carnivores (all lutjanids, all serranids, and *Sphyræna barracuda*) for a total of 24 species. Fish densities (#/100 m<sup>2</sup>) were calculated for each area for all 75 species and included all sightings except for scarids and haemulids where only the  $\geq 5$  cm sightings were used. Size information was used to calculate biomass for each fish species using the standardized conversion equations shown in Appendix Two (this volume).

Regional norms for different indicators were calculated by averaging reported survey values without weighting by the spatial extent of reefs or number of sites conducted within a given survey area. Several parameters were analyzed by students t-test and by 1-way and 2-way Analysis of Variance (ANOVA). The Plymouth Marine Laboratory's PRIMER software, Version 5 (Clarke and Gorley, 2001), was used to examine similarity among different indicators with either Bray Curtis similarity or two-dimensional multidimensional scaling (MDS) ordination.

To examine relationships between fish data and benthic habitat variables, regression analysis was used to analyze (1) mean herbivore density and mean macroalgal index for 17 assessments and (2) algal canopy height and density of the  $\geq 25$  cm corals. To examine the relationship of coral condition with respect to anthropogenic threats, survey areas were classified into one of three threat categories (high, medium, low) based on Bryant et al.'s (1998) global threat analysis (Table 1). The modeled threat layer of "overexploitation of marine resources," which is based primarily on proximity of a reef to coastal settlements, was used to examine relationships of fish density and biomass to presumed fishing pressure.

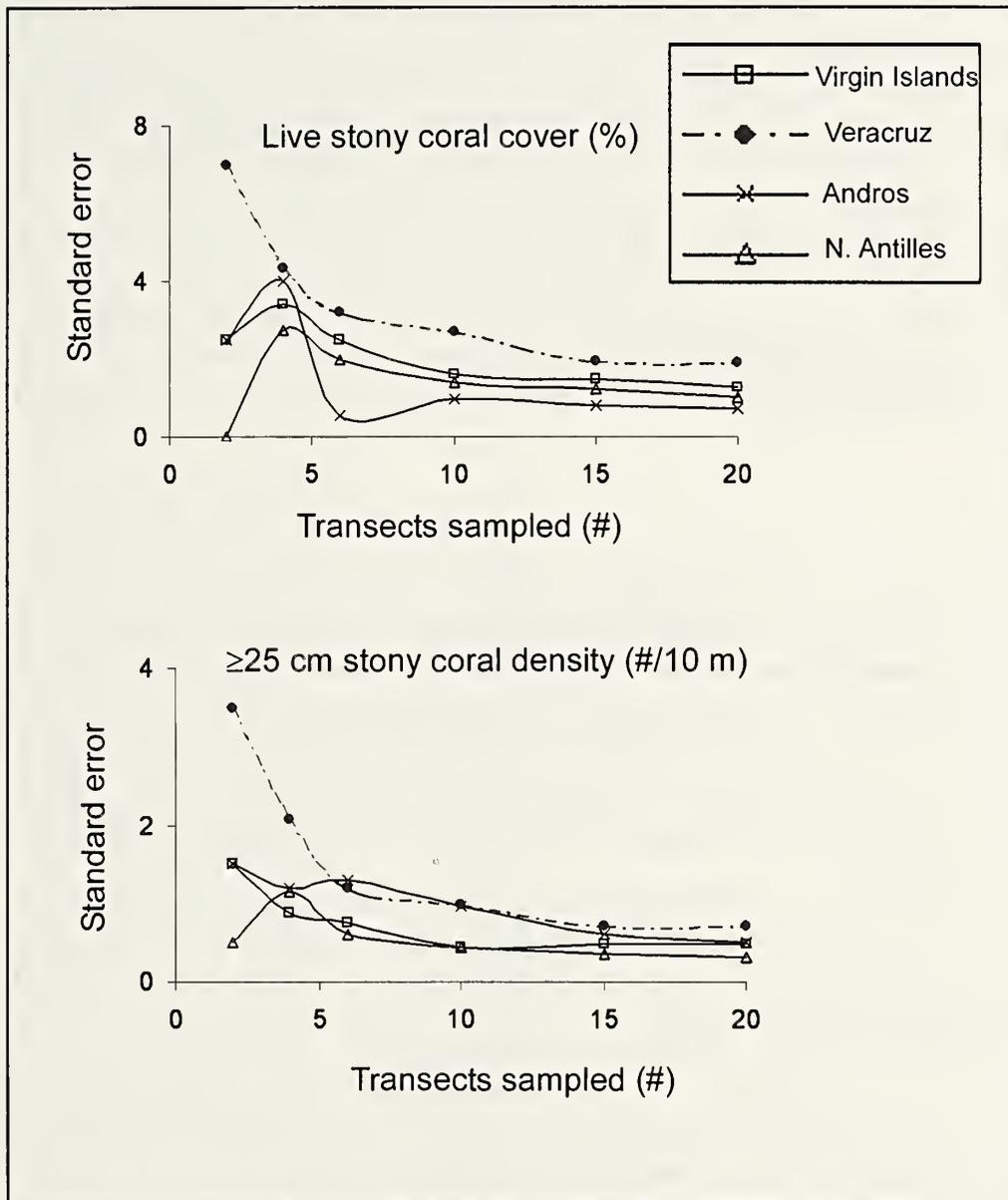
## RESULTS

Results in the Synthesis differ somewhat from those reported in the individual papers in this volume due to differences in how sites were defined or stratified, in the selection criteria for calculating site means and, for stony corals, in the definition of old partial mortality (see Methods).

### Sampling Effort

The characteristics of the sampling effort for the 20 assessments in this volume are summarized in Table 1. Each assessment is identified by its unique identification number (ID#) in both figures and tables. At least some deep ( $>5$  m) habitats were assessed by all teams but shallow ( $\leq 5$  m) reefs were examined extensively only in Andros (ID#2) and the Abrolhos, Brazil (ID#20). For this reason, most of the spatial comparisons in this synthesis are restricted to deep ( $>5$  m) habitats. A detailed summary of the mean sampling effort for these deep sites in each assessment is given in Table 2.

Sampling size (as number of transects) versus standard error for live coral cover and coral density from four different areas of the western Atlantic [Virgin Islands (ID#13), Veracruz, Mexico (ID#6), Andros (ID#2), windward Netherlands Antilles (ID#14)] are shown in Figure 3. For both parameters, the error was reduced most substantially within the first six transects after which only small improvements were



**Figure 3.** Sampling size (number of transects) versus standard error for four sites in different areas of the western Atlantic for live coral cover and coral density. For all parameters, the error is reduced most significantly within the first six transects, after which only small improvements are observed with increasing number of transects.

observed with increasing sample size. Figure 4 provides a summary of sampling size for several AGRRA benthic indicators versus standard error. When percent partial-colony mortality (recent and old) and coral diameter (as number of corals) were compared for four sites [Cayman (ID#12), Curaçao (ID#18), Belize (ID#10), and Andros (ID#2)], error was reduced most dramatically within the first 50 corals sampled. Algal relative abundance (macroalgae, crustose corallines and turfs) and macroalgal canopy height data were compared (as number of quadrats) for Veracruz, Abrolhos (ID# 20), Cayman, and Andros, with the standard error most reduced within the first 40 quadrats sampled.

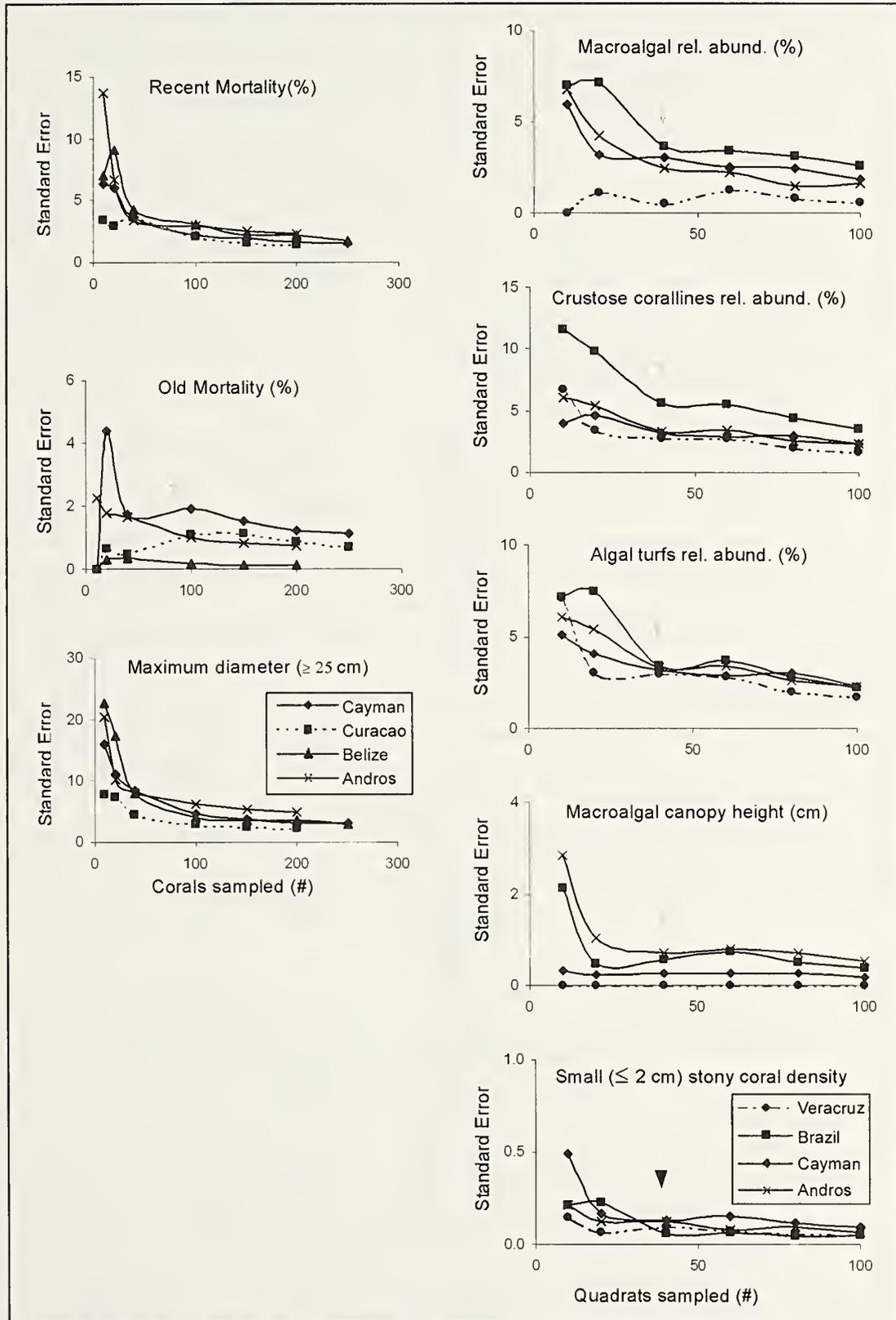
In nearly all of the fish belt-transect surveys, an average of 600 m<sup>2</sup> was assessed at each site with the exceptions being Akuma/Xcalak (ID#8) and Curaçao (ID#18), each with less than 200 m<sup>2</sup>. In both Mária la Gorda, Cuba (ID#11) and the Yucatan (ID#7), 50 x 2 m transects (a total of six per site) were used in accordance with an early version of the fish protocol, whereas 10 x 2 m transects (30/site) were employed in the small reefs of the Abrolhos (ID#20). The 14 remaining teams each performed 30 x 2 m transects (10/site).

### Stony Corals

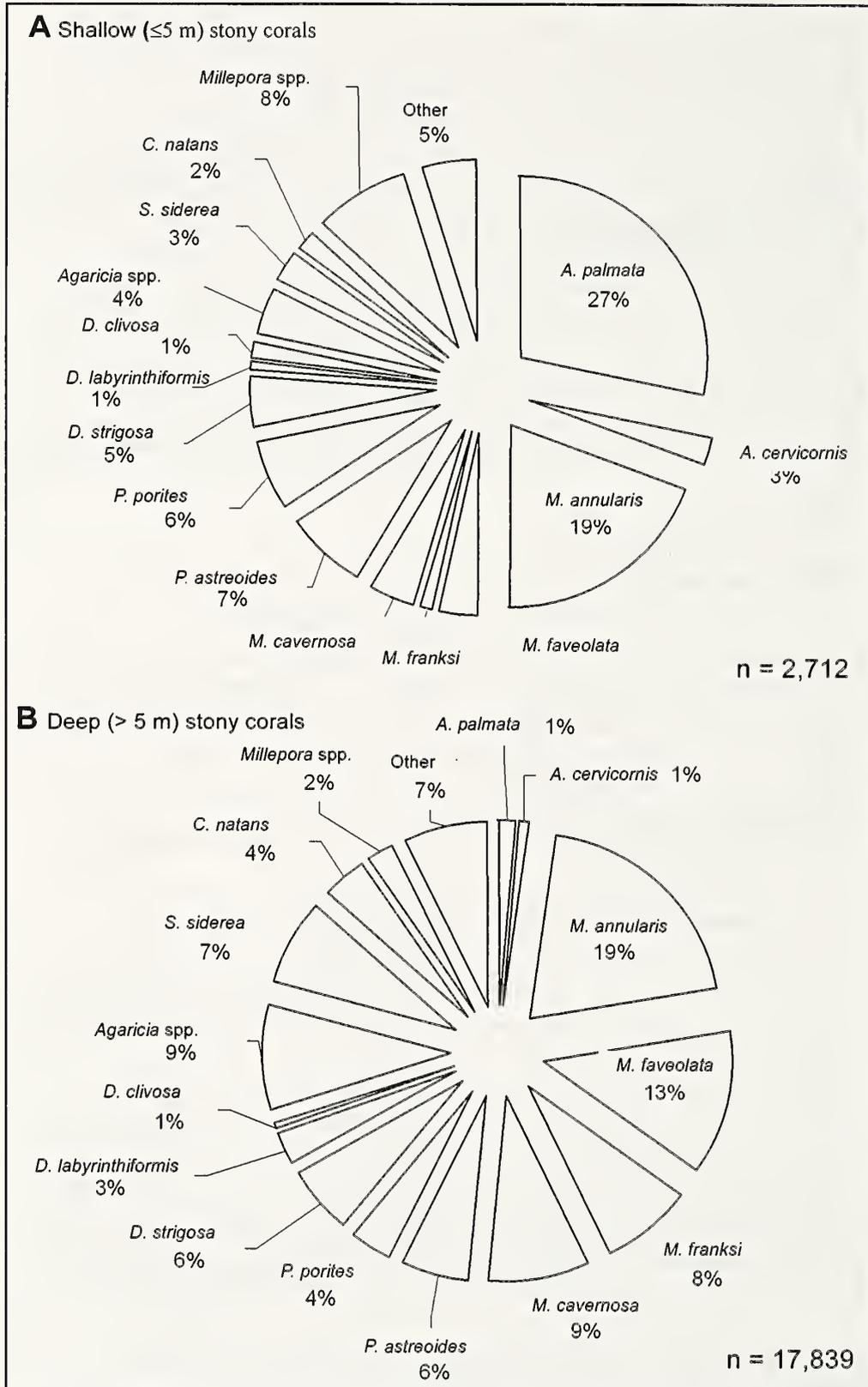
*Composition and abundance.* The 22,553 colonies sampled in the 20 surveys combined included 37 scleractinian and two hydrozoan species. The majority of these stony corals were assessed in deep (n=18,913 at >5 m) rather than shallow (n=3,640 at ≤5 m) sites. Of these, 1,122 colonies (5%) were surveyed in the Abrolhos (ID#20), where the endemic Brazilian scleractinian, *Mussismilia braziliensis*, was the dominant coral. For the remaining corals in the wider Caribbean excepting Veracruz (ID#6) where they were not assessed, standing dead colonies constituted 13.5% of the colonies at shallow sites and 2.5% at deep sites.

The relative species composition, expressed as a percent of the total of corals (excluding all standing dead colonies and all the Brazilian corals) is shown in Figure 5A,B. Numerically the most abundant species in shallow reefs were *Acropora palmata* (27% of total), *Montastraea annularis* (19% of total), and *Porites* spp. (7% *P. astreoides*, 6% *P. porites* of total) (Fig. 5A). Live *Acropora palmata* was the dominant coral at 60% of all shallow reef-crest sites. Taxa that are usually dominant on shallow Caribbean reefs, such as *Agaricia tenuifolia* and *Millepora* spp. were less abundant in the AGRRA regional dataset. For shallow reefs, there was especially high between-survey species variability caused, in part, by differences in the type of shallow reef assessed (reef crest, back reef, patch reef-see individual papers in this volume and Table 1). The high number of *Acropora palmata* reflects the relatively large number (Table 1) of reef crest sites assessed off Andros (ID#2) where this coral was common. Live *Acropora palmata* was the dominant coral (>1/3 of coral population) at 17 of 23 shallow reef-crest sites summarized in this volume.

Deep reefs displayed less species variability among sites since most assessments were conducted in fore reefs [exceptions being bank reefs surveyed in the Flower Garden Banks, Gulf of Mexico (ID#5) and Mouchoir Bank in the windward Netherlands Antilles (ID#14); see Table 1 and papers in this volume]. *Montastraea* spp. dominated most deep



**Figure 4.** Sampling size (number of corals and quadrats) for several AGRRA benthic indicators versus standard error for four sites in different areas of the western Atlantic. For corals, error is reduced most dramatically within the first 50 corals sampled. For quadrats, the error is reduced most significantly within the first 40 quadrats sampled.

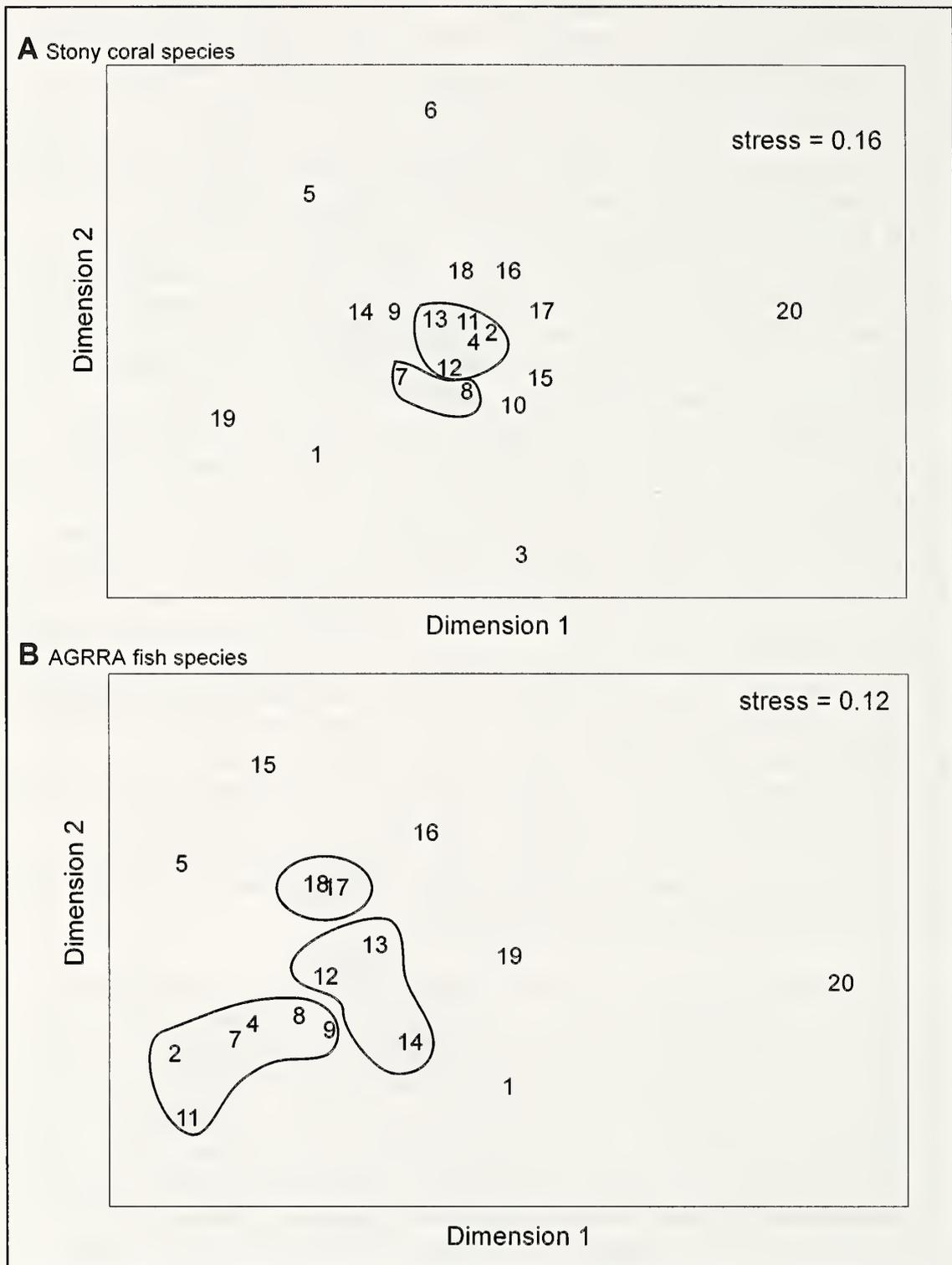


**Figure 5.** Composition of all stony corals (25 cm) from all assessments except Brazil (ID #20) combined for (A) shallow sites (5 m), and (B) deep sites (>5m). Includes only corals sampled within a maximum of 10 transects per site.

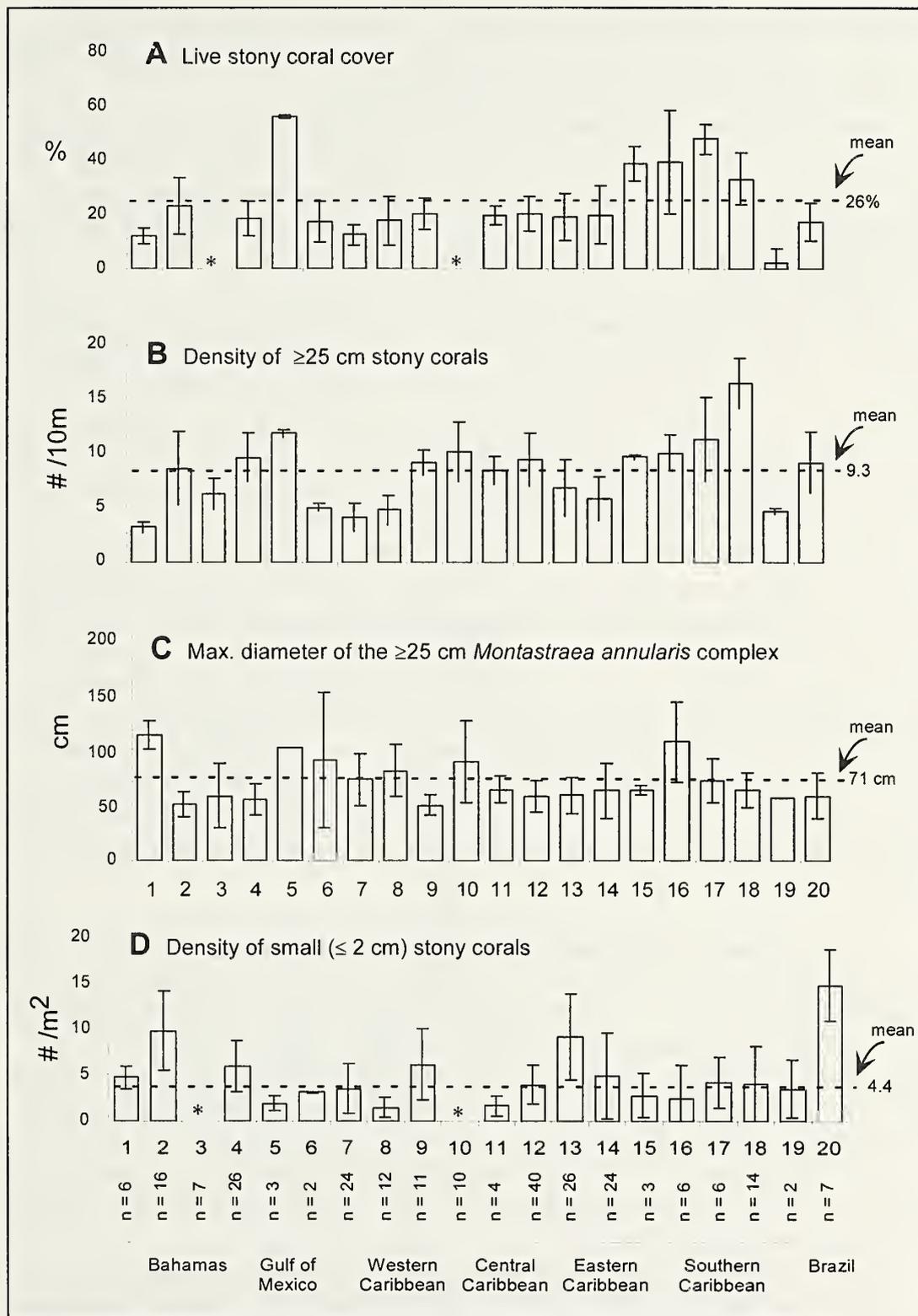
reef assemblages as follows: *M. annularis* (19% of total), *M. faveolata* (13% of total), *M. cavernosa* (9% of total) and *M. franksi* (8% of total). Other commonly observed taxa on fore reefs included *Agaricia* spp. (9% of total), *Siderastrea siderea* (7% of total), *Diploria strigosa* (6% of total), *Porites* spp. (*P. astreoides* 6%, *P. porites* 4% of total), and *Colpophyllia natans* (4% of total). *Acropora cervicornis*, a once-dominant fore-reef species, was rare. In the multidimensional scaling analysis based on mean species density in each assessment, outliers included Abaco (ID#1) and San Salvador (ID#3) in the Bahamas, the Flower Gardens (ID#5) and Veracruz, (ID#6) in the Gulf of Mexico, Costa Rica (ID#19), and the Abrolhos (ID#20). Cluster analysis revealed two clusters of high similarity (70% Bray Curtis similarity), shown in gray on the MDS ordination in Figure 6A. Yucatan (ID#7) and Akumal/Xcalak (ID#8; also in the Yucatan) were in the first cluster; the second cluster consisted of Andros (ID#2), Turks and Caicos (ID#4), Mária la Gorda (ID#11), the Cayman Islands (ID#12) and the Virgin Islands (ID#13).

Small ( $\leq 2$  cm) coral density averaged  $4/m^2$  for the region with shallow reefs having fewer of these “recruits” ( $3.3/m^2$ ) than deep reefs ( $4.4/m^2$ ) (Tables 3A, 3B). By far the highest reported densities ( $9-15/m^2$ ) for both shallow and deep reefs were in the Abrolhos (ID#20). On deep reefs, low densities ( $<2/m^2$ ) were observed in the Flower Gardens (ID#5), Veracruz (ID#6), Akumal/Xcalak, (ID#8), and Mária la Gorda (ID#11) (Table 3B). Their species richness was also higher for deep ( $\Sigma=35$  coral species) than for shallow ( $\Sigma=15$  species) reefs, although this was partly a function of survey effort. Species composition of the recruits did not reflect the large ( $\geq 25$  cm) framework-builders present for the region as a whole. Brooders such as *Porites* spp. and *Agaricia* spp. dominated the assemblages with *Porites astreoides* being the most abundant species in both shallow and deep reefs. Broadcast spawners were far less common and the *Montastraea annularis* species complex only comprised  $\sim 2\%$  of small corals observed overall. *Acropora palmata* (0.8% on shallow reefs) and *A. cervicornis* (0.2% on deep reefs) were both rare. An exception was *Siderastrea siderea*, for which the overall percentage of small corals was similar to, or greater than, adult abundance (3% versus 3% in shallow and 12% versus 7% in deep for  $\leq 2$  cm and  $\geq 25$  cm colonies, respectively).

*Live cover and size.* Live coral cover averaged 18% for shallow reefs with the highest ( $\sim 38\%$ ) value reported for the reef crest sites on Andros (ID#2) and the lowest ( $\sim 3\%$ ) for a patch reef in Costa Rica (ID#19) (Table 3A). On deep reefs, live coral cover ranged from 3% to 58%, averaging 26% for the region as a whole (Table 3B, Fig. 7A). The highest observed coral cover on deep reefs was in the Flower Gardens (ID#5). Four other assessments that are located in the southern-southeastern Caribbean [Los Roques (ID#16), Bonaire (ID#17), Curaçao (ID#18)) and St. Vincent (ID#15)] also had coral covers that exceeded the regional average (Table 3B). Apart from Curaçao (ID#18), Abrolhos (ID#20), and Costa Rica (ID #19), the density of large ( $\geq 25$  cm) colonies generally correlated with live coral cover for the other 15 survey areas ( $r^2 = 0.8$ ,  $p < 0.001$ ,  $n = 15$ , deep depths only). Shallow reefs had overall lower mean densities ( $\sim 7.9/10m$ ) than deep reefs ( $\sim 9.3/10m$ ) (Fig. 7B), although a significant correlation ( $r = 0.7$ ,  $p < 0.01$ ,  $n = 13$ ) existed between the shallow and deep coral densities for areas in which both depths were assessed. The density of large ( $\geq$ one meter maximum diameter) colonies of



**Figure 6.** (A) MDS ordination of 20 survey areas (deep sites only) based mean species density for 20 most common large stony corals ( $\approx 25$  cm diameter). Groupings indicate Bray-Curtis similarity of 60% or more for all 20 surveys. (B) MDS ordination of 17 survey areas where fish were surveyed using belt transects (deep sites only). The clustering is based on mean species density for 40 most common species (excluding haemulids). See Table 1 for ID codes and text for discussion.



**Figure 7.** Comparison between means and standard deviation of deep sites for all 20 AGRRA assessments. (A) live coral cover; (B) coral density (number of  $\geq 25$  cm diameter corals/10m); (C) colony diameter of the *Montastraea annularis* complex, and (D) small ( $\leq 2$  cm) coral density. The stars indicate a parameter that was not measured and the dashed line indicates the mean for all surveys. See Table 1 for ID codes.

*Montastraea* spp. at deep sites ranged from 0.01 to 3.1/10 m. Areas showing greatest densities were Flower Gardens, Bonaire, Los Roques, Curaçao, while areas with lowest densities were Brazil, Veracruz, Abaco, and Costa Rica.

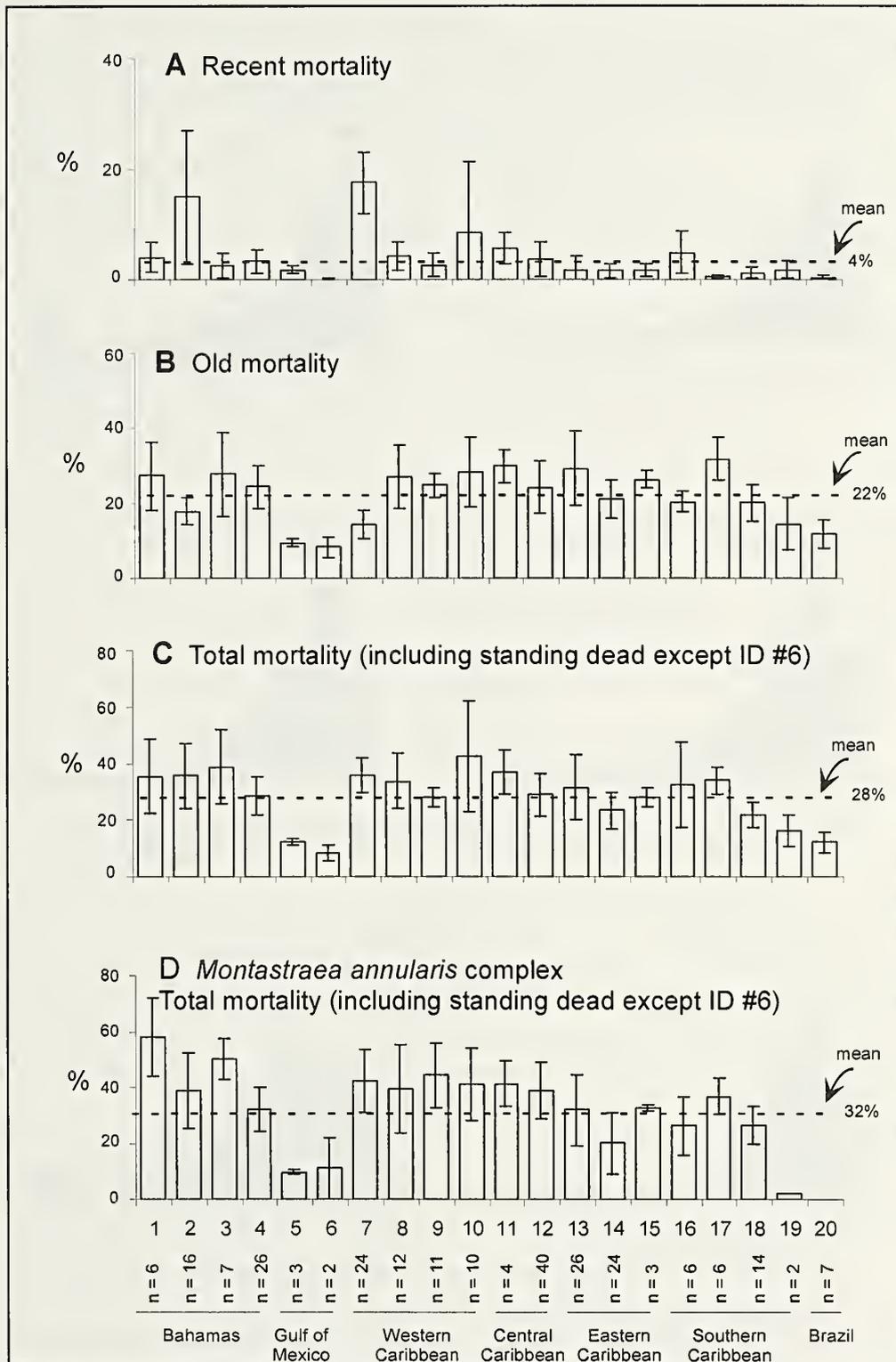
Mean size (based on maximum colony diameter) for 17 of the 18 most common species of corals was generally larger in shallow sites (n=53) than in deep sites (n=249). The mean diameter for all 7,398 colonies of the *Montastraea annularis* complex of species (*M. annularis*, *M. faveolata* and *M. franksi*) averaged 71 cm for deep reefs in the region (Fig. 7C) with the largest colonies having been observed in Abaco (ID#1), Los Roques (ID#16) and the Flower Gardens (ID#5).

Deep sites with higher than average live coral cover and abundance generally had larger than average colony diameters for the *Montastraea annularis* species complex (especially Los Roques and the Flower Gardens). Exceptions included Abaco and Akumal/Xcalak (ID#8) which had above average sizes for this species complex and lower than average coral cover (about 13% and 20%, respectively), and Bonaire (ID#17) where coral cover was the second-highest recorded (~46%) but maximum colony diameter for the *M. annularis* complex was only “average” (Fig. 7A,C).

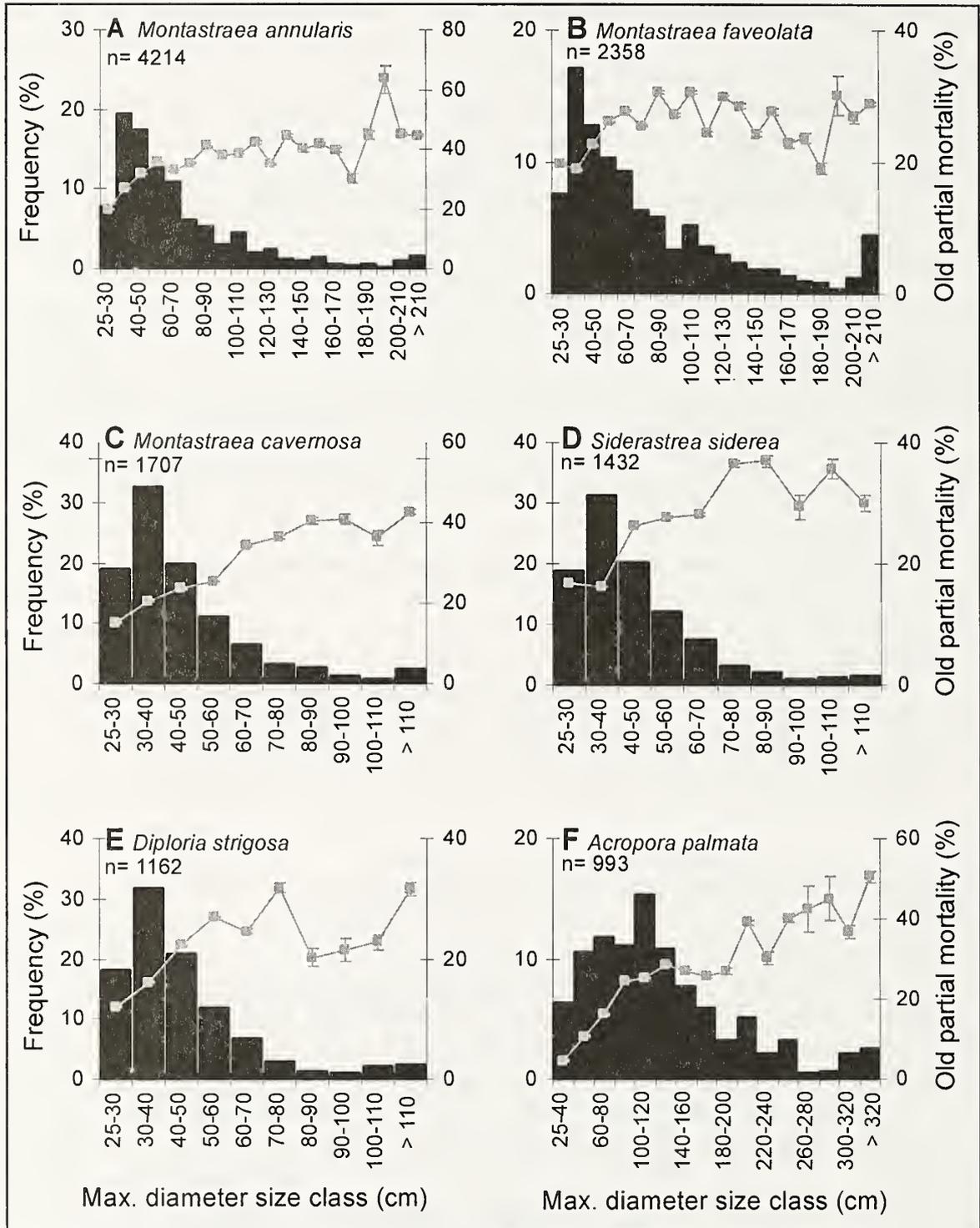
Size frequency distributions for five of the numerically-most-common taxa, *Montastraea annularis*, *M. faveolata*, *M. cavernosa*, *Siderastrea siderea*, and *Diploria strigosa* (all broadcast spawners; two being included in the above analysis), showed that most colonies were in the 30-40 cm size class (Fig. 8). *Acropora palmata* (another broadcaster) was most frequently observed in the 100-120 cm-size class. Brooding species (e.g., *Agaricia* spp. and *Porites* spp.) were predominantly distributed in smaller size classes. Overall, coral sizes were fairly similar across the region, with greatest variability evident on smaller spatial scales, particularly within sites. Areas thought to have more marginal conditions for coral growth showed consistently larger coral sizes of certain species as follows: Veracruz (*S. siderea*, *M. annularis*); Abaco (*M. annularis*, *Colpophyllia natans*); and Costa Rica (*D. strigosa*).

*Mortality.* Recent partial mortality for the  $\geq 25$  cm colonies (expressed as the mean proportion (%) of the colony’ surface area as seen from above) ranged from 0.1 to 27% across the 20 survey locations with a regional average of 4% for both shallow and deep reef types. High levels of recent mortality ( $\sim > 15\%$ ) were observed on deep reefs in Andros (ID#2) and Yucatan (ID#7) (Fig. 9A) while the highest levels on shallow reefs occurred in Akumal/Xcalak (ID#8). Low recent mortalities ( $\sim < 1\%$ ) were observed in Veracruz (ID#6), Bonaire (ID#17), and the Abrolhos (ID#20) for deep reefs (Fig. 9A) and in Veracruz, San Salvador (ID#3), and Cayman (ID#12) for shallow reefs. Recent mortality levels were not correlated between shallow and adjacent deep reefs within the same survey (Tables 3A, 3B).

Old partial mortality (excluding the standing dead corals) averaged 22% for deep reefs and 27% for shallow reefs (Tables 3A, 3B). Deep reefs in the Flower Gardens (ID #5) and Veracruz (ID#6) had the lowest reported old mortality ( $\sim < 10\%$ ) (Fig. 9B) while shallow reefs in the TCI (ID#4), Belize (ID 10), and the Virgin islands (ID#13) had the highest old mortalities ( $\sim > 40\%$ ). Standing dead corals (expressed as a percentage of the total number of colonies assessed) were significantly more common ( $p < 0.05$ ,  $n = 10$  assessments) on shallow reefs (14%) than on deep reefs (4%) largely because of the



**Figure 8.** Comparison (as means and standard deviations) of the in deep (> 5 m) sites in all 20 AGRR surveys: **(A)** recent partial colony mortality; **(B)** old partial colony mortality; **(C)** total partial colony mortality (including standing dead); **(D)** *Montastraea annularis* complex total partial colony mortality (including standing dead). The star indicates a parameter that was not measured and the dashed line indicates the mean for all surveys. See Table 1 for ID codes.



**Figure 9.** Size frequency distributions for six commonly observed coral species created by combining data (shallow and deep) from all assessments: (A) *Montastraea annularis*; (B) *Montastraea faveolata*; (C) *Montastraea cavernosa*; (D) *Siderastrea siderea*; (E) *Diploria strigosa*; and (F) *Acropora palmata*. Also plotted for each size class is partial old colony mortality, with the scale indicated on the second Y axis. Note the significant increase in partial mortality with increasing size, particularly within the smaller size classes, in each of these species.

abundance of dead colonies of *Acropora palmata*. Areas in which many of the shallow colonies were standing dead included Belize (ID#10) and San Salvador (ID#3) with 17% and 25% respectively, and Los Roques (ID#16) with 52% (Tables 3A, 3B). Total mortality (including colonies that were standing dead) averaged 28% of the colony surfaces in deep reefs (Fig. 9C), and 39% in the shallow reefs

Mortality levels (recent and old) varied dramatically among the individual coral taxa (Table 4). *Agaricia tenuifolia* had the highest recent partial mortality (>16% of colony surfaces affected) followed by *Montastraea franksi* (8%). *Agaricia tenuifolia* also had the highest old partial mortality (38%) followed by *M. annularis* (33%) (Table 4). Species with the largest number of standing dead colonies were *Acropora palmata* (32%), *Agaricia tenuifolia* (29%), and *Acropora cervicornis* (18%). On average, these three species had three times or more the amount of standing dead corals compared to the other species in which standing dead was recorded. Total mortality values in the *M. annularis* complex on the deep reefs (Fig. 9D) were greatest in Abaco (ID#1) and San Salvador (ID#3) and lowest in Costa Rica (ID#19).

For many of the taxa, smaller corals (~30 cm diameter) showed either no mortality or 100% mortality (i.e., they were completely dead), whereas larger corals (>60 cm diameter) most commonly displayed partial-colony mortality. Recent mortality showed no significant correlation with colony diameter; however, old mortality levels generally increased with increasing colony size through the smaller size classes. Thus increases in old mortality were most evident in the lower size classes up to the mode of the population, above which fluctuations were more apparent (Figure 8).

*Disease, bleaching, overgrowth, and predation.* Approximately 6% of the surveyed corals from all sites combined were infected with an identifiable disease. Signs of disease were observed in at least 20 coral species and, as illustrated in Table 4, many diseases were not species-specific but affected a variety of species. *Stephanocoenia intersepta* and *Acropora cervicornis* had the highest prevalence (21 and 13%, respectively) while the spatially dominant *M. annularis* species complex also had high occurrence of disease (~10%). The most commonly observed disease was white plague, followed by black-band, yellow-blotch (yellow-band), white-band, dark spots, and white pox (patchy necrosis) (Table 4). Black-band was most commonly observed in *M. franksi*, white plague in *M. faveolata* and *M. annularis*, white-band in *Acropora cervicornis* and *A. palmata*, and yellow-blotch in *M. annularis*. In some cases, the disease type was not identified or specified for several species with apparently high prevalences of disease (e.g., *A. cervicornis*, *S. intersepta*, *A. palmata*). Areas with the highest prevalence of disease included Andros (ID#2) and Akumal/Xcalak (ID#7), where 18% and 14%, respectively, of all deep (>5m) colonies were infected (Table 3B). These areas were assessed during August, 1998 and March, 1999. Prevalences of yellow-blotch were relatively more common in the southern Caribbean (e.g., in Curaçao, Bonaire, and Los Roques).

An average of 6% of the surveyed corals from all sites combined showed signs of bleaching, most often occurring as “partly bleached” or “pale”. Four areas reported greater than 20% of colonies bleached (ID#s 10, 13, 14 and 17), while only four areas had less than 2% bleached (ID#s 4, 12, 18 and 20). Differences in the amount of bleaching

and disease varied among taxa (Table 4). *Montastraea annularis* species complex (13-15% of colonies) and *Millepora complanata* (14%) exhibited the highest prevalence of bleaching. Bleaching was also often observed in *Diploria* spp., *Siderastrea siderea*, and *Acropora palmata*.

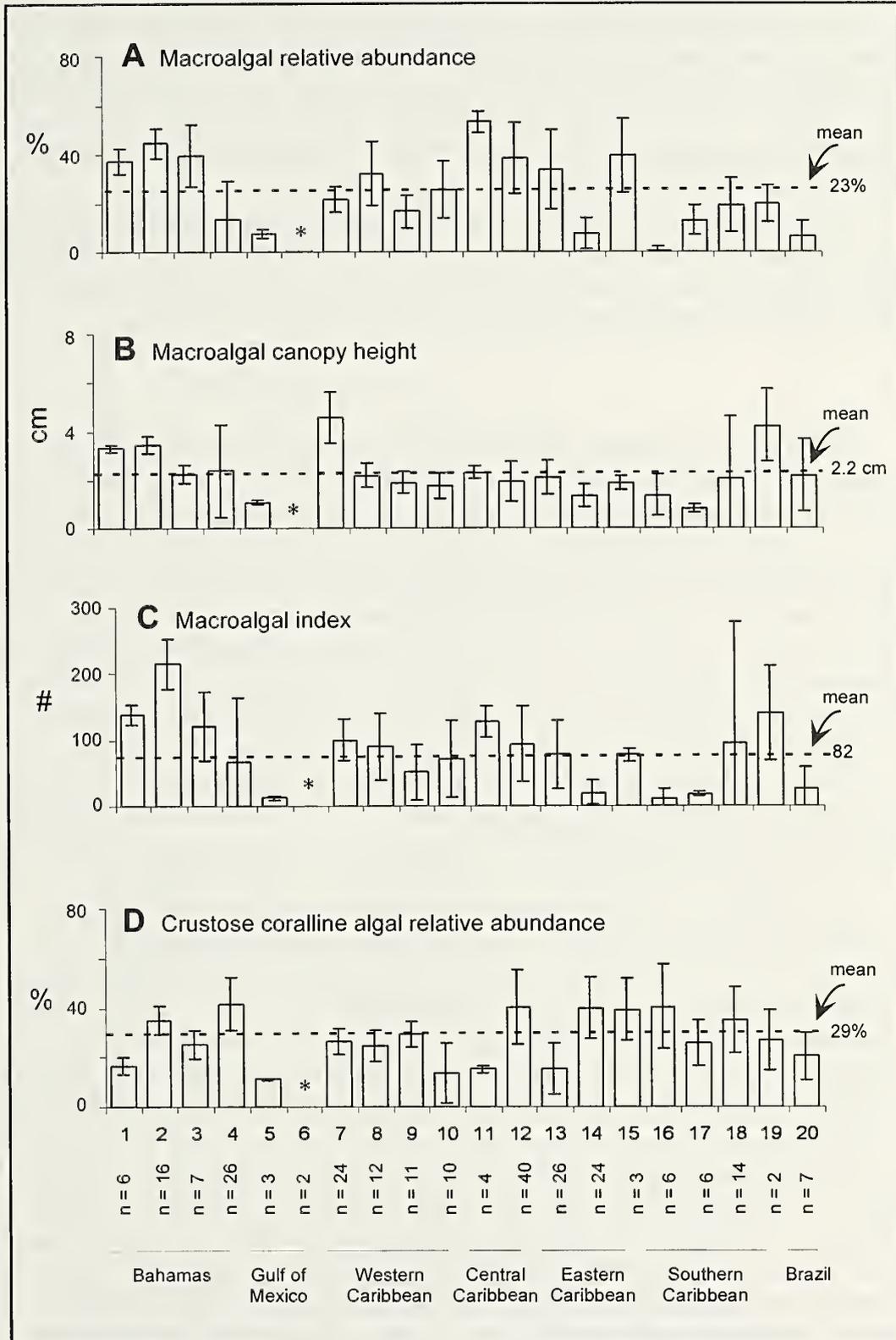
Macroalgal overgrowth was reported on an average of 4% of the colonies assessed for the entire region. Unusually high occurrences (>30% of all colonies) of macroalgal overgrowth were recorded in Lighthouse Atoll, Belize (ID#9) and Costa Rica (ID#19). However in 9 of the 17 deep surveys for which overgrowth was noted, fewer than 2% of the corals were affected. The percentage of colonies with evidence of tissue mortality due to fish bites was highly variable, being relatively high ( $\geq 10\%$ ) in Costa Rica, Lighthouse Atoll and the Virgin Islands (ID#13). Recent mortality caused by predation by snails (primarily *Coralliophila abbreviata*) or worms (*Hermodice carunculata*) was highest in shallow sites (e.g., Akumal/Xcalak (ID#8), Virgin Islands and Los Roques (ID#16)). Occurrences of damselfish algal gardens (as the percentage of affected colonies) were also highly variable, in part because of inconsistent reporting, but were recorded on an overall average of 6% of the corals. The highest occurrences (>40%) were at Lighthouse Atoll and Mária La Gorda (ID #11), while Costa Rica and the Turks and Caicos (ID #4) were the only assessments for which no damselfish algal gardens were reported (Tables 3A, 3B).

#### Algae and *Diadema*

Algal communities on deep reefs were composed, on average, of turf algae (48% relative abundance) with crustose coralline algae (29%) and macroalgae (23%) being relatively less abundant. Deep reefs with highest values for turf algae (>70%) included the Flower Gardens (ID#5) and the Abrolhos (ID#20) whereas those with the lowest (<21%) included Andros (ID#2), Cayman (ID#12) and St. Vincent (ID#15). Elevated (>35%) macroalgal relative abundances were found at three areas in the Bahamas (ID#1, 2, 3) and several areas in the central and eastern Caribbean (ID#s11-13, 15) (Fig. 10A). Macroalgal relative abundance was low (~5 %) in the Flower Gardens, windward Netherlands Antilles (ID#14) and Abrolhos and rare (<1 %) in Los Roques (ID#16). Shallow reefs ( $\leq 5$  m) had similar algal compositions (turfs>crustoses>macroalgae), although between-site variability was considerably higher.

Macroalgal canopy heights were similar in deep (2.2 cm) and shallow (2.3 cm) reefs when averaged for the region. On deep reefs (Fig. 10B), the highest (>4 cm) canopy heights were recorded in Costa Rica (ID#19) and the Yucatan (ID#7), and the lowest (<1 cm) were in the Flower Gardens (ID#5) and Bonaire (ID#17). When macroalgal canopy height and relative abundance are considered together as a macroalgal index, there was a range of nearly 20-fold (Fig. 10C) between the lowest indices (<12) in Los Roques (ID#16) and the highest index (>200) in Andros (ID#2).

The range of values for the relative abundance of crustose coralline algae was moderate (~30%) for deep sites and in no cases exceeded 50% (Fig 10D). The macroalgal:crustose coralline ratios (not shown) averaged 1.1 (range 0.1 – 3.5) with Bonaire, Curaçao (ID#18), Los Roques, TCI (ID# 4), and the windward Netherlands Antilles (ID#14) all having relatively low (<0.6) ratios. A weak, non-significant



**Figure 10.** Comparison (as means and standard deviations) of algae in deep (> 5 m) sites in the 20 AGRRA areas: **(A)** relative abundance of macroalgae; **(B)** macroalgal canopy height; **(C)** macroalgal index, and **(D)** relative abundance of crustose coralline algae. The star indicates a parameter that was not measured and the dashed line indicates the mean for all surveys. See Table 1 for ID codes.

relationship was found between the percent live stony coral cover and the relative abundance of macroalgae ( $p=0.17$ ,  $r^2=0.12$ ). Statistically significant relationships were found between canopy height and both live coral cover ( $p<0.01$ ,  $r^2=0.37$ ) and large coral density ( $p<0.01$ ,  $r^2=0.38$ ) (Fig. 11). Canopy heights were also positively related to the density of *Siderastrea siderea* ( $p=0.015$ ,  $r^2=0.33$ ) and negatively related to the density of *Colpophyllia natans* ( $p=0.03$ ,  $r^2=0.29$ ).

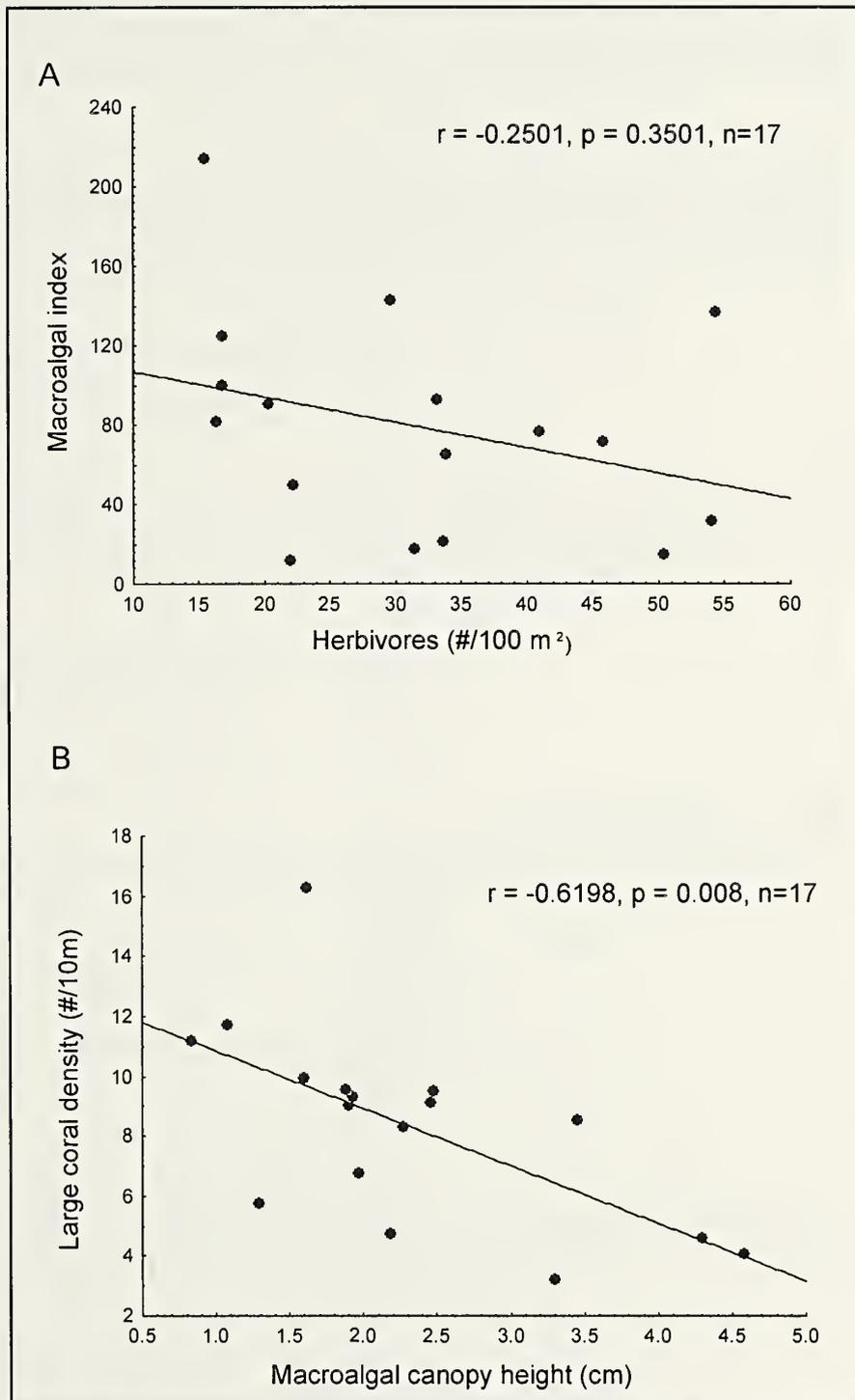
Densities of *Diadema antillarum*, both within and among surveys, were highly variable, with no individuals reported in about half (8/17) of the areas for which data are available (Table 1). Mean urchin density for all surveys reported in this volume was  $2.9/100\text{ m}^2$  with the highest densities ( $23/100\text{ m}^2$ ) occurring in Costa Rica.

## Fishes

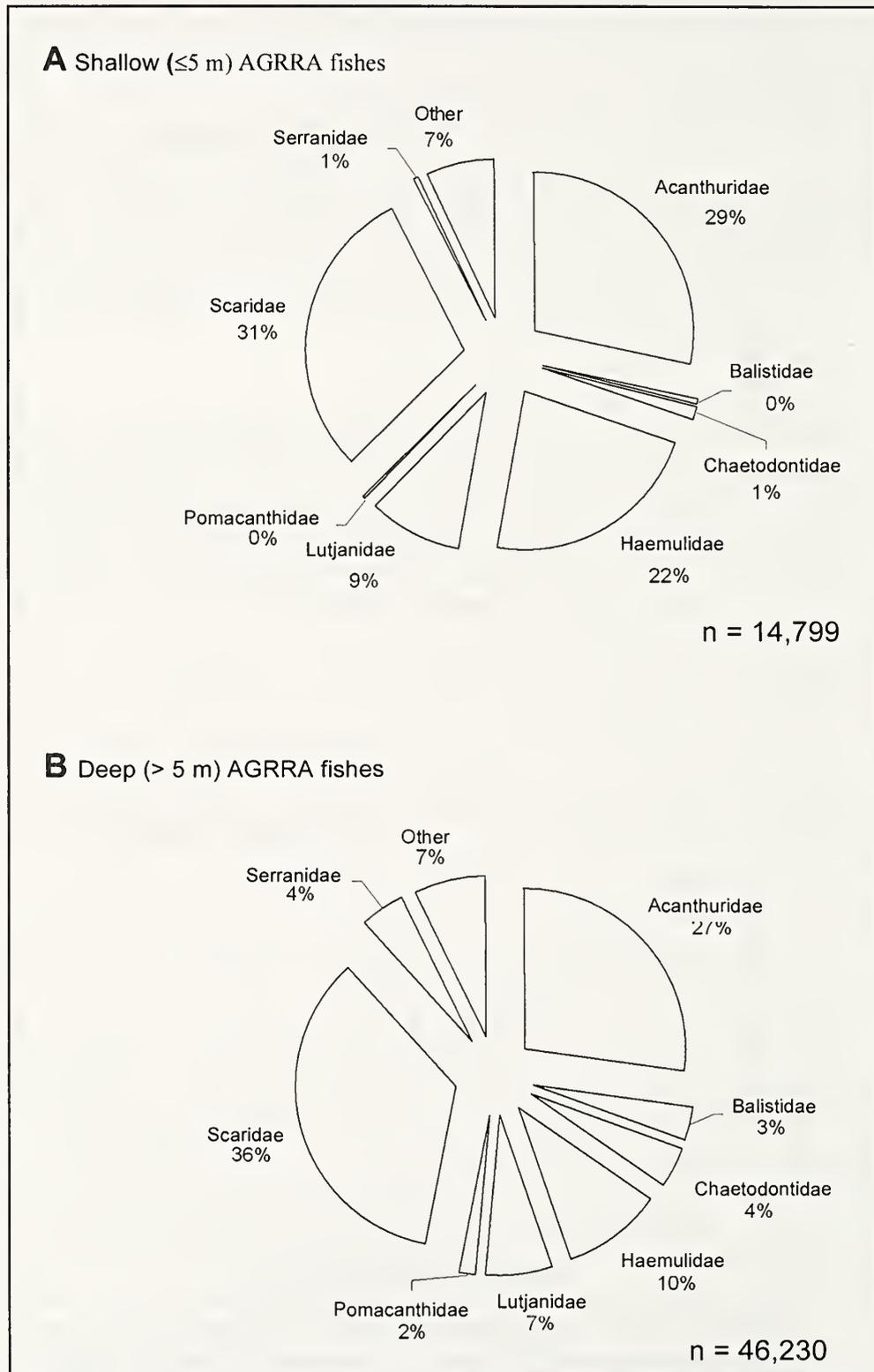
A total of 71,102 fishes were sighted in belt transects consisting of 49,888 individuals in the deep sites and 21,214 in the shallow sites (Fig.12). Of these, a total of 10,073 (14%) were surveyed in Brazil (See Kikuchi et al., this volume). For the non-Brazil sites, the most abundant of the AGRRA families were acanthurids and scarids, followed by haemulids, lutjanids, pomacanthids, balistids, and serranids. The results of the multidimensional scaling (MDS) ordination based on mean AGRRA species abundance recorded at deep sites for the 17 belt-transect assessments is shown in Figure 6B. Areas with the least similarity in fish community structure [Abrolhos (ID#20), followed by Costa Rica (ID#19), Abaco (ID#1), Los Roques (ID#16), St. Vincent (ID#15), and the Flower Gardens (ID#5)] are mostly characterized by unusual environmental conditions. Cluster analysis revealed three groups, each with strong similarity (Bray-Curtis similarity of greater than 60%, areas shaded gray in Figure 6B), corresponding to areas in the southern Caribbean, parts of the Bahamas and the western and central Caribbean, and the central-eastern Caribbean regions, respectively.

Total AGRRA fish densities were nearly twice as high in shallow ( $85/100\text{ m}^2$ ) as in deep ( $49/100\text{ m}^2$ ) sites, mainly as a result of higher densities of haemulids and acanthurids in  $\leq 5\text{ m}$ . For deep sites, comparisons among the 17 areas with comparable data records indicate that fish densities were highest in the Abrolhos (ID#20), followed by Los Roques (ID#16) and Curaçao (ID#18), while the lowest values were found off Andros (ID#2) (Fig. 13A). Herbivores dominated the deep fish assemblages (average of  $35/100\text{ m}^2$ ) at approximately six times the density of carnivores (average of  $6/100\text{ m}^2$ ) (Fig. 13B,C). Excluding data from the Abrolhos, total fish biomass (not shown) displayed a five-fold difference among surveys, ranging from  $2,600\text{ g}/100\text{ m}^2$  in Akumal/Xcalak (ID#8) to  $12,640\text{ g}/100\text{ m}^2$  in Los Roques. Using the Bryant et al (1998) *Reefs at Risk* overexploitation threat classification (see Table 1), total AGRRA fish biomass was  $6,459\text{ g}/100\text{ m}^2$  for the “high exploitation” reefs,  $5,943\text{ g}/100\text{ m}^2$  for reefs under “intermediate exploitation”, and  $6,846\text{ g}/100\text{ m}^2$  for reefs experiencing “low exploitation”. No significant difference in total fish biomass could be detected among the three different exploitation-threat categories using a 1-way ANOVA analysis.

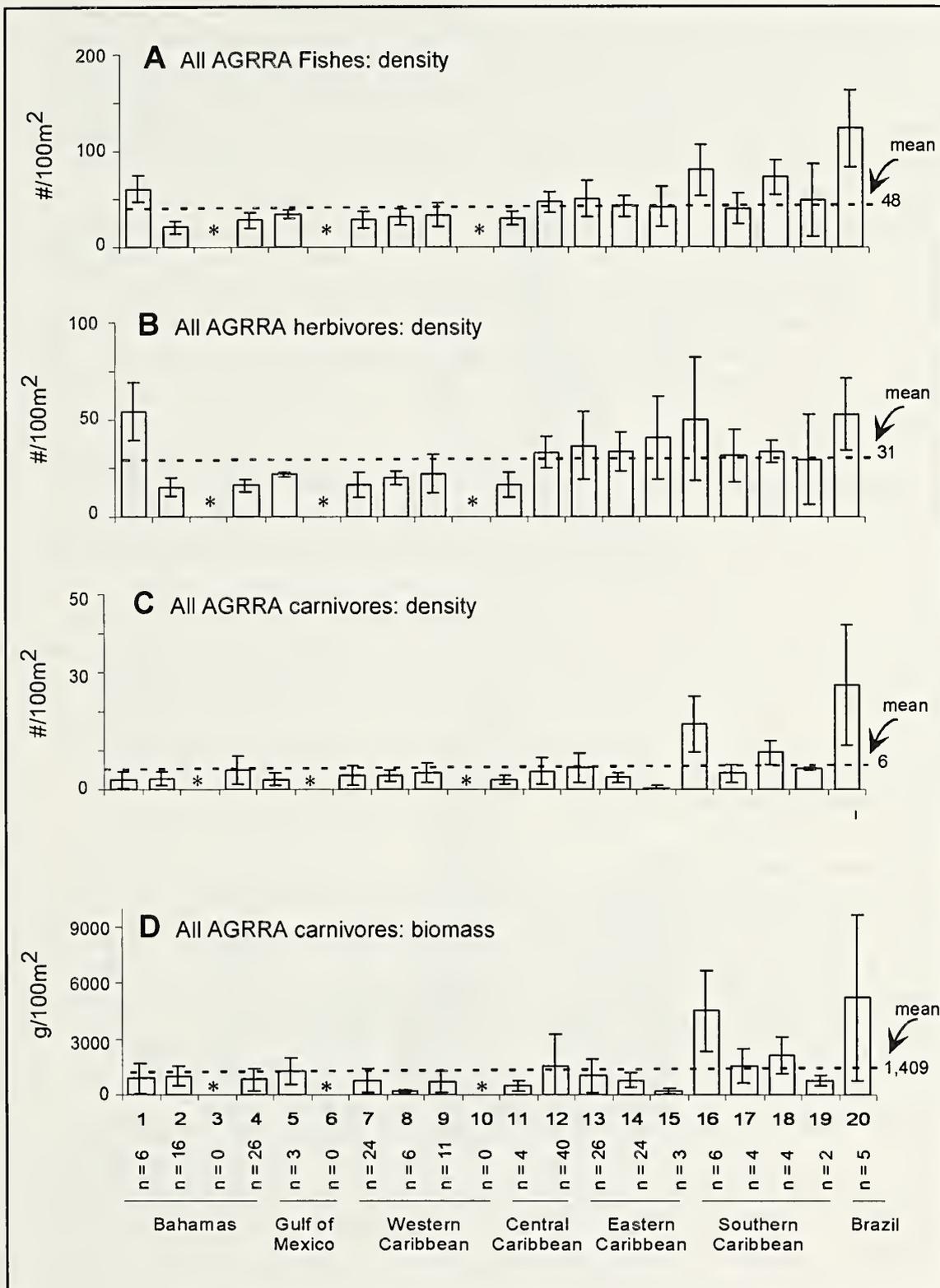
*Herbivores.* Deep reefs in Abaco (ID#1), the Abrolhos (ID#20), and Los Roques (ID#16) all had higher than average densities of herbivorous fishes, whereas lower than



**Figure 11.** Regression plots of fish, coral, and algal indicators: **(A)** mean herbivore density (acanthurids, scarids  $\geq 5$  cm and *Microspathodon chrysurus*) and mean macroalgal index for 17 assessments; **(B)** algal canopy height and coral density (stony corals  $\geq 25$ cm diameter/per transect).



**Figure 12.** Composition of AGRRA fishes for major families for all assessments except Brazil (ID #20) combined for (A) shallow sites ( $\leq 5$  m), and (B) deep sites ( $> 5$  m). Includes only fishes sampled within a maximum of 10 transects per site.



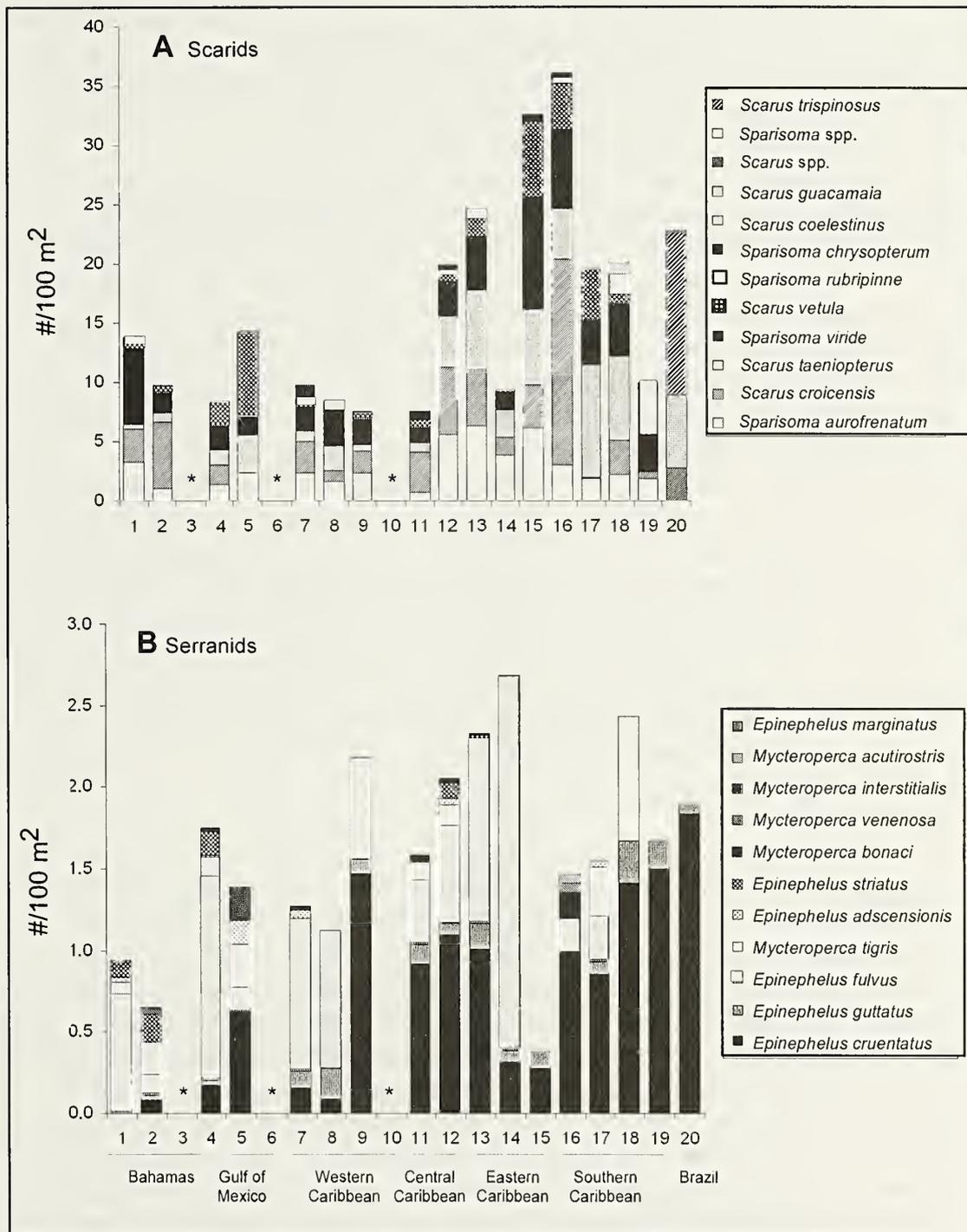
**Figure 13.** Comparison between means and standard deviation of deep sites (>5 m) for 17 AGRRA assessments: **(A)** all AGRRA fishes, **(B)** herbivorous fishes (acanthurids, scarids  $\geq 5$  cm and *Microspathodon chrysurus*); **(C)** carnivorous fishes (serranids, lutjanids, haemulids  $\geq 5$  cm). The star indicates a parameter that was not measured and the dashed line indicates the mean for all surveys.

average densities were found in Andros (ID#2), the Turks and Caicos (ID#4), Yucatan (ID#7), and Mária la Gorda (ID#11) (Fig. 13B). Herbivore biomass (not shown) ranged from a high of 7,484 g/100 m<sup>2</sup> in Costa Rica (ID#19) to a low of 1,508 g/100 m<sup>2</sup> in the Turks and Caicos). *Microspathodon chrysurus* (yellowtail damselfish) had a mean density of 2.2/100 m<sup>2</sup> with greatest numbers (>4/100 m<sup>2</sup>) reported in Los Roques, Lighthouse Atoll (ID#9), and Costa Rica and the lowest (~0.1/100 m<sup>2</sup>) in Andros. The density of acanthurids for all surveys combined averaged 11.8/100 m<sup>2</sup>. Acanthurids were widespread throughout the region and all three species, *Acanthurus coeruleus* (blue tang), *A. chirurgus* (doctor fish) and *A. bahianus* (ocean surgeon), were present in all 17 areas surveyed. *A. coeruleus* was most abundant (averaging 5.7/100 m<sup>2</sup>), with a high of 30.7/100 m<sup>2</sup> in Bonaire (ID#17), followed by *A. bahianus* (average = 5.3/100 m<sup>2</sup>), which reached a high of 8.7/100 m<sup>2</sup> in the Virgin Islands. *Acanthurus chirurgus* was least abundant (average of 1.0/100 m<sup>2</sup>) except for the Abrolhos where its density (13.7/100 m<sup>2</sup>) was unusually high.

Scarids had a mean density of 13.7/100 m<sup>2</sup> and were most abundant in the eastern and southern Caribbean. The highest scarid densities (36.0/100 m<sup>2</sup>), which occurred in Los Roques, were nearly five times those reported for Mária La Gorda) and Lighthouse Atoll (Fig. 14A). Parrotfish species composition was similar across the region except for the Abrolhos which contained a Brazilian endemic, *Scarus trispinosus* (greenlip parrotfish), not found elsewhere in the western Atlantic (Fig. 14A). *Scarus croicensis* (striped), *Sparisoma aurofrenatum* (redband), *Scarus taeniopterus* (princess), *Sparisoma viride* (stoplight) and *Scarus vetula* (queen) were the five most abundant parrotfish species overall, with mean densities of 3.8/100 m<sup>2</sup>, 3.6/100 m<sup>2</sup>, 3.1/100 m<sup>2</sup>, 2.8/100 m<sup>2</sup>, and 1.1/100 m<sup>2</sup>, respectively. Large-sized parrotfishes, including *Scarus guacamaia* (rainbow), *Scarus coelestinus* (midnight), and *Scarus coeruleus* (blue), were observed only occasionally and were more common in the southern Caribbean than in other subregions.

Regression analysis showed no significant relationship ( $p > 0.05$ ,  $n = 17$ ) between macroalgal index and herbivore density at the scale of the entire region (Fig. 11A). Similarly, no significant relationships were found between herbivore density and any other algal indicator (e.g., macroalgal canopy height or relative abundance of macroalgae, turfs or crustose corallines). Regionally, total herbivore density or biomass showed no relationship with coral density or with such measures of habitat complexity as coral diameter or height, although significant relationships were found between individual scarid and acanthurid species. For example, densities of *S. vetula* (queen) and *S. taeniopterus* (princess) parrotfishes were positively related to live coral cover ( $r^2 = 0.8$ ,  $p < 0.05$ ,  $n = 17$ ;  $r^2 = 0.6$ ,  $p < 0.05$ ,  $n = 17$  respectively). The density of *A. chirurgus* (doctor fish) was negatively correlated to old partial mortality ( $r^2 = -0.53$ ,  $p < 0.05$ ,  $n = 17$ ) while that of *A. bahianus* (ocean surgeon) was negatively related to large coral density ( $r^2 = -0.49$ ,  $p < 0.05$ ,  $n = 17$ ).

*Carnivores.* The density of carnivorous fishes averaged 6.0/100 m<sup>2</sup> for the region and was similar in shallow and deep reefs. Lutjanids predominated the carnivore assemblages overall with nearly twice the densities of serranids. Proportionally more serranids occurred in deep reefs than in shallow, although the reverse was true for



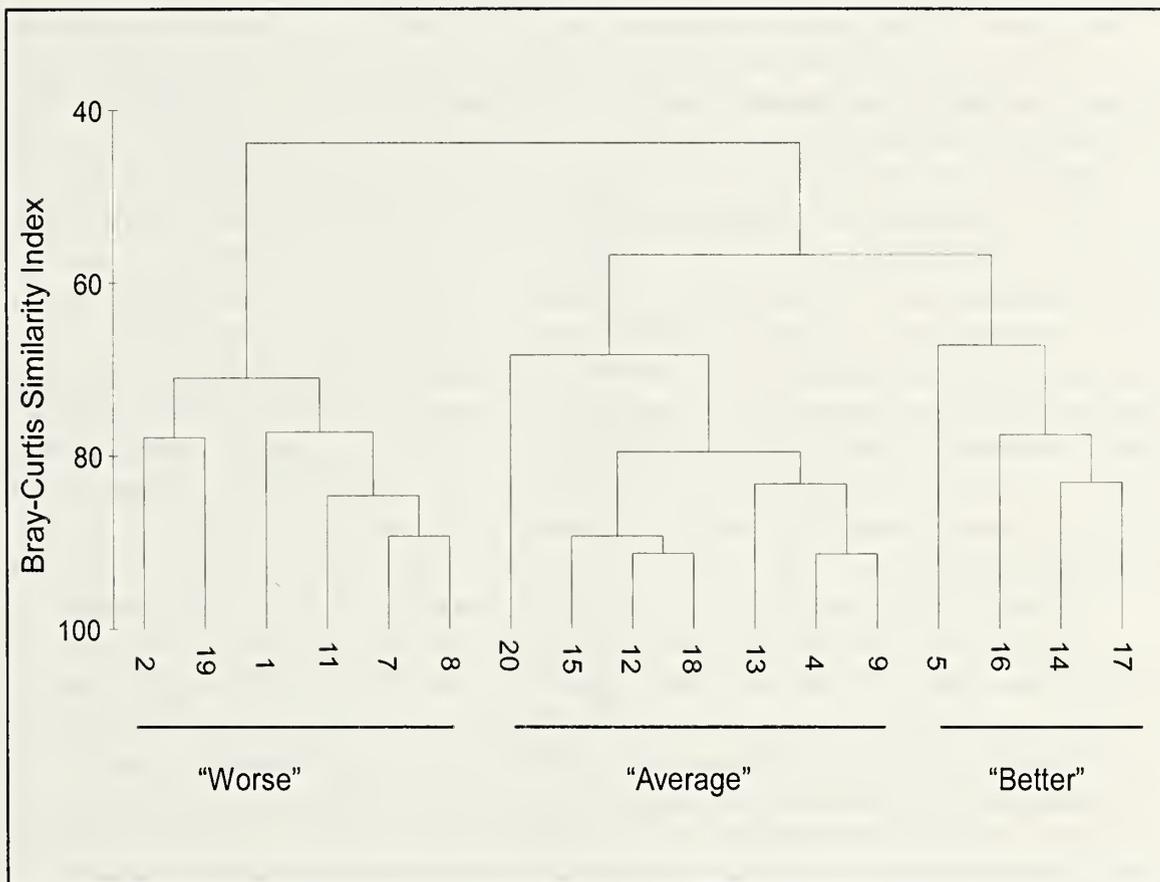
**Figure 14.** Stacked bar plot of fish family composition by survey (deep sites only) (A) scarid and (B) serranids. The star indicates a parameter that was not measured.

lutjanids. When comparisons are limited to the deep reefs (Fig. 13C), the areas with highest carnivore densities were the Abrolhos (ID#20) and Los Roques (ID#16), while the lowest were found in St. Vincent (ID#15) and Mária La Gorda (ID#11). Excluding the Abrolhos, carnivore biomass displayed a 40-fold difference across all other assessments, ranging from a low of 191 g/100 m<sup>2</sup> in Akumal/Xcalak (ID#8) to a high of 4,515 g/100 m<sup>2</sup> in Los Roques (Fig. 13D). No significant difference ( $p = 0.84$ ,  $n = 3$ ) was detected in total carnivore biomass between highly threatened reefs (~1,100 g/100 m<sup>2</sup>) and either medium- (1,204 g/100 m<sup>2</sup>) or low-threat reefs (~1,280 g/100 m<sup>2</sup>).

The most abundant lutjanids were *Ocyurus chrysurus* (yellowtail snapper), *Lutjanus apodus* (schoolmaster snapper), and *L. mahogoni* (mahogany snapper) at densities of 1.6/100 m<sup>2</sup>, 0.8/100 m<sup>2</sup>, and 0.4/100 m<sup>2</sup>, respectively. Densities of *Ocyurus chrysurus* recorded in the Abrolhos (ID#20) were one of the highest (~23.1/100 m<sup>2</sup>) for any species. Serranids were present in all surveys at densities of less than 2/100 m<sup>2</sup> and were most abundant in the windward Netherlands Antilles (Fig. 14B). In all surveys, except Andros (ID#2) and the Abrolhos, smaller-bodied species including *Epinephelus cruentatus* (graysby), *E. guttatus* (red hind), *E. adscensionis* (rock hind), and *E. fulvus* (coney) dominated the grouper assemblages. The density of coneys decreased substantially in surveys conducted in the southern Caribbean where graysbys were proportionally more common (Fig. 14B). The most commonly seen large-bodied grouper was *Mycteroperca tigris* (tiger) with a mean abundance of 0.4/100 m<sup>2</sup>. *Epinephelus striatus* (Nassau grouper) was rarely seen except in the Bahamas subregion with Andros having the highest density (0.2/100 m<sup>2</sup>). Single [*E. marginatus* (dusky), *Mycteroperca rubra* (comb)] to no [*Epinephelus itajara* (goliath), *E. morio* (red), and *M. phenax* (scamp)] sightings were recorded in transects for a number of serranids on the AGRRA list. *Sphyraena barracuda* (great barracuda) was recorded in 10 of the 17 surveys, although usually at very low numbers (<0.2/100 m<sup>2</sup>).

### Index of Reef Health

To evaluate the overall condition of each survey on a relative scale, a preliminary biotic reef health index was developed for the deep (> 5 m) dataset utilizing 13 of the AGRRA indicators: live coral cover, large ( $\geq 25$  cm) coral density, small coral density (Fig. 7D), maximum diameter of the *Montastraea annularis* complex, recent partial mortality, old partial mortality, total mortality (including standing dead), prevalence of coral diseases, macroalgal index, relative abundance of crustose coralline algae, *Diadema* density, herbivorous fish density, and carnivorous fish density. Only 17 areas could be analyzed because of missing [Veracruz (ID#6)] or incomparable [(San Salvador (ID#3) and Belize (ID#10), where fish were quantified in cylinders)] fish data. A similarity matrix was calculated and analyzed using a Bray-Curtis similarity cluster dendrogram (Fig. 15). Three broad classes, distinguishable at the 70% similarity level, were labeled as “worse,” “average,” and “better” health categories. Only four areas fell into the “better” grouping: the Flower Gardens (ID#5), windward Netherland Antilles (ID#14), Los Roques (ID#16), and Bonaire (ID#17), each of which was characterized by low recent mortality, low prevalence of diseases, low macroalgal index, and relatively high fish densities. Seven areas fell into the “average” grouping and most of these had the majority of their



**Figure 15.** Bray Curtis similarity diagram of proposed reef health classification based on 13 AGRRA indicators. See Table 1 for ID codes and text for discussion.

indicators near the regional norms. Surveys in the most unfavorable class (“worse”) were Abaco (ID#1), Andros (ID#2), Yucatan (ID#7), Akumal/Xcalak (ID#8), Mária La Gorda (ID#11), and Costa Rica (ID#19). These areas were generally typified as having high recent mortality, high prevalence of disease, moderate-to-high macroalgal index, and low fish densities.

## DISCUSSION

### Sampling Issues

*Sampling effort.* Differences in the sampling effort carried out for each AGRRA assessment resulted in part because of differences in the areal extent each team attempted to characterize as well as logistical sampling constraints (e.g., time, support, and accessibility). The fewest sites (two) were for the Flower Garden Banks (ID#5), which was also the deepest (~20 m) and furthest offshore of the reef areas studied. The greatest effort was for the Cayman Islands (ID#12) where a total of 42 sites were assessed. Determining how much effort to allocate towards characterizing sites depends on the indicator and on its variance at different spatial scales. Results of the power analyses at the site scale suggest that, for a number of AGRRA benthic indicators, the greatest gain in terms of precision versus effort occurs within the first six transects which corresponds to roughly 60 corals and 40 quadrats. For the more specific indicators of rare species (i.e. abundance of large-bodied groupers, size frequency distribution of some corals), higher overall sampling would be necessary to get an adequate sample size to test significant differences.

The coefficients of variation (CV), defined as the standard deviation divided by the mean), for 12 of the AGRRA indicators at four spatial scales (deep sites only) are given in Table 5. For most indicators, the highest CV occurs at the smallest spatial scale (<0.1 km), however, there is still significant variation at the area and subregional scales (~1-100 km). This inverse relationship between CV and spatial scale suggests that in terms of sampling effort either slightly more effort should be allocated at the site scale (i.e., number of transects sampled per site) in comparison to the larger scales (i.e., number of sites sampled per area and number of areas sampled per subregion) and/or to observer training (see below). It also indicates that the size of the regional “signal” for some indicators (e.g., partial old mortality; herbivore fish abundance) is quite small but comparatively large for others (partial recent mortality, macroalgal canopy height, carnivore abundance).

*Site selection and habitat classification.* Considerable variance in AGRRA indicators at all but the smallest spatial scales can arise by sampling slightly different reef types. In this initial synthesis, only two categories of reefs were recognized (deep and shallow) and all sites in the deep category were used because the purpose was to compare the assessments presented in the volume. A majority of the deep sites were located on fore-reef slopes but some were located in other habitats. Habitat variation clearly has an influence on some AGRRA indicators (e.g., coral community composition, standing dead

coral) and these sites should be factored out in future comparisons. Reef slopes themselves can also have different morphologies that influence fish densities and other AGRRA parameters. For example, fore-reef slopes around the Turks and Caicos (ID#4) and on the leeward sides of Los Roques (ID#16) and Bonaire (ID#17) are characterized by sharp drops (walls) at depths of approximately 10m (the depth at which many surveys were undertaken). These “edges” tend to attract fishes at higher abundances compared to adjacent zones both shallower and deeper. Site selection and classification are even more critical for shallow-reefs where large differences in community coral composition, fish abundance, and algal parameters can occur across very small spatial scales (<100m). I have observed that reef habitats surrounded by areas of deep water (or similar sites with high relief surrounded by vast areas of no relief) will tend to attract fishes (“oasis effect”) and have higher fish densities than areas wherein habitat is more evenly distributed. In the Abrolhos (ID#20), the unusually high fish densities are attributed, in part, to the shallowness and small size of the reefs. As the number of AGRRA assessments in the database grows, geomorphic and structural complexity characteristics should also be used to help classify reef types and design more representative sampling strategies for field assessments.

*Methodology and observer bias.* Observer bias is a potential contributor to variance in the benthic and fish data and needs to be addressed in any study that relies on more than a single observer to collect data. Many of the participants contributing to this synthesis had considerable experience with Caribbean reefs and, with few exceptions, one or more of the observers in each team had received formal training in the AGRRA protocols. Based on our workshop experiences, variance among observers who have undergone consistency training is small compared with the total variance at all spatial scales for any given indicator. When observers have not undergone such training, a number of the AGRRA indicators can be scored systematically in ways that lead to bias in the data. This is particularly true for fish where small differences in belt-transect width, length and swim time can lead to systematic differences in fish density estimates (Sale, 1998). Several of the surveys in this volume used different transect sizes or even different methods. Given the costs and time to collect AGRRA data, particularly from remote locations, it is recommended that the protocol be followed more closely in future field efforts and that more effort be spent on minimizing observer bias through consistency training of all observers with core AGRRA personnel.

## Stony Corals

*Community structure.* A mean live stony coral cover of 26% for deep sites is considered fairly representative of the region during 1998-2000, although perhaps somewhat elevated since the most structurally complex sites were sometimes selectively targeted by AGRRA assessors. Historic baselines for coral cover do not exist in most areas, but a pre-1980 regional estimate based on available coral cover data was summarized by Gardner (2002) at between 40 and 50%. Thus Western Atlantic reefs have undergone significant losses over the past several decades yet despite many large-scale disturbances (e.g., the 1983-84 die-off of *Diadema antillarum*, numerous outbreaks of

disease and bleaching events) a fair amount of living coral remains. In some areas, however, and not reflected in these percentages, the original large corals that broadcast larvae have likely been replaced by smaller brooding species.

More substantial changes have probably occurred in the shallow (<5 m) reef crests and patch reefs where faster-growing acroporid corals once dominated (e.g., Goreau, 1959; Geister, 1977; Adey, 1978; Gladfelter, 1982). Results of the AGRRA assessments for the shallow reefs presented in this volume indicate an overall live coral cover of 18%; however, the small and spatially disproportionate sampling and high variability in reef types that were assessed preclude using this average as representative of the region. AGRRA assessments since 2000 have been directed specifically to more systematically sample shallow reefs.

For shallow and deep reef sites, the composition indicates typical species dominance patterns on western Atlantic reefs and reflects the amount of survey effort for a particular reef type. Coral species composition and abundance data for the 20 areas are influenced by: (1) environmental and/or biogeographic factors; (2) reef types that were assessed; and (3) their disturbance histories. One of the two clusters in the MDS analysis of the deep reefs is caused in part by the geographic location of its areas (ID#s 2, 4, 11, 12, 13) in the insular Caribbean (Fig. 6A). Outliers represent geographic extremes or areas that are currently marginal for reef growth. For example: Abaco (ID#1), the furthest north, is exposed to large Atlantic swells and to cooler water temperatures; Abrolhos (ID#20), the furthest south, is in a different biogeographical subregion and also experiences open ocean swells; both Costa Rica (ID#19) and Veracruz (ID#6) are heavily influenced by sediment runoff; and the Flower Gardens (ID#5), the deepest of the assayed reefs and somewhat isolated from other reef systems in the northwestern Gulf of Mexico, lack several common Caribbean scleractinian species (Roberts et al., 2002). The distinctiveness of the deep reefs in San Salvador (ID#3) (compared to other deep reefs) is thought to result from the way sites were grouped using the 5m cut off depth, which resulted in four "other" reef types being grouped with three reef-slope sites.

The abundance of "recruits" (estimated as small corals with diameters of  $\leq 2$  cm) is an important indication of a reef's potential for growth and for recovery after major disturbances. The total area sampled at each site ( $\sim 2\text{-}3\text{ m}^2$ ) was probably too small for these data to be highly reliable at the site scale. Patterns at the subregional level were not evident, but larger scale comparisons indicate that corals are currently recruiting most successfully in the Abrolhos (ID#20) and on the deep reefs in Andros (ID#2) and the Virgin Islands (ID#13). That the AGRRA data shows species composition of coral recruits often did not reflect the major coral-reef builders present is similar to other studies (e.g., Rogers et al., 1986).

*Mortality.* The AGRRA data support previous findings that show partial mortality differs among coral species and partially reflects the life history strategies and population dynamics of species and their susceptibility to various physical and biological factors (Bak and Luckhurst, 1980; Hughes and Jackson, 1980, 1985; Meesters et al., 1997). Large, long-lived broadcasting corals such as *Montastraea* spp. tend to exhibit higher amounts of partial tissue mortality while smaller, short-lived brooding species (e.g., some species of *Porites* and *Mycetophyllia*) tend to exhibit either complete mortality or no

mortality. For nearly all species, levels of old partial mortality increase with colony size (particularly in the early size classes) suggesting that corals are more likely to suffer severe, irreversible tissue damage the longer they live (e.g., Ginsburg et al., 2001). Since between-reef variability of mortality is strongly influenced by the species composition and sizes of the corals present in each reef (Bythell et al., 1993; Meesters et al., 1996), these variables should be considered when examining patterns at diverse spatial scales.

The mortality signal is also a function of how long the corals remain within the “recent” state before transitioning to the “old” state and finally to unidentifiable substrate or rubble. This rate of transformation is strongly influenced by sedimentation, bioerosion, and overgrowth all of which vary both spatially and temporally. Given these influences and geographic variations in the prevalence of disease and bleaching events, hurricanes and other disturbances in 1998-2000, it is not surprising that levels of recent partial mortality (along with the amount of standing dead coral) were highly variable at nearly all examined spatial scales (0.1-1000 km). In contrast, old partial mortality showed much lower variation at larger spatial scales ( $> \sim 0.1$  km) suggesting similar processes were affecting this indicator for the entire region.

A useful distinction can be made between background mortality of stony corals caused by chronic stressors and acute mortality caused by intermittent major disturbances (see also Steneck and Lang, this volume). Some inferences can be made about each of these sources given the range of environmental conditions that affected the reefs sampled in this volume. Chronic or background coral tissue damage and mortality can result from a variety of low-intensity stressors including predation, disease, bioerosion, sedimentation, competition and abiotic perturbations (Meesters et al., 1996, 1997). Many instances of partial mortality are small and within the abilities of corals to regenerate new tissue; however, above certain sizes ( $\sim 10\text{cm}^2$ ), dead skeletal areas tend to become overgrown or eroded by other organisms (Meesters et al., 1996; Clark, 1997). Under increasingly severe chronic conditions, especially for species with slow rates of regeneration (e.g., most massive corals), there will be increases in partial mortality followed by decreases in size structure, increases in the number of physiologically distinct colonies, and finally shifts in the species assemblages. Abaco (ID#1) and Costa Rica (ID#19) are two of the most chronically stressed reef areas assessed in this volume. The deep Abaco reefs had the highest levels of total mortality for the *M. annularis* complex (Fig. 9) coupled with the lowest densities of large ( $> 1\text{m}$ ) colonies (Table 3B). Costa Rica had proportionally more colonies of *Siderastraea*, which is well-known for its high tolerance of sediment stress (Cortés and Risk, 1985) than any of the other 19 areas, but relatively low levels of total partial mortality for the *M. annularis* complex and of partial old mortality for all stony corals. These patterns underscore that differences in partial mortality are influenced by the species composition of corals present and coral size and should be considered together when examining patterns at various spatial scales.

Establishing a present-day baseline for background coral mortality allows us to distinguish the magnitude of mortality following severe disturbance events such as disease epizootics, bleaching, or hurricanes. Based on the AGRRA data presented in this volume, our current best estimate of background (chronic) partial mortality on reefs can be approximated from the regional averages of  $< 4\%$  for recent mortality and  $< 22\%$  for old mortality. However, a more accurate estimate of chronic mortality is achieved by

removing those areas known to have been recently impacted by severe disturbance events (e.g., ID#s1,2,3,7,8,10). This results in an estimate of <2% for recent mortality and <21% for old mortality. Associating a time frame with these recent and old mortality signals would allow us to make an inference about the *rates of mortality*. Assuming the recent mortality signature (i.e., raised calices around corallite, unless calices have been removed by parrotfishes) remains visible for approximately one year, present-day rates of annual turnover (“cumulative partial mortality”) might be on the order of 2%, excluding any regeneration of soft tissues. By extension, this would imply that old mortality signatures remain visible for approximately a decade, after which coral skeletons are no longer identifiable. Although these values appear to represent a reasonable upper limit of background mortality to expect on reefs today, it is unknown how this estimate compares to historic background levels when chronic stresses may have been quite different. Hughes and Connell (1999) argue that rates of coral mortality can be naturally quite high even in the absence of major disturbance events, thus historic mortality levels may not be drastically different from those observed today. If so, this would imply that 30-50 years ago a regional average of 2% recent and ~20% old mortality may have existed, and that decreases in live coral cover may be more a function of changes in the rates of coral regeneration and recruitment rather than of changes in the rates of coral mortality.

Coral tissue damage that results in large lesions or even complete colony mortality often results from major disturbance events (e.g., disease epizootics, bleaching, hurricanes). Shortly after the disturbance, this type of acute mortality is reflected mainly in high values for recent mortality and standing dead. For example, high levels of recent mortality were related to the presence of certain coral diseases at the subregional and local scales (Table 4B) including black-band disease (Andros, Kramer et al., this volume), white plague (Akumal/Xcalak, Steneck and Lang, this volume) and yellow-blotch disease (Curaçao, Bruckner and Bruckner, this volume). Mass mortality events often affect corals regardless of size but certain species are more susceptible than others. For this reason, species-specific levels of recent mortality are probably a more reliable way of examining spatial and temporal variation in disturbance events.

The amount of standing dead coral, often overlooked in current monitoring methods, is an excellent indicator for hindcasting past disturbance events possibly exceeding a decade (provided no major hurricanes have directly affected the area in the meantime and if rates of overgrowth and bioerosion are not too high). Reliability in counting and identifying standing dead massive corals to the genus or species level depends on how much time has passed since the disturbance event and how much effort an observer puts into exposing and identifying their skeletons. The highest occurrences of standing dead coral on deep reefs were reported from Belize (ID#10) and San Salvador (ID#3), two areas strongly influenced by the 1998 ENSO (see two papers by Peckol et al., this volume). As the reefs examined included some deeper patch-reef and back-reef communities, how much this signal represents the ENSO disturbance history versus differences in reef type from those in most of the other assessments is unclear. In general, the standing dead signature is more dramatic on shallow reefs where dead *Acropora palmata* are easily identified and can persist upright for decades, particularly if they become encrusted with crustose coralline algae which makes the skeletons less susceptible to bioerosion (personal observations).

*Diseases and bleaching.* AGRRA data indicate that diseases were present throughout most of the wider Caribbean region in 1998-2000 with very few areas exhibiting no occurrences, not even on reefs removed from close human influence. These data are consistent with numerous reports suggesting that the prevalence, extent and type of diseases in Caribbean stony corals continue to increase (Richardson, 1998; Green and Bruckner, 2000; Wheaton et al., 2001). An unexpected observation from the AGRRA surveys was the high prevalence of disease in the *Montastraea annularis* complex observed on some of the deep reefs, particularly given their relatively remote locations (e.g., Cayman, Kievman et al, this volume; Andros, Kramer et al, this volume; Virgin Islands, Nemeth et al, this volume).

Outbreaks of disease (or epizootics) nearly always resulted in high spatial variability at the community and individual level and differentially affected species. On the regional scale, all size classes of corals were affected by diseases. However, on some local scales, size effects were evident. For example, higher prevalences of yellow-blotch disease were correlated with larger (>0.5 m) coral sizes in Curaçao (Bruckner and Bruckner, this volume). Spatial patterns of disease occurrence on Andros suggested that at least some diseases may be highly contagious or spread rapidly and easily. Some of the deep reefs with high prevalences of diseases, for example, of white plague and black-band off Andros (Kramer et al., this volume) and of yellow-blotch off Curaçao (Bruckner and Bruckner, this volume), also had high coral densities and complex reef structures dominated by the *M. annularis* species complex and may be more susceptible to rapid spread of a disease because of the close proximity of colonies. However, no diseases were observed on the well-developed reefs in the Flower Gardens (Pattengill et al., this volume) (although noted to exist in low amounts outside of transects) suggesting that: (1) some regions may be less exposed to diseases; (2) other stressors such as bleaching make corals particularly susceptible to diseases that are present on most reefs; and/or (3) some coral genotypes may be more resistant to diseases than others.

The AGRRA data suggest a strong linkage between bleaching-related mortality and infectious diseases which could be due to increased pathogen activity and/or sensitivity of corals to their effects during periods of elevated sea surface temperatures. Recent mortality in areas that had been affected by mass bleaching during the 1998 ENSO event was closely associated with the presence of black-band disease in Andros (Kramer et al., this volume) and white plague in Andros (ibid.) and the Yucatan (Ruiz et al., this volume; Steneck and Lang, this volume). However, these relationships are somewhat confounded since not all disease outbreaks are tied to temperature and because the impact of a disease will vary based on how long it lasts and how fast it spreads. Increased water temperatures have been related to elevated prevalence of five coral diseases (bacterial bleaching, black-band disease, white plague, aspergillosis and dark spots disease) (reviewed by Harvell et al., 1999; Rosenberg and Ben-Haim, 2002; see also Porter, 2001). Clumped distributions observed in the AGRRA surveys mentioned previously, may be due both to the intrinsic intensity of the disease and local patterns of increased sea surface temperature similar to a post-bleaching-related outbreak of black-band disease in Florida (Kuta and Richardson, 1996), whereas “normal background diseases” observed at the local scale on many of the other AGRRA surveys may have a

more random distribution pattern similar to that reported by Edmunds (1991). It is still unclear if bleaching causes corals to be more susceptible to opportunistic pathogens, and/or if pathogens normally present exacerbate levels of bleaching and bleaching related mortality. The relationships among disease, bleaching and mortality, and the temporal and spatial scales in which these processes are operating, will influence whether the effects of these events are transient or lethal (e.g., Kramer and Kramer, 2002).

*Condition of keystone corals.* The AGRRA data provide further information concerning the regional decline of *Acropora* (Aronson and Precht, 2001), which is particularly evident for *A. palmata* as standing dead signatures in several locations (e.g., Los Roques, Villamizar et al., this volume; Tobago Cays, Dechamps et al., this volume). *Acropora palmata* still predominates species composition on many shallow reefs in the region although the proportion of dead-to-living colonies varies dramatically on the local scale. Surprisingly large stands of live *A. palmata* were found off Andros, Bahamas (Kramer et al., this volume) and, more recently, on the southwestern coast of Cuba (AGRRA database, unpublished data). In contrast, *Acropora cervicornis* was present on many of the deep reefs that were assessed, but was very sparse with no thickets or haystacks even remotely resembling the stands reported from Jamaica (Goreau, 1959), Bonaire (van Duyl, 1982) and Florida (Dustan and Halas, 1987) as recently as the late 1970's to early 1980's. The fragile nature of branches of *A. cervicornis* allows it to be broken down to rubble soon after death; thus the likelihood of observing standing dead colonies is substantially lower than for other species. *Acropora cervicornis* rubble can persist for several decades (Shinn et al., in press) however, and was noted in several areas (e.g., Cayman, Curaçao).

Significant gaps still exist in our understanding of the geographical distribution and condition of acroporids. For some areas (Bonaire, Curaçao, Cayman, Mária la Gorda, Costa Rica) shallow reef crests are of limited spatial extent and thus were not sampled. Elsewhere (Yucatan, Belize, San Salvador, Turks and Caicos, and the Virgin Islands), the extensive reef crests were not assessed because of inaccessibility or high wave energy. Greater sampling of shallow reef-crest habitats within the wider Caribbean should be a priority since so little region-wide information is currently available. Nevertheless, the historic range does not appear to be reduced or lost in either acroporid species although occurrence and abundance data suggest range reductions have occurred at the local scale. The presence of acroporid recruits was higher than expected in some areas (e.g., Andros, Cayman) and the existence of a few localized healthy populations of *Acropora palmata* is encouraging. However, very few *A. cervicornis* recruits were observed (Virgin Islands, Nemeth, this volume; Netherlands Antilles, Klomp et al., this volume) and there were few signs of significant reestablishment of/recolonization by this species.

Four species within the *Montastraea* family (*Montastraea annularis*, *M. faveolata*, *M. franksi* and *M. cavernosa*) numerically accounted for about half (~50%) of the frame-building corals on the deep reefs assessed in this volume except for Abrolhos where the endemic *Mussismilia braziliensis* predominated (Kikuchi et al., this volume). The density (as numbers/10 m) and sizes of *Montastraea* colonies can be used to indicate the stability of environmental conditions (e.g. Done, 1995) as well as infer characteristics

of mortality, regeneration, and recruitment (e.g., Bak and Meesters, 1998). High densities of large (>~1 m) colonies are indicative of environmentally stable “old growth” conditions (e.g., the Flower Gardens, Bonaire, Los Roques, and Curaçao). In contrast, the lowest densities are found in areas that are marginal for reef growth (e.g., Abaco, Costa Rica). Smaller colonies (<30 cm) are less likely to have partial mortality but more likely to experience complete mortality, and thus may be a more sensitive indicator of environmental growth conditions. In current versions of the AGRRA protocol all corals equal to, or greater than, 10 cm maximum diameter are now sampled.

High recent mortality coupled with the disease outbreaks described above were observed in the *Montastraea annularis* species complex at several areas during 1999-2000. Because of their slow rates of tissue growth (Hubbard and Scaturo, 1985), colonies with large injuries from fast-spreading diseases (e.g., white plague) have a high likelihood of impaired skeletal growth (Hughes and Jackson, 1985) or diminished reproductive output (Szmant and Gasman, 1990) and some may not recover at all. Given the importance of the *M. annularis* complex as the current and historic principal framebuilder of intermediate-depth reefs, these large, and perhaps unprecedented, disease and bleaching impacts are cause for great concern. Improving our understanding of its dynamics (reproduction, settlement, mortality) and exploring any cases (genotypic, geographic, habitat, depth-dependent) of resistance and/or resilience to these impacts as may occur naturally should help better direct management strategies towards their persistence and recovery.

## Fishes

*Community structure.* Fish community structure at any given locality can be attributed to a wide range of factors including environmental conditions, habitat complexity, and fishing and management regulations. Other factors such as predator-prey interactions (Hixon, 1991), larval supply and recruitment (Cowen, 2000), history of disturbance (Syms and Jones, 2000), quality and intactness of adjacent habitats (e.g., Munday, 2002), and natural biogeographic variation within species may also contribute to the overall variance in fish community structure. Most of the AGRRA species are found throughout the western Atlantic with the exception of the Abrolhos. Some species are known to be much more common in certain areas than others. For example, Nassau grouper are much more prevalent in the Bahamas than in other parts of the Caribbean. However, when combined into a carnivore index, these species differences become less distinct because there may well be other carnivore species present to fill the niche occupied by one that is not present.

Herbivore species composition and sizes were remarkably consistent across surveys. That herbivore densities were more variable, with consistently higher numbers in the eastern and southern Caribbean and lower numbers in the Bahamas, Gulf of Mexico, Mária la Gorda (ID#11) and western Caribbean (Fig. 13B), may be partially related to geomorphic factors discussed above.

Several significant relationships (positive and negative) between habitat variables and the abundance of individual fish species were observed and need to be explored in more detail before conclusions can be drawn regarding their significance. However, that

coral size, density, and partial mortality were not directly related to overall fish density (or biomass) nor with herbivore or carnivore indicators at the scale of the entire region suggests that some tropical fish-habitat relationships identified at smaller scales (e.g., Luckhurst and Luckhurst, 1978; Lindquist and Gilligan, 1986; Nemeth et al., this volume) do not hold at larger spatial scales. One possible reason for the lack of a relationship at the regional scale is that guilds of fishes are utilizing the best available habitat within each of the assessment areas even though the habitats may be quite different from one area to another. Habitat quality in itself may not be the limiting factor determining fish abundance, but other factors such as larval supply, recruitment, and trophic predator-prey relationships may be more important. Clearly definitions of habitat quality vary and may include other measures not included in this analysis such as percentage of *A. palmata* (shallow), percentage of *M. faveolata* (deep), or independent measures of rugosity. Since 2000, the newer version of the AGRRA protocols incorporates an independent measure of substrate rugosity which, in the future, will help to classify reefs based on relief and to better test relationships between structural complexity and other benthic, algal, and fish variables.

*Overfishing.* Differences in fish community structure that can be either directly or indirectly linked to human fishing have been documented in numerous studies (e.g. Roberts, 1991, 1995). While areas that now receive higher protection from fishing [Flower Gardens (ID#5), Los Roques (ID#16), Cayman (ID#12), and Bonaire (ID#17)] displayed slightly higher total biomass than areas with little protection (e.g., the Bahamas and western Caribbean), the difference was not significant. Since in some areas sites were surveyed within and outside of protected areas (e.g., Abaco (ID#1), Cayman (ID#12), Virgin Islands (ID#13), a more rigorous analysis would require assigning levels of protection at the site scale. However, assigning AGRRA sites with an unbiased estimate of protection from fisher-related mortality across the entire Caribbean is difficult because of the wide range of management and harvesting practices in place. An unbiased measure of total fish extraction, broken down by species and area (preferably at a fine spatial scale), is needed before fish density and biomass patterns related to fishing can be examined robustly at large spatial scales.

Results of the 1-way ANOVA analysis using Bryant et al.'s (1998) "overexploitation threat index" show no clear pattern for total biomass or herbivore biomass or carnivore biomass and can be interpreted to suggest several possibilities. For example, the modeling of overexploitation threat may not accurately portray fishing pressures on reefs. In fact, this is likely since the threat index is based on a global-level analysis using a reef's proximity to coastal settlements which vary greatly in size and in the proportion of people that harvest reef fishes. Not factored in the index as other factors that can influence fish populations are habitat quality, adjacent habitat availability (shelf area), and levels of management. It is also likely that AGRRA *total* fish density (and biomass) data are not very sensitive to the type of fishing pressure common at many of the areas assessed in this volume because they are heavily weighted by herbivorous fishes which, except for some large parrotfish such as *Scarus guacamaia*, are seldom heavily targeted. One of the most widely discussed examples in which nearly all reef fishes are harvested is Jamaica (e.g., Koslow et al., 1994). Indeed, during AGRRA assessments

conducted along the north coast of Jamaica in August 2000, total biomass ( $<1,500 \text{ g}/100 \text{ m}^2$ ), herbivore biomass ( $<1000 \text{ g}/100 \text{ m}^2$ ) and carnivore biomass ( $<150 \text{ g}/100 \text{ m}^2$ ) were all significantly lower than had been found at any of the areas in this synthesis (Klomp et al., in press). It would appear that the herbivore (and total fish) density (and biomass) indicators are only strongly affected by intensive overfishing, and that a regional signal associated with more selective targeted fishing may be more difficult to detect.

Carnivore density (as well as size and biomasses) is considered a more sensitive indicator of the type of fishing pressure occurring at many of the areas in this synthesis. As discussed earlier, the high densities (and biomass) reported in the Abrolhos (ID#20) are thought to arise primarily from differences in both habitat type and assessment methodology. For the remaining areas, spatial patterns are apparent and suggest that portions of the western (ID#s7-9) and eastern (ID#s13-15) Caribbean are consistently low in carnivore densities while portions of the southern Caribbean (ID#s16-18) are consistently high. The overall low number of sightings for larger-bodied groupers and snappers ( $\sim 1/100 \text{ m}^2$ ) as a whole suggests the entire region is overfished for many of these more heavily targeted species. Interestingly, the Bahamas subregion in particular Andros (ID#2), has a much higher proportion of large-bodied groupers (*E. striatus*, *M. tigris*, *M. venenosa*) to smaller-bodied species (*E. cruentatus*, *E. guttatus*, *E. fulvus*) than the rest of the region. The Bahamas subregion as a whole may have the least exploited grouper populations in the western Atlantic, an observation also supported by the large number of active grouper spawning aggregations (Sadovy, 1999).

The lack of a significant relationship between carnivore density (or biomass) and overexploitation threat can be explained in part to the way this threat was modeled and in part also to the observation that the assessment areas probably do not represent a full gradient between “pristine” (unfished) and extremely overfished. In addition, the fish AGRRA belt transects are not considered the ideal size for quantifying the larger, solitary carnivores that are targeted by light-to-moderate fishing intensities. The narrow width and small areal coverage per site ( $600 \text{ m}^2$ ) can often result in observers missing cryptic and/or shy serranids and lutjanids that are present on the reef. Statistical power is low because of the overall low number of observations. Fish biomass reports based on Bohnsack-Bannerot cylinders are generally lower than those derived from belt transects, although, to my knowledge, no systematic comparison has been conducted. In addition, the large length-class intervals used by the observers reduce the power of the methodology to detect small differences in fish size. Ideally, belt transect data should be considered in conjunction with the Roving Diver sighting frequency data, as suggested by Schmitt et al. (2002).

Although not examined in this synthesis, many of the individual papers in this volume include sighting frequencies derived from Roving Diver surveys. Based on species accumulation curves for three sites, Nemeth et al. (this volume) determined that a minimum of six Roving Diver surveys (or greater than seven hours search time) would be needed to approximate the actual species diversity in the deep St. Croix reefs. However, much less time would be required to determine the presence and qualitative density of the 75 or so AGRRA species which could also be used to improve our ability to distinguish light-to-moderate levels of fishing pressure.

*Algae and herbivory.* The lack of any significant relationship between algal index and herbivore density/biomass in our data are somewhat in contrast to those of Williams and Polunin (2001) who reported that herbivore biomass showed a significant negative correlation with macroalgal cover and a positive correlation to cropped substrate in five localities across the wider Caribbean. Differences in the assessment techniques for quantifying algal cover are thought to be responsible for these different results. The AGRRA technique scores algal abundance and canopy heights in areas where there is at least 80% algal cover and does not record the absolute abundance of macroalgae for the entire hard substrata of a reef. In fact, the inverse relationship between live stony coral cover and macroalgal canopy height that is evident in the AGRRA data supports the notion that macroalgal canopy height may be a function of the available hard substratum. Many algal species are unpalatable for fishes and canopy heights may only correlate with herbivorous fish density at relatively low abundances where algal choices are limited. However, these results also suggest that there are probably other factors (e.g., environmental conditions limiting algal growth) contributing to each of these signals at the regional scale. Densities of *Diadema antillarum* were low at nearly all surveys except for Costa Rica and thus probably do not exert a strong influence over algal assemblages at this time.

## Reef Health

That three groupings resulted from the similarity analysis (Fig. 15) was somewhat surprising since the indices employed have ecological functions spanning a range from the individual-to-community-to-ecosystem levels (Table 6). In part, the high degree of similarity within the groups (>70%) arises because several indicators reinforced one another (e.g., inverse relationship between coral cover/density and macroalgal index as discussed earlier). Clearly, the choice of indicators had strong bearing on the outcome of this type of similarity analysis. Indicators that displayed the highest large-scale variation (e.g., recent mortality, carnivore density) had more influence on the groupings than those that had minimal large-scale variation (e.g., old partial mortality, colony diameter) (Table 5). None of the areas in the “better” category had experienced severe damage to its stony corals from the 1998 ENSO event although high percentages of mottled and pale colonies were seen in both Bonaire (ID#17) and the windward Netherlands Antilles (ID#14) at the time of their assessment (February, 1999 and December 1999, respectively). In contrast, four of the six assessments in the “worse” category were strongly influenced by this event as was Belize (ID#10) (see Peckol et al., this volume). However, the other two survey areas in the worse category [Costa Rica (ID#19) and Abaco (ID#1)] are subject to high chronic stresses, as reflected mainly in their low coral density, low coral cover, and high macroalgal index. Most of the areas in the “worse” category also had subnormal densities of carnivores while those in the “better” category had above-normal densities.

Patterns observed in the overall state of the region and inferred causes (natural versus anthropogenic) for these patterns have begun to emerge from this preliminary analysis. It was expected that coral reefs remote from human population centers might be more intact and in better condition than those adjacent to population centers where there is a higher potential for exposure to pollution, overfishing, nutrient enrichment,

sedimentation, recreational diving, and anchor damage. Results from this synthesis suggest such may not be the case. Often more remote reefs showed as much, if not more, evidence of reef degradation than reefs closer to human coastal development. For example: Andros (ID#2) had high recent partial mortality of corals, high macroalgal index, and few fish; Anegada and Guana in the British Virgin Islands (ID#13) had high prevalences of coral disease; at Los Roques (ID#16), *A. palmata* in the shallow barrier was nearly 100% standing dead; and in Mouchoir Bank, Turks and Caicos Island (ID#4), macroalgal abundance was high. Fringing reefs on some highly populated islands [i.e., Bonaire (ID#17), windward Netherlands Antilles (ID#14)] seem to have avoided sea surface temperature “hotspots” while remote reefs in the Bahamas [(i.e., Andros, San Salvador (ID#3))] did not.

The observations that remote reefs are in poor shape does not imply that human actions are not influencing reef condition; this has been demonstrated conclusively by numerous studies (e.g., Ginsburg, 1994). Rather, it suggests that the driving forces influencing reef condition across the region are complex and probably involve multiple sources operating over several spatial and temporal scales. Regional (e.g., *Diadema* dieoff, some coral diseases) and subregional (ENSO-driven bleaching and associated diseases) stressors superimposed on localized states (e.g., reef development, degree of overfishing, disturbance history, abiotic conditions), and the degree to which reefs have ecologically adapted to these states over the past several decades, is a more plausible explanation for the current patterns. It also implies that human “proximity” now has a global reach, given atmospheric build-ups and transport of pollutants. The driving forces of western Atlantic reef degradation are operating at large spatial scales and management efforts should be directed towards these same regional scales.

A critical issue that has practical consequences on the outcome of any reef health assessment relates to the benchmark used for the rankings. In this preliminary analysis, departures from ecologic norms were used as the principal criteria for judging health. However, departures from normality are not necessarily unhealthy since natural reef ecosystems are expected to experience routinely disturbance events such as hurricanes (Rogers, 1993). A more meaningful measure of reef health may thus be resilience (Holling, 1973) but there is significant debate with respect to the time scales necessary to gauge reef resilience (Done et al., 1996). Evaluating reef condition based principally on ecologic criteria without regard to other criteria (e.g., socioeconomic, abiotic parameters) leads to its own bias. For example, a heavily managed reef can be ecologically subnormal but may be judged healthy if it fulfils a designated purpose. Thus, while the intent here was to examine patterns of normality and classify survey areas based on their broad groupings, applying judgmental health labels to these groupings is probably premature.

One criticism to the AGRRA approach is that indicators and values are largely based on what we know about the functioning of coral reefs today rather than the type of pristine natural reefs that may have existed in the past. What represents a “normal” or even “healthy” reef today may well be “degraded” with respect to an earlier baseline (Jackson, 1997). Recognizing that ecological baselines have shifted (Jackson, 1997, 2001; Greenstein et al., 1998), the value of making multiple observations across multiple spatial scales that can approximate the average state for the region today is still very high. The initial AGRRA norms synthesized in this volume can now be used as yardsticks to

evaluate reef condition like those for human health (e.g., blood pressure, infant mortality, prevalence of heart disease, life expectancy). In addition, these observations can be used with certain limitations within the hypothesis-testing framework (Underwood, 2000). The results presented in this synthesis, and in the papers of this volume, represent a preliminary analysis of a large amount of data. As additional AGRRA assessments are undertaken, the regional norms will shift to become more representative of the region, particularly for shallow reefs. In addition, more powerful analyses will be possible as our classification and stratification of reef types improves and as more species-specific indices are incorporated into the dataset.

### CONCLUSIONS

- A mean live coral cover of 26% in the deep sites suggests significant losses have occurred over the past several decades but that substantial amounts of coral remain.
- Significant bleaching-induced mortality associated with the 1998 ENSO event was most apparent in the western Caribbean and Bahamas subregions.
- Linkages between infectious diseases, bleaching, and recent mortality were evident and are thought to be a result of increased pathogen activity and sensitivity of corals to disease and mortality during periods of elevated sea surface temperatures.
- Coral and fish community species composition across the entire region were most strongly influenced by environmental and biogeographic factors.
- AGRRA data provides further evidence on the regional decline of acroporids, although some moderate occurrences of *A. palmata* were identified and the historic range does not appear to be reduced or lost in either species.
- Large and perhaps unprecedented disease and bleaching impacts on the *M. annularis* species complex were documented by many of the AGRRA surveys. Given the importance of *Montastraea* as the current and historic principal frame builder of reef slopes, this decline is cause for great concern.
- The overall low number of sightings for larger-bodied groupers and snappers ( $< \sim 1/100 \text{ m}^2$ ) as a whole suggests the entire region is overfished for many of these more heavily targeted species.
- Herbivore density (or biomass) and macroalgal index were not related at the scale of the entire region suggesting that there are probably other factors (availability of substrate, abiotic factors) contributing to each of these signals.
- More remote reefs showed as much evidence of reef degradation as reefs more proximal to human coastal development.
- Driving forces influencing present-day reef condition across the region are complex, and probably involve multiple sources operating over several spatial and temporal scales.

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Table 1. Summary site information for the 20 AGRRA assessments in this volume.

ID#	AGRRA Region	Survey		Modeled Threat Index <sup>1</sup>		Benthic			Sites (#)			Reef type				Other <i>Diadema</i> density (#/100 m <sup>2</sup> )				
		Area	Date(s)	Combined	Over-exploitation	Shallow (≤5 m)	Deep (>5 m)	Total	Shallow (≤5 m)	Deep (>5 m)	Total	Sampling unit (m <sup>2</sup> )	Shallow (≤5 m)	Deep (>5 m)	Other					
1	Bahamas	Abaco, Bahamas	Aug., 1999	H	H	2	6	8	2	6	8	30 x 2	1	1	6	0				
2	Bahamas	Andros, Bahamas	Aug., 1998	M	M	15	16	31	15	16	31	30 x 2	15		16	0.6 ± 1.2				
3	Bahamas	San Salvador, Bahamas	June, 1998	M	L	4	7	11	4	7	11	7.5	2	2	3	4	---			
4	Bahamas	Turks & Caicos Islands	Aug., 1999	M	L	2	26	28	2	26	28	30 x 2	1	1	24	2	0			
5	Gulf of Mexico	Flower Garden Banks, USA	Nov., 1999	L	L	0	2	2	0	2	2	30 x 2			2	1.1 ± 0.4				
6	Gulf of Mexico	Veracruz, Mexico	July, 1999	H	H	3	3	6	---	---	---	---		3	3	---				
7	Western Caribbean	Yucatan, Mexico	June - Sept., 1999	M	L	0	24	24	0	24	24	50 x 2			24	0				
8	Western Caribbean	Akumal/Xcalak, Mexico	March, 1999	M	L	2	12	14	0	6	6	30 x 2		2	12	0.2 ± 0.9				
9	Western Caribbean	Lighthouse Atoll, Belize	July - Oct., 1999	M	L	0	11	11	0	11	11	30 x 2			11	0				
10	Western Caribbean	Belize	May, 1999	M	L	3	10	13	3	10	13	7.5		3	5	5	---			
11	Central Caribbean	Maria La Gorda, Cuba	July, 1999	M	M	0	4	4	0	4	4	50 x 2			4	1.8 ± 1.9				
12	Central Caribbean	Cayman Islands	June, 1999, & June, 2000	M	M	2	40	42	2	40	42	30 x 2	1	1	39	1	0.2 ± 1.4			
13	Eastern Caribbean	Virgin Islands	July, 1999 - Aug., 2000	H	H	4	26	30	4	23	27	30 x 2		4	23	2	1	2.6 ± 5		
14	Eastern Caribbean	Windward Netherlands Antilles	Dec., 1999	M	L	0	24	24	0	24	24	30 x 2			21	3	0			
15	Eastern Caribbean	Grenadines, St. Vincent	June, 1999	M	M	2	3	5	2	3	5	30 x 2	2		3		2.4 ± 3			
16	Southern Caribbean	Los Roques, Venezuela	Oct., 1999	L	L	7	6	13	7	6	13	30 x 2	4	1	2	6	5.5 ± 10.8			
17	Southern Caribbean	Bonaire	Feb., 1999	M	M	0	6	6	0	4	4	30 x 2			6		0			
18	Southern Caribbean	Curaçao	Aug., 1998 & Jan., 2000	H	H	0	14	14	0	4	4	30 x 2			14		0			
19	Southern Caribbean	Costa Rica	Oct., 1999	H	M	1	2	3	1	2	3	30 x 2		1	1	1	23 ± 33			
20	Brazilian	Abrolhos, Brazil	Feb.-April, 2000	M	L	6	7	13	6	5	11	30 x 2		6		7	0			
<b>Total</b>						<b>53</b>	<b>249</b>	<b>302</b>	<b>48</b>	<b>223</b>	<b>276</b>		<b>23</b>	<b>4</b>	<b>10</b>	<b>16</b>	<b>221</b>	<b>9</b>	<b>19</b>	<b>2.3 ± 5.6</b>

<sup>1</sup>The threat index (H=high, M=medium, L=low) for each area is based on model outputs from Bryant et al.'s (1998) global *Reefs at Risk* classification.

<sup>2</sup>30 x 2 = 30 m long x 2 m wide belt transects; 50 x 2 = 50 m long x 2 m wide belt transects; 7.5 = 7.5 m radius cylinders

Table 2. Summary of mean sampling effort in deep sites for each assessment.

ID#	Mean depth (m)	Mean sampling effort/site in deep (> 5m) sites (for a maximum of 10 transects)				
		Benthic component (#)			Fish component-belt transects	
		Transects	Corals	Quadrats	#	Area (m <sup>2</sup> )
1	8	10.0	32	42	10.0	600
2	10	9.9	84	43	9.9	596
3	9	9.3	58	42	---	---
4	14	9.6	91	48	9.8	591
5	21	10.0	118	48	10.0	600
6	10.5	10.0	50	41	---	---
7	10.5	9.6	40	48	6.0	600
8	13	9.2	45	43	3.0	180
9	7.5	9.2	83	45	10.0	600
10	10	10.0	100	19	---	---
11	7.5	10.0	84	43	6.0	600
12	10.5	9.6	90	48	9.7	582
13	10.5	9.8	66	47	10.0	600
14	13	9.5	55	40	10.0	600
15	10.5	10.0	96	49	10.0	600
16	10	5.8	56	18	9.8	590
17	10	8.0	95	32	9.8	585
18	13.5	8.4	138	19	3.3	195
19	6.5	10.0	47	48	10.0	600
20	6	9.6	88	48	10.0	600

Table 3. Summary of means and standard deviations for selected benthic AGRRA indicators for the 20 assessments (for a maximum of 10 transects/site).

ID#	Mean depth (m)	Stony coral density			Partial-colony mortality (%)		Stony corals (%)							
		$\geq 25$ cm stony corals (#)	Live stony coral cover (25%)	$\geq 25$ cm corals (#/10 m)	$< 2$ cm corals (#/m <sup>2</sup> )	Recent	Old	Standing dead	Bleached	Disced	Algal overgrowth	Fish bites	Snails/Worms	Damselfish gardens
<b>A. Shallow (<math>\leq 5</math> m) sites</b>														
1	4	109	18.5 ± 10.6	5.8 ± 2.3	4.5 ± 1.3	2.0 ± 8.3	24.1 ± 30.6	11.0	0.9	2.8	---	---	---	---
2	1	1275	37.8 ± 16.7	10.0 ± 3.2	2.5 ± 1.3	4.4 ± 11.7	21.6 ± 25.4	8.3	14.3	5.9	4.2	5.1	8.5	2.2
3	4	165	---	5.5 ± 2.2	---	0.8 ± 3.8	28.8 ± 28.7	15.2	1.8	1.2	0.6	---	---	---
4	3	137	9.5 ± 2.6	7.5 ± 2.0	2.3 ± 2.7	1.3 ± 7.7	42.3 ± 34.6	24.8	0	4.4	---	---	---	---
5	---	---	---	---	---	---	---	---	---	---	---	---	---	---
6	4.5	137	18.7 ± 9.3	6.0 ± 2.8	1.0 ± 0.7	0.4 ± 2.9	6.9 ± 11.8	---	5.8	0	35.8	1.5	---	---
7	---	---	---	---	---	---	---	---	---	---	---	---	---	---
8	2	134	19.2 ± 11.8	7.5 ± 2.4	2.7 ± 3.3	26.9 ± 22.9	37.8 ± 23.3	6.0	10.4	13.4	0	6.7	27.6	0
9	---	---	---	---	---	---	---	---	---	---	---	---	---	---
10	4	292	---	11.1 ± 5.7	---	3.4 ± 10.5	43.1 ± 31.4	17.1	20.2	1	1.4	---	---	---
11	---	---	---	---	---	---	---	---	---	---	---	---	---	---
12	3.5	104	19.3 ± 6.1	5.8 ± 2.4	4.5 ± 3.3	0.5 ± 3.3	16.9 ± 22.7	---	0	4.8	0	0	3.8	1.0
13	4.5	248	14.0 ± 7.2	8.0 ± 3.3	4.2 ± 2.7	3.6 ± 5.3	44.8 ± 33.9	1.6	28.6	2.8	37.1	17.3	25.8	2.0
14	---	---	---	---	---	---	---	---	---	---	---	---	---	---
15	3.5	160	29.5 ± 5.7	8.3 ± 1.4	3.1 ± 2.9	1.9 ± 7.2	22.1 ± 21.5	6.3	2.5	0	0.6	0.6	18.1	1.3
16	2.5	332	14.6 ± 12.4	8.1 ± 4.0	1.3 ± 2.0	5.4 ± 9.8	21.1 ± 25.3	52.1	3.6	0.9	2.1	1.2	21.4	0.6
17	---	---	---	---	---	---	---	---	---	---	---	---	---	---
18	---	---	---	---	---	---	---	---	---	---	---	---	---	---
19	2	44	3.4 ± 1.0	6.1 ± 2.7	0.3	1.3 ± 6.3	28.0 ± 26.1	6.8	4.5	6.8	40.9	11.4	0	4.5
20	4	503	10.6 ± 5.4	12.4 ± 6.0	9.4 ± 2.4	0.1 ± 0.9	18.3 ± 21.9	0	0.4	0	---	---	---	---
<b>Mean</b>	<b>3.2</b>	<b>280</b>	<b>17.7 ± 9.5</b>	<b>7.9 ± 2.2</b>	<b>3.3 ± 2.5</b>	<b>4 ± 7.1</b>	<b>27.4 ± 11.6</b>	<b>14.9</b>	<b>7.2</b>	<b>3.4</b>	<b>12.3</b>	<b>5.5</b>	<b>15.0</b>	<b>1.7</b>
<b>B. Deep (<math>&gt; 5</math> m) sites</b>														
1	8	194	13.5 ± 10.1	3.7 ± 1.7	4.7 ± 1.2	4.4 ± 12.4	26.1 ± 28.7	6.2	14.9	1.0	8.2	---	---	1.0
2	10	1347	25.0 ± 14.3	10.1 ± 4.8	9.7 ± 4.2	16.5 ± 29.9	17.9 ± 24.3	4.1	8.5	17.7	7.9	0.1	0.1	2.9
3	9	406	---	6.9 ± 2.7	---	2.9 ± 8.8	29.5 ± 29.7	11.6	3.2	0.5	1.5	1.7	0.2	1.2
4	14	2362	19.1 ± 8.7	10.2 ± 3.7	5.9 ± 2.7	3.3 ± 11.8	24.2 ± 25.2	1.1	0.3	5.4	---	---	---	---
5	21	235	57.8 ± 13.1	12.4 ± 2.8	1.9 ± 0.8	1.8 ± 6.1	9.5 ± 16.4	1.3	3.4	0	3.4	7.2	---	1.3
6	10.5	151	22.2 ± 13.0	6.9 ± 3.4	3.1 ± 0.1	0.1 ± 1.1	8.2 ± 13.6	---	2.6	0	35.1	---	---	---
7	10.5	951	14.5 ± 6.9	5.3 ± 2.3	3.5 ± 2.7	17.8 ± 19.9	14.5 ± 16.4	5.6	10.4	13.9	---	---	---	1.7
8	13	541	20.9 ± 13.1	6.1 ± 2.4	1.7 ± 1.1	5.0 ± 12.0	26.3 ± 25.9	4.1	11.1	3.7	0.4	3.9	---	9.4
9	7.5	909	21.1 ± 9.9	9.6 ± 2.5	6.1 ± 3.8	2.9 ± 9.0	24.8 ± 24.7	0.4	10.0	3.5	31.7	10.5	8.6	40.9
10	10	1004	---	11 ± 5	---	6.3 ± 18.0	29.9 ± 29.9	11.8	17.6	1.9	1.1	---	---	---
11	7.5	334	20.2 ± 5.1	9.1 ± 2.6	1.7 ± 1.1	5.5 ± 12.8	29.4 ± 28.1	1.8	5.4	4.8	1.2	---	---	47.6
12	10.5	3617	20.5 ± 9.1	10.2 ± 4.2	3.9 ± 2.1	4.2 ± 13.2	24.0 ± 25.1	1.1	1.4	5.3	0.3	0.4	0.2	2.5
13	10.5	1723	22.3 ± 12.1	8.9 ± 4.4	9.1 ± 4.6	1.5 ± 5.6	29.8 ± 29.6	1.0	26.8	6.2	8.6	8.0	1.2	5.2
14	13	1309	21.4 ± 12.7	6.5 ± 3.2	4.9 ± 4.6	1.4 ± 7.7	20.8 ± 24.8	0.8	26.5	1.4	0.2	0.4	0.4	1.3
15	10.5	288	39.1 ± 9.8	10.4 ± 2.9	2.8 ± 2.4	1.6 ± 5.7	26.2 ± 25.6	0.3	11.8	4.2	3.8	1.4	1.0	17.4
16	10	335	41.6 ± 2.2	10.4 ± 3.2	2.5 ± 3.5	4.1 ± 10.1	20.0 ± 18.8	6.6	4.2	3.0	1.2	3.6	2.1	14.3
17	10	570	46.5 ± 13.3	13.3 ± 4.1	4.2 ± 2.7	0.7 ± 2.5	31.3 ± 27.5	2.3	27.2	5.3	1.1	6.5	0.4	13.3
18	13.5	1925	35.2 ± 11.9	17.9 ± 5.3	4.0 ± 4.1	1.1 ± 3.6	20.0 ± 20.4	0.5	0.1	10.4	0.8	1.5	0.7	3.5
19	6.5	93	2.5 ± 1.1	5.4 ± 1.8	3.5 ± 3.1	1.8 ± 7.1	14.3 ± 18.5	---	9.7	9.7	22.6	2.2	---	---
20	6	619	20.5 ± 9.9	11.5 ± 4.6	14.6 ± 3.8	0.4 ± 3.9	11.3 ± 18.6	0	2.9	0	---	---	---	0.3
<b>Mean</b>	<b>10.6</b>	<b>945.7</b>	<b>25.8 ± 13.3</b>	<b>9.3 ± 3.3</b>	<b>4.4 ± 3.4</b>	<b>4.2 ± 4.7</b>	<b>21.9 ± 7.2</b>	<b>3.5</b>	<b>9.9</b>	<b>4.9</b>	<b>7.6</b>	<b>3.6</b>	<b>1.5</b>	<b>1.5</b>

--- = no data

Table 4. Comparison of recent and old partial-colony mortality, incidence of standing dead, bleaching, and disease for the 20 most common stony coral species or taxa (all colonies  $\geq 25$  cm in diameter), for a maximum of 10 transects/site.

Coral species or taxon	Stony corals (#)	Partial-colony mortality (%)		Standing dead	Bleached	Stony corals (%) (not including standing dead)						
		Recent	Old (minus standing dead)			Total	Disease <sup>1</sup>				Other	Unknown
							BBD	WBD	WP	YBD		
<i>Montastraea annularis</i>	4220	6.9	33.3	1.9	15.3	9.7	1.0	6.0	2.3	0.3	0.1	
<i>Montastraea faveolata</i>	2368	5.5	24.2	0.4	12.5	9.4	1.0	6.3	1.2	0.8		
<i>Montastraea cavernosa</i>	1727	3.1	23.5	1.2	10.0	1.7	0.6	0.4	0.1	0.6		
<i>Siderastrea siderea</i>	1465	2.5	22.4	0.4	11.9	4.4	0.8	3.7	1.6	2.7	0.1	
<i>Montastraea franksi</i>	1397	8.4	19.2	0.6	14.0	10.2	3.7			1.2		
<i>Porites astreoides</i>	1211	2.1	11.7	0.5	4.8	2.5				2.5		
<i>Diploria strigosa</i>	1195	4.3	19.6	2.8	11.1	2.8	0.5	1.5	0.3	0.3	0.2	
<i>Agaricia sp.</i>	1170	3.1	22.6	0.8	4.1	3.4	0.1	2.2		0.7	0.3	
<i>Acropora palmata</i>	1465	3.8	24.3	32.2	10.0	9.1	2.4			1.0	5.6	
<i>Mussismilia braziliensis</i>	871	0.1	15.0	0.0	1.0	0.0						
<i>Porites porites</i>	869	3.8	26.8	2.1	2.7	2.0				2.0		
<i>Colpophyllia natans</i>	760	3.2	19.5	1.7	4.6	4.4	1.5	1.9		1.1		
<i>Diploria labyrinthiformis</i>	501	5.8	19.4	1.6	10.3	3.2	0.6	1.4	0.2	1.0		
<i>Agaricia agaracites</i>	376	2.6	18.9	0.3	3.5	3.5	0.3	2.9	0.3			
<i>Millepora sp.</i>	333	0.5	15.4	3.6	0.6	0.6				0.6		
<i>Millepora complanata</i>	267	4.6	20.6	5.6	14.3	0.4	0.4					
<i>Acropora cervicornis</i>	277	7.3	23.5	18.4	6.6	12.8	12.4			0.4		
<i>Agaricia tenuifolia</i>	308	16.5	38.0	29.2	4.1	0.9					0.9	
<i>Stephanocoenia intersepta</i>	207	1.6	29.4	0.5	5.3	21.4	1.5		0.5	19.4		
<i>Meandrina meandrites</i>	155	4.5	15.0	0.0	1.9	5.8	1.3	2.6				
<b>Total (N); Mean (others)</b>	<b>21,142</b>	<b>4.5</b>	<b>22.1</b>	<b>5.2</b>	<b>7.4</b>	<b>5.4</b>						

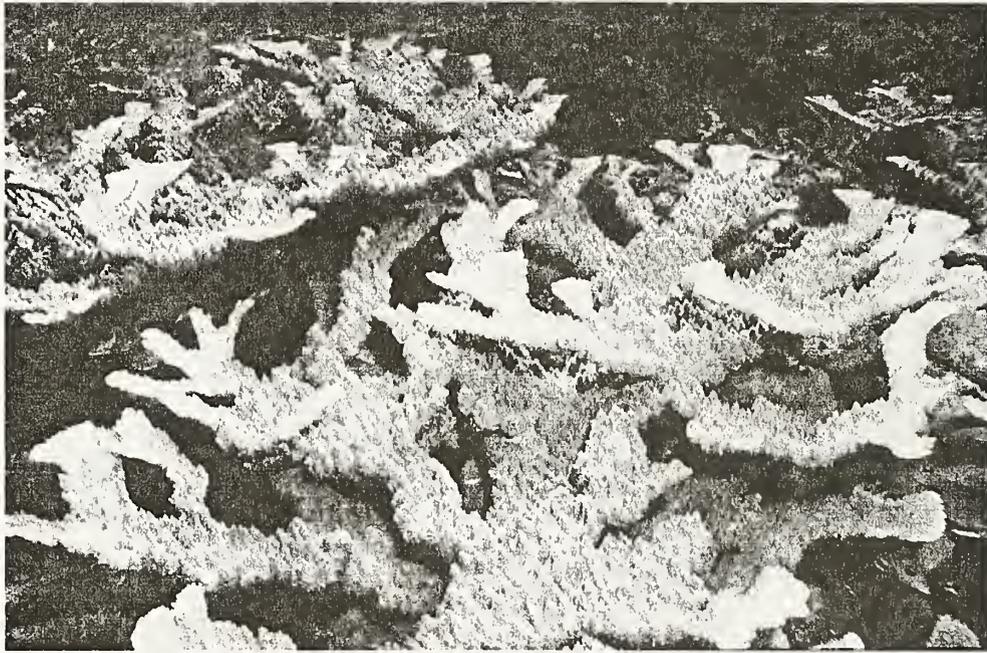
<sup>1</sup>BBD = black-band disease; WBD = white-band disease; WP = white plague; YBD = yellow-blotch (=yellow-band) disease; Other = dark spots, white pox (=patchy necrosis), etc.

Table 5. Comparison of the coefficient of variation (standard deviation divided by the mean) for 12 AGRRA indicators in the deep sites (>5 m) at four spatial scales.

AGRRA Indicator	Coefficient of Variation (%)			
	within Transects (~0-0.01 km)	within Site/ between Transects (~0.01-0.1 km)	within Assessment/ between Sites (~1-100 km)	between Assessments (~100-1000 km)
Live coral cover (%)	---	37	34	49
Density of ≥25 cm stony corals (#)	---	37	23	38
Maximum diameter of the <i>M. annularis</i> complex (cm)	44	36	33	28
Recent partial-colony mortality (%)	122	138	78	116
Old partial-colony mortality (- standing dead) (%)	100	50	26	30
Total mortality (including standing dead) (%)	93	44	28	33
Relative abundance macroalgae (%)	76	56	56	58
Macro canopy height (cm)	41	38	36	87
Relative abundance crustose coralline algae (%)	64	46	33	57
Total AGRRA fish density (#/100 m <sup>2</sup> )	---	46	40	54
AGRRA herbivore density (#/100 m <sup>2</sup> )	---	49	36	40
AGRRA carnivore density (#/100 m <sup>2</sup> )	---	104	57	106

Table 6. Summary of 15 AGRRA indicators that were used to develop a preliminary biotic health index showing major characteristics of each for the surveys synthesized in this volume.

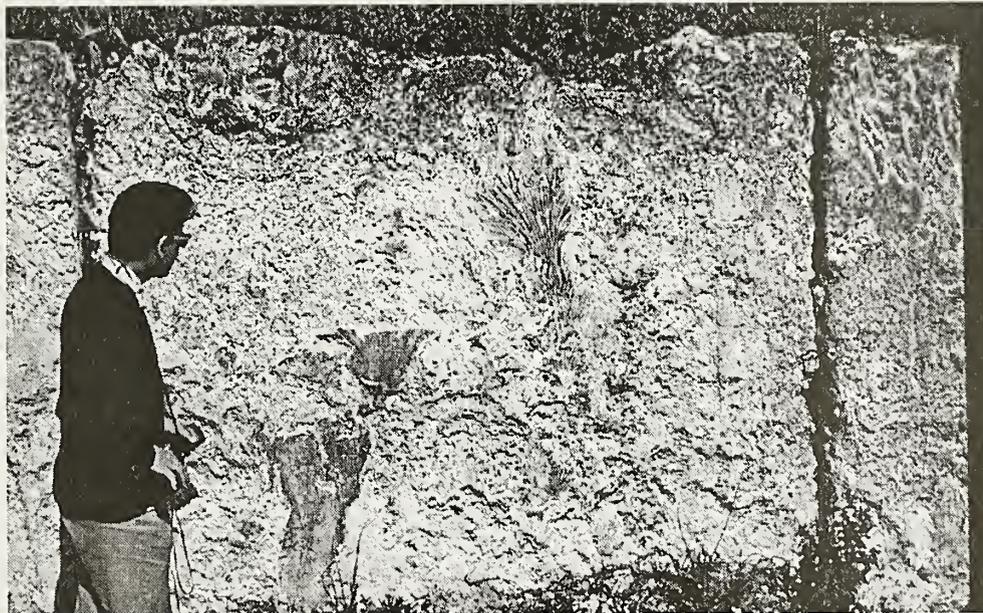
AGRRA indicator	Ecological relevance	Spatial scale variation	Approximate temporal signature	Stress type Indicator:	Qualitative relationship	Regional		This volume	
						Average (Norm)	Endpoints	Lowest	Highest
Live stony coral cover (%)	Community	L	S-L	Either	Low=unfavorable	26	3 - 56	Costa Rica	
Large coral density (# ≥25 cm)/10 m	Community	S, L	I to L (include standing dead)	Chronic	High = favorable	9	4-18	Flower Gardens	
Maximum diameter of <i>M. annularis</i> complex (cm)	Individual Community	S	I to L	Chronic	Low = unfavorable	71	49 - 115	Abaco	
Recent partial-colony mortality (%)	Individual Ecosystem	S	S	Acute	High = favorable	4	1 - 18	Curacao	
Old partial-colony mortality (-standing dead) (%)	Individual Ecosystem	S	I	Chronic	High = unfavorable	22	8 - 31	Veracruz	
Standing dead corals (%)	Individual Ecosystem	S	S-L	Either	Low = favorable	4	0 - 12	St. Vincent	
Diseased corals (%)	Individual Ecosystem	S	S	Acute	High = unfavorable	5	0 - 18	Belize	
Bleached corals (%)	Individual Ecosystem	S	S	Acute	Low = favorable	10	0 - 27	Flower Gardens	
Small corals (# ≤2 cm/m <sup>2</sup> )	Individual Ecosystem	S	S	Acute	High = unfavorable	4	2 - 15	Andros	
Macroalgal index	Community Ecosystem	S, I	S, I	Either	Low = unfavorable	4	2 - 15	Curacao	
Relative abundance of crustose coralline algae (%)	Community Ecosystem	L	S, I	Chronic	High = favorable	82	12 - 215	Bonaire	
<i>Diadema</i> density (#/100 m <sup>2</sup> )	Community Ecosystem	L	S, I	Chronic	Low = favorable	29	11 - 42	Yucatan	
Total AGRRA fish density (#/100 m <sup>2</sup> )	Community Ecosystem	L	S, I	Chronic	Low = unfavorable	2	0 - 23	SEVERAL	
AGRRA herbivore density (#/100 m <sup>2</sup> )	Community Ecosystem	S	I	Chronic	High = favorable	49	21 - 123	Costa Rica	
AGRRA carnivore density (#/100 m <sup>2</sup> )	Community Ecosystem	S, L	S	Either	Low = unfavorable	31	15 - 54	Andros	
AGRRA density (#/100 m <sup>2</sup> )	Community Ecosystem	S, L	S	Either	High = favorable	6	0.4 - 26	Abaco	
								St. Vincent	
								Abrolhos	



**Plate 2A.** Historically, many of the shallow crests of ocean-facing Caribbean reefs looked much like this luxuriant thicket of *Acropora palmata*. Beginning in the early 1970's, colonies of *A. palmata* and of *A. cervicornis* began to die from the effects of what has become known as white-band disease. (Photo Robert N. Ginsburg)



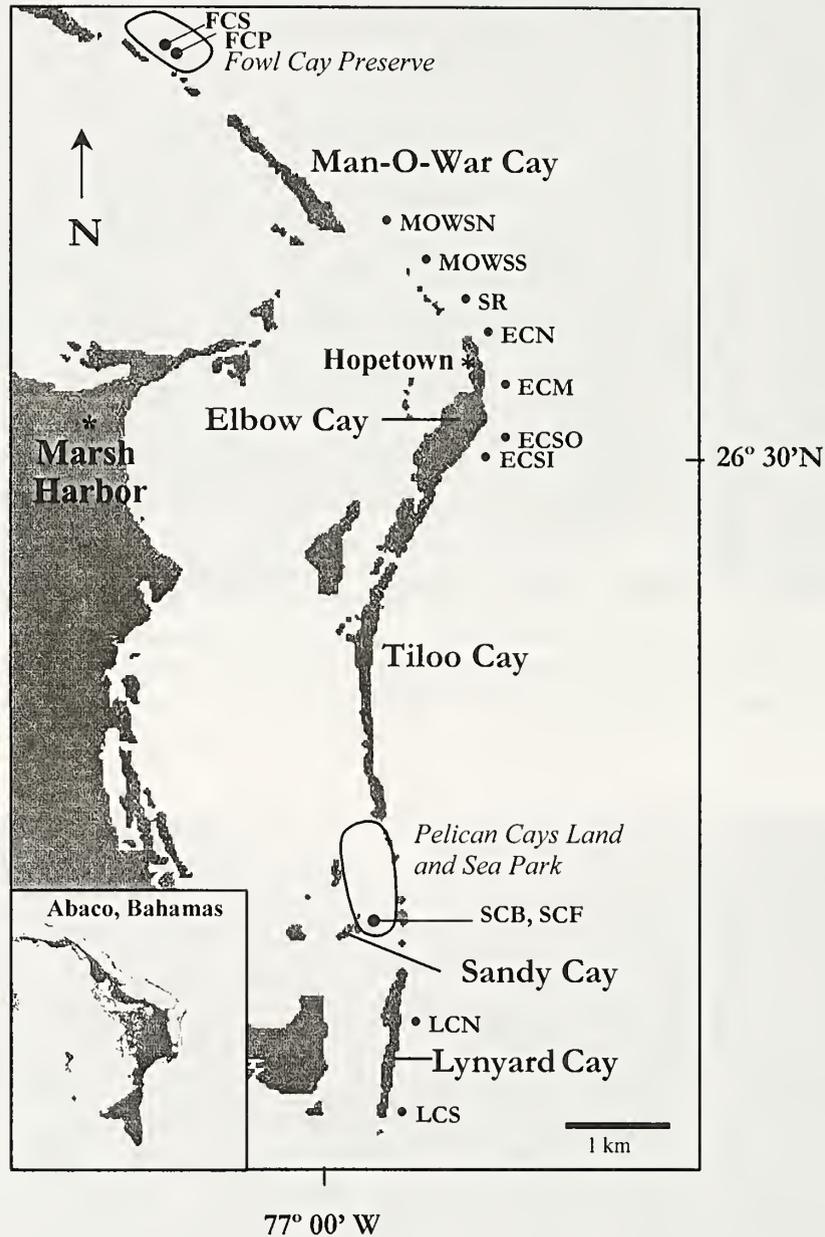
**Plate 2B.** The decline of acroporids has been so pervasive in the last three decades that numerous reef crests are now little more than vast cemeteries of dead coral. Many large dead acroporids are still "standing" in growth position (as shown here). AGRRA surveys have revealed a number of healthy reefs off Andros (this volume) and Cuba (subsequent assessments) where live acroporids are locally abundant. (Photo Robert S. Steneck)



**Plate 3A.** How much coral is in a coral reef limestone? The wall above in an abandoned quarry, now a State Park on Windley Key, Florida, shows a section of a Late Pleistocene reef limestone, *ca.* 125,000 years old. Measurements of the total area of coral that is visible to the naked eye within a five block area gave an average of 39% . Only the larger massive coral skeletons are in growth position, and most coral is fragmented. Including the smaller fragments and sand-sized grains of coral would increase the total to nearly 50% for this patch reef community. (Photo Robert N. Ginsburg)



**Plate 3B.** AGRRA indicators can be used to provide norms of reef condition, like the value of calibrating the geological record of a coral community exemplified 3A. As additional AGRRA assessments are undertaken, the regional norms will become more representative of the region thus moving towards the development of a regional baseline of *current* coral reef health. The combination of both past and current temporal perspectives is key in understanding how to preserve coral reefs for future generations. (Photo Bernhard Riegl)



**Figure 1.** AGRRA survey sites offshore Abaco, Bahamas, with outlines for the Fowl Cay Preserve and Pelican Cays Land and Sea Park.

Abbreviations: LCS = Lynyard Cay south, LCN = Lynyard Cay north, SCB = Sandy Cay backreef, SCF = Sandy Cay forereef, ECSI = Elbow Cay south inner, ECSO = Elbow Cay south outer, ECM = Elbow Cay middle, ECN = Elbow Cay north, SR = Storr's Reef, MOWSS = Man O' War Cay south of south channel, MOWSN = Man O' War Cay north of south channel, FCP = Fowl Cay pinnacles, and FCS = Fowl Cay shallow.

# A RAPID ASSESSMENT OF CORAL REEFS NEAR HOPETOWN, ABACO ISLANDS, BAHAMAS (STONY CORALS AND ALGAE)

BY

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## ABSTRACT

Coral reefs at 13 sites ranging in depth from 1-16 m near Hopetown, Abaco Islands, Bahamas were surveyed utilizing the Atlantic and Gulf Rapid Reef Assessment (AGRRA) benthos protocol. A total of 35 species of scleractinian corals and 2 species of calcareous hydrocorals were observed. The overall coral cover averaged just over 14%. Among corals that were at least 10 cm in diameter, small colonies (<40 cm diameter) predominated in all sites except for the Fowl Cay pinnacles where 68% were larger than 60 cm in diameter. Large colonies (>40 cm diameter) were also found in the Lynyard Cay spur-and-groove formations and the Sandy Cay fore reef. Zero-4% of the colonies were affected by disease. Total (recent + old) partial-colony mortality ranged from 9-31% (both extreme values being found in outer reef crests). Turf algae were the most common algal functional group overall. Macroalgae were ubiquitous, however, with relative abundance values of about 25-47%. Macroalgal indices (a proxy for biomass) ranged from 64 in the Sandy Cay back reef to 184 in the Fowl Cay outer reef crest.

## INTRODUCTION

The AGRRA protocol is being applied throughout the Bahamas and Caribbean to document the condition of reefs with a technique that allows interregional comparisons. This study targeted the reefs of the central Abaco Islands because of their location near the northeasternmost extension of the Bahamas platform. Reefs at the extremes of their geographic range occur near important physiological thresholds for stony corals and may respond earlier to global climate change or anthropogenic impacts than more centrally located reefs. With the exception of Bermuda, Abaco's reefs are nearest the northern limit of shallow reef formation in the wider Caribbean. Additionally, the reefs of central Abaco are near an area of relatively high population density and user pressure from local and tourist fishers and divers.

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The Abaco Islands archipelago, which is composed of middle-to-late Pleistocene eolianites, skirts the eastern margin of Little Bahama Bank (Locker, 1980). Great Abaco itself is of middle Pleistocene age and acts as a barrier to the eastward transport of fine sediments from the Bank interior. A shallow lagoon, the Sea of Abaco, is bounded by Great Abaco to the west and southwest. A string of windward cays to its east and northeast act as a barrier to waves. However, deep tidal channels can act as “energy windows” allowing energy flux (from open-ocean waves and tidal currents) into the lagoon. For example, at Sandy Cay which lies within the lagoon, a well-developed fringing reef thrives on a topographic high where it is exposed to open-ocean waves provided by the wide passes between the string of Lynyard Cay, the Pelican Cays (three islands), and Tiloo Cay. The reef community on the seaward side of Lynyard Cay also lies atop a topographic high; however, the steep slope of the narrow shelf may prevent the reef structure from being well developed (Kenneth Banks, personal observations).

A relatively continuous bank-barrier reef is known to extend just offshore of the eastern cays of Abaco from the northern tip of the Little Bahama Bank, west of Walker’s Cay, southward to the middle of Elbow Cay near Hopetown, a distance of approximately 160 km. Southward of this point the shelf becomes narrow with a steep slope limiting the extent of suitable reef development area. However, spur-and-groove formations have developed at southern reefs offshore of Lynyard Cay on the topographic high spot. Storr (1964) previously described several lines of unusual parallel narrow linear reefs offshore of Johnny’s Cay, several kilometers north of Hopetown on Elbow Cay. Otherwise the Abacos’ reef flora and fauna have not been studied.

Natural impacts on the reefs of Abaco are poorly documented. Frequent hurricane activity (e.g., Hurricanes Erin, 1995, and Floyd, 1999) undoubtedly affects their community dynamics. Moreover, the extent of anthropogenic impacts is not currently known although Woodley et al. (2000) mentioned that overfishing, discharge of human wastewater, and loss of coastline habitat are contributing to degradation of marine ecosystems in the Bahamas. Two of the study areas, Sandy Cay and Fowl Cay, are within marine protected areas (Pelican Cays Land and Sea Park and Fowl Cay Bahamas National Trust Preserve, respectively) where harvesting of marine organisms is not permitted (Dodge, 1999; Woodley et al., 2000). Sandy Cay (in the park) and Lynyard Cay (south of the park and distant from population centers) appear to be relatively undisturbed by direct anthropogenic influences yet development is increasing in the Abacos archipelago. The parks currently do not have staff, and some poaching undoubtedly occurs, but many of the local dive charter operators attempt to aid in enforcement of the park rules. Additionally, all interviewed local inhabitants of Hopetown, including charter boat captains, dive operators, restaurateurs and resort operators, concur that these are “no take” areas. Charter fishing activities are generally targeted on pelagic fish species, or take place in the backwaters and tidal creeks of the Sea of Abaco, and probably don’t impact the reefs significantly. However, preliminary observations in November 1997 at locations of Storr’s (1964) study suggested that fish populations have been reduced (Kramer et al., personal communication). The northerly reefs (Man O’ War, Fowl, and Elbow Cay) surveyed in this investigation are in close proximity to the more developed areas around Marsh Harbour, Hopetown and Man O’ War Cay, locally known as the Hub of Abaco. The residents of this area depend heavily on commercial-scale harvesting of finfish (hook and line and spearguns), spiny lobster

(spearguns) and conch (collected by divers). Catches are exported to the United States or sold locally to supply the tourist industry. Hence, this project provides important data for more effective management of Great Abaco's southern reefs and a baseline for future comparisons.

## METHODS

From August 9-16, 1999 a team of seven divers trained in the AGRRA protocols surveyed reefs near Hopetown on Elbow Cay (Fig. 1). Aerial photographs and discussions with local residents, followed by ground-truthing, allowed the selection of representative reefs along a north-south geographical gradient within small-boat operating distance of Hopetown. The four sites at Sandy and Fowl Cays were strategically located in marine preserves. The Storr's Reef site was located offshore of the linear reef forms studied by Storr (1964).

At each site we focused on areas with the greatest reef development and highest cover of live scleractinian corals. Locations without good reef development (such as the southern end of Elbow Cay and Tiloo Cay) were not visited. Surveyed habitats included a well-developed fringing reef at Sandy Cay (the enormous thicket of dead *Acropora palmata* on its reef crest was not surveyed). The Sandy Cay back reef lacks extensive development of *Acropora cervicornis* but several patches were present in the surveys. Its fore reef, with massive corals on a steep slope extending from near sea level to a sandy plain at 10m depth, resembles comparable habitats in the bank-barrier reef. Surveys on the bank-barrier reef were conducted at: low-relief outer reef crests (e.g., Storr's Reef and Elbow Cay north); low-relief spur-and-groove formations (Lynyard Cay); a shallow pinnacle reef (Elbow Cay south, outer site); and structurally complex pinnacles that rise from depths of 12-15 m to near the surface (Fowl Cay, where transects circumnavigated the deep pinnacle bases).

The AGRRA benthos protocol Version 2 (see Appendix One, this volume) was followed with the following modifications. Stony corals with maximum widths of 10 cm or larger were included because Abaco's live corals are typically of smaller sizes than those in other Caribbean locations and coral cover was low (personal observations). Coral diameters were measured to the nearest 5 cm. Sediments in the algal quadrats were fanned with quick hand motions prior to estimating the abundance of crustose coralline algae. Coral identifications were aided by reference to Humann's (1996) field guide. Latitude and longitude were determined using a global positioning system (GPS) receiver, Magellan GPS 315. Readings were confirmed or updated as necessary in August 2000 after selective availability (SA) was turned off. Quality assurance surveys of reference transects were initially performed at two sites (Hopetown Reef and Sandy Cay) by all team members who compared results to determine any differences regarding taxonomy, measurements, mortality, etc., and reach a consensus.

## RESULTS

Thirteen sites ranging in depth from 1-16 m were surveyed along the offshore shelf between Lynyard Cay in the south and Fowl Cay in the north (Fig. 1; Table 1). We found the reef morphology at most offshore sites to be typical of bank-barrier reefs (Goreau, 1959; Glynn, 1973). For example, the low-relief spur-and-groove system off Lynyard Cay runs normal to the prevailing wave assault at depths of 6-11 m. Most of these reefs were composed primarily of massive framework builders such as *Montastraea* spp. and *Diploria* spp. The branching elkhorn coral, *Acropora palmata*, was not evident as a primary framework builder except in the Sandy Cay fore reef. However, the deep pinnacle reefs at Fowl Cay have numerous *Agaricia agaricites* and large (up to 400 cm diameter) colonies of *Montastraea faveolata*. Massive corals and *Acropora cervicornis* are found in the back reef of Sandy Cay.

### Stony Corals

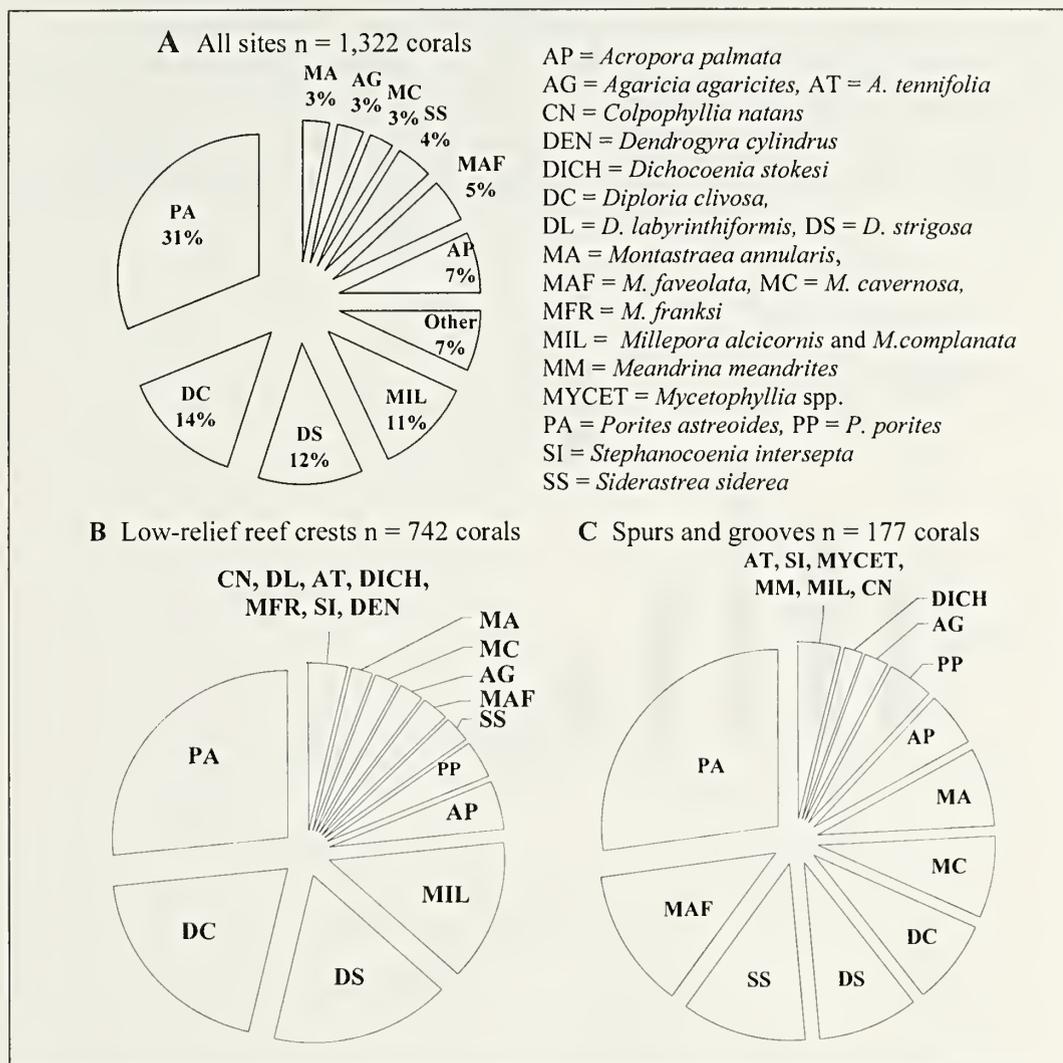
*Density and percent cover.* A total of 1,443 stony corals that were  $\geq 10$  cm in diameter were counted in 186 transects. Coral density averaged 7 colonies/10 m transect (0.7 colonies/m) when all sites were combined (Table 1). Density values ranged from 5.0-8.5 colonies/10 m transect ( $\sim 0.50$ -0.85 colonies/m). Two of the three sites with the highest density were in the Sandy Cay protected area.

Live stony coral cover, which averaged over all sites was just over 14% (mean=14.3, sd=7.6, n=13 sites), varied from about 8.5% in the Elbow Cay north outer reef crest to about 22.5% in the Sandy Cay fore reef (Table 1). Two of the three sites with the highest coral cover (Sandy Cay fore reef and Fowl Cay pinnacles) were in marine protected areas.

The pooled average coral density in the four protected sites (Sandy and Fowl Cays) was slightly higher than the pooled density for the nine sites that are not protected (0.78 colonies/m versus 0.68 colonies/m). Similarly, the percent live coral cover averaged for the four protected sites (mean=16.4, sd=4.4) was somewhat greater than that for the nine that are not protected (mean=13.3, sd=2.8). Neither of these differences between protected and unprotected sites was significant (Rank Sum Test, Ambrose and Ambrose, 1995).

*Species.* A total of 36 taxa of scleractinian corals and two species of calcareous hydrocorals were observed including species seen off the surveyed transects (Table 2). *Porites astreoides* was the most abundant species of the  $\geq 10$  cm corals in all sites (Fig. 2A) with the exception of the deep Fowl Cay pinnacles where mounding *Montastraea faveolata* and foliose *Agaricia agaricites* predominated. *Diploria clivosa* and *D. strigosa* were also numerically abundant in the shallow (<6 m) outer reef-crest habitats (Fig. 2B) whereas the spur-and-groove formations at Lynyard Cay had high abundances of *Montastraea faveolata* and *Siderastrea siderea* (Fig. 2C). *Millepora* spp., *Acropora palmata* and *Montastraea faveolata* were regularly observed in most sites.

*Size.* In most (11/13) sites, corals that were smaller than 40cm in diameter (largely comprised of *Porites astreoides* and *Diploria* spp.) made up the largest portion of the  $\geq 10$  cm coral community (Fig. 3A,B). Large corals were infrequently encountered with size classes of over 200 cm comprising 0-10% of the pooled surveyed colonies. At the Fowl

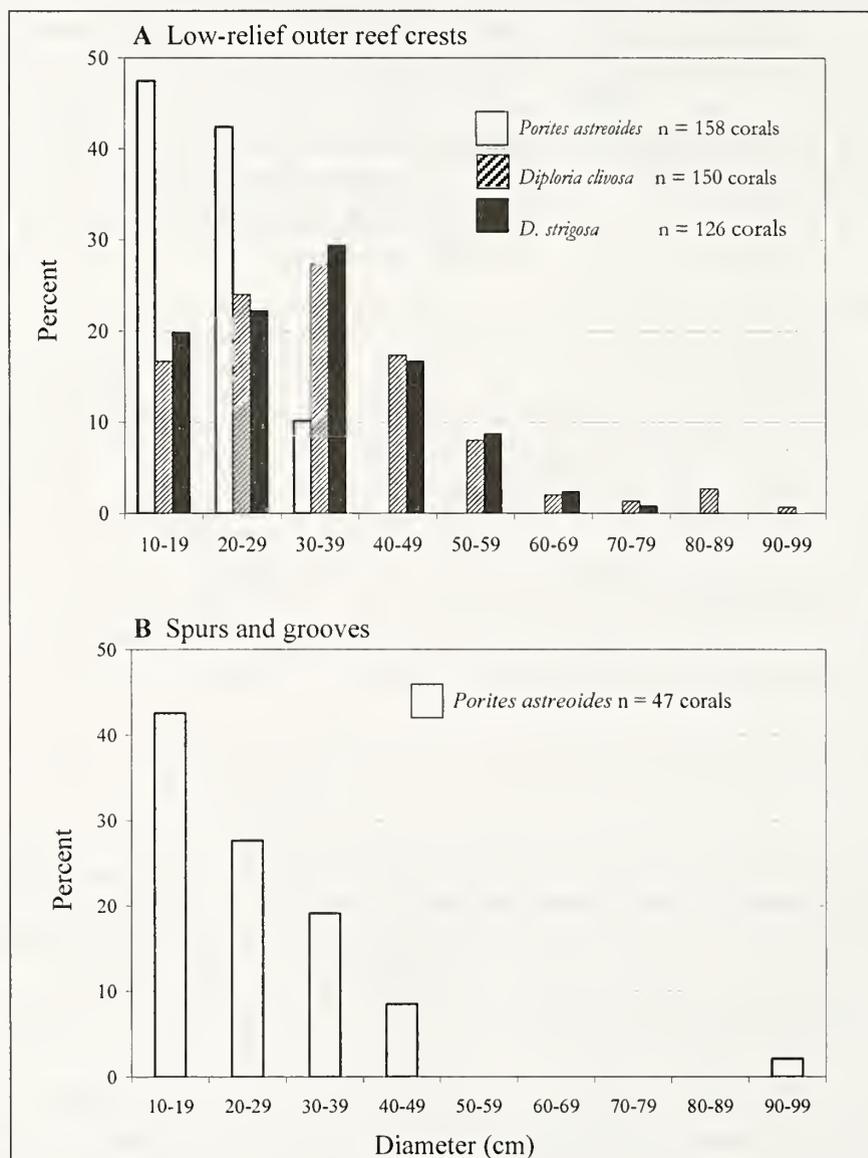


**Figure 2.** Species composition and mean relative abundance of all stony corals ( $\geq 10$  cm diameter) in (A) All Abaco, Bahamas sites, (B) Low-relief outer reef crests (7 sites), and (C) Spurs and grooves (2 sites). Coral species with  $< 1\%$  are placed in a grouped category; Other in (A) = *Agaricia tenuifolia*, *Colpophyllia natans*, *Dendrogyra cylindrus*, *Dichocoenia stokesii*, *Diploria labyrinthiformis*, *Madracis mirabilis*, *Manicina areolata*, *Meandrina meandrites*, *Montastraea franksi*, *Mycetophyllia* spp., *Porites porites*, and *Stephanocoenia intersepta*.

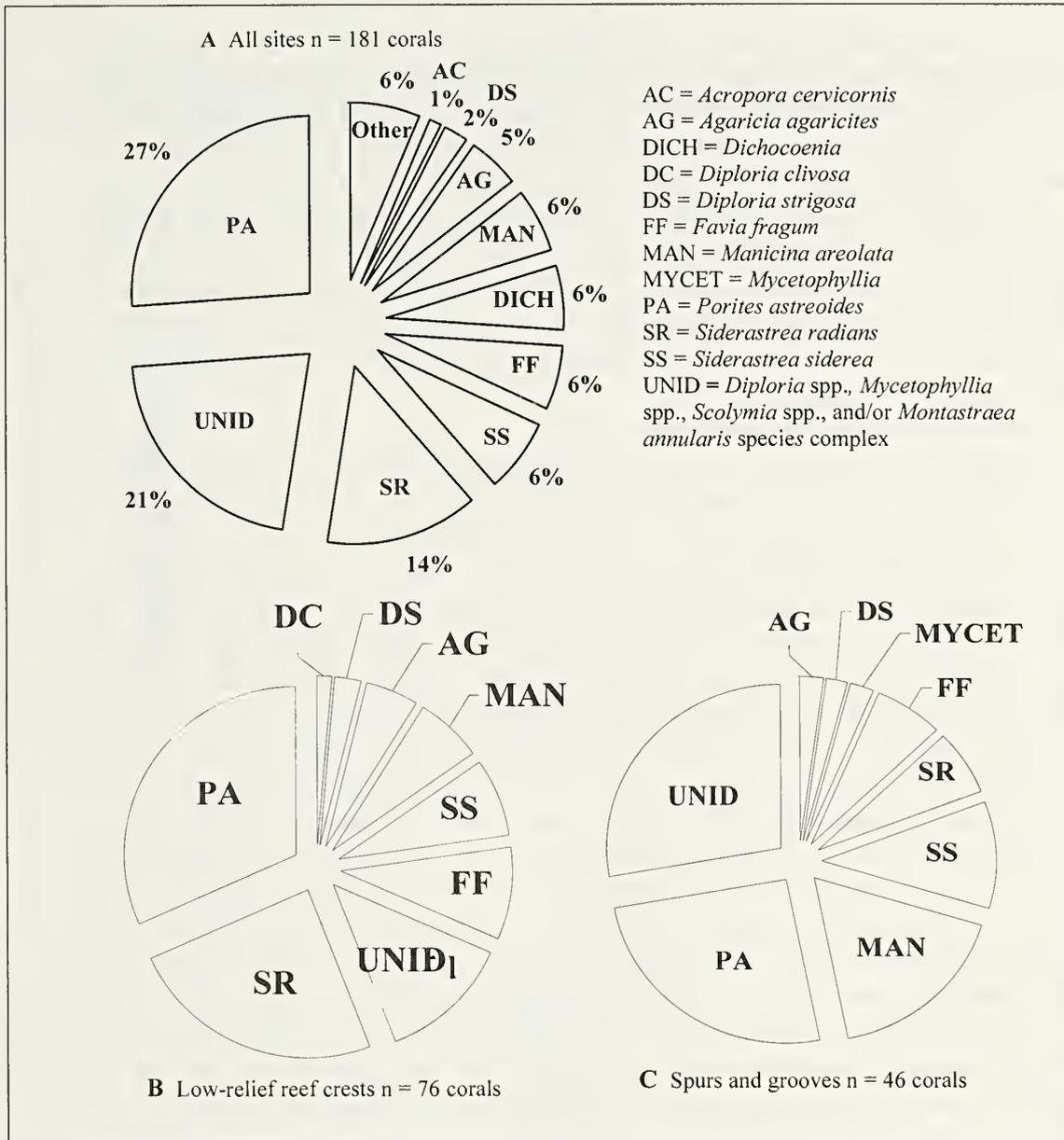
Cay pinnacles, however, smaller ( $< 40$  cm) corals were less than 20% of the total and nearly 20% of all colonies were larger than 150 cm in diameter. Mean colony diameters (Table 3) overall ranged between 32 cm (Man O'War Cay north of the south channel) and 62 cm (Sandy Cay fore reef). The largest size classes ( $> 100$  cm) were mainly comprised of *Montastraea faveolata* (Fowl Cay pinnacles) and *Acropora palmata*; small corals (10-20 cm) were mostly represented by *Porites astreoides*, *Diploria strigosa*, *Diploria clivosa* and *Agaricia agaricites*.

**Recruits.** *Porites astreoides* was the most common coral recruit. Together with *Siderastrea radians* and *S. siderea*, these three species accounted for nearly half (47%) of all recruits observed in the surveys (Fig. 4A). The "unidentified" category accounted for 21% and *Manicina*, *Dichocoenia*, *Favia* and *Agaricia agaricites* composed an additional

23%. The remainder (9%) were mostly reef builders that were only observed once or twice as recruits, including *M. annularis*, *D. strigosa*, *D. clivosa*, *A. cervicornis*, *A. palmata*, and two non-reef-building genera, *Scolymia* and *Mycetophyllia*. Similar recruit frequencies were observed in the low relief-reef crest sites (Fig. 4B), but *Manicina* recruits were more abundant than average at the spur and groove sites (Fig. 4C). Mean coral recruit density ranged from about 0.1-0.5/0.0625 m<sup>2</sup> (~1.6-8 colonies/m<sup>2</sup>) (Table 4).



**Figure 3.** Size-frequency distributions of  $\geq 10$  cm diameter colonies of (A) *Porites astreoides*, *Diploria clivosa* and *D. strigosa* in low-relief outer reef crests (7 sites) and (B) *Porites astreoides* in spurs and grooves (2 sites) off Abaco, Bahamas.



**Figure 4.** Species composition and mean relative abundance of all stony coral recruits ( $\geq 2$  cm diameter) in (A) All Abaco, Bahamas sites, (B) Low-relief outer reef crests (7 sites), and (C) Spurs and grooves (2 sites). Coral species with  $< 1\%$  are placed in a grouped category; other in (A) = *Acropora palmata*, *Diploria clivosa*, *Montastraea annularis*, *Mycetophyllia*, *Scolymia*.

## Stony Coral Condition

**Partial Mortality.** Recent partial-colony mortality (hereafter recent mortality) of the  $\geq 10$  cm stony corals was lowest (nearly zero) in the Sandy Cay back reef and averaged  $< 3\%$  in eight sites (Table 3). The three lowest mean values for old partial-colony mortality (hereafter old mortality) and total (recent and old) mortality (hereafter total mortality) occurred in the two outer reef crests near Man O' War Cay and in the Sandy Cay back reef ( $\sim 8$ - $16\%$  and  $9$ - $16\%$ , respectively). The highest values for both recent mortality ( $6\%$ ) and total mortality ( $31\%$ ) were found in the Elbow Cay north outer

reef crest. Similar patterns of mortality were displayed on corals in the low-relief reef crest and those in the spur-and-groove reefs (Fig. 5A,B). However, more colonies were affected by old mortality than by recent mortality. Also, a greater percentage of the recent mortality occurred on a smaller portion of the affected corals. Old mortality showed a more uniform distribution with respect to amount of the colony affected.

Overall, an average of 3.6% of all the large stony corals were “standing dead” (100% total mortality, still in growth position). Three sites (both near Man O’ War Cay and Fowl Cay pinnacles) had no standing dead colonies while the highest value (8%) was observed in the Elbow Cay north site (Table 3). The species most frequently scored as standing dead were *Acropora palmata*, *Diploria strigosa*, *Millepora complanata*, *M. alvicornis* and *Montastraea annularis*.

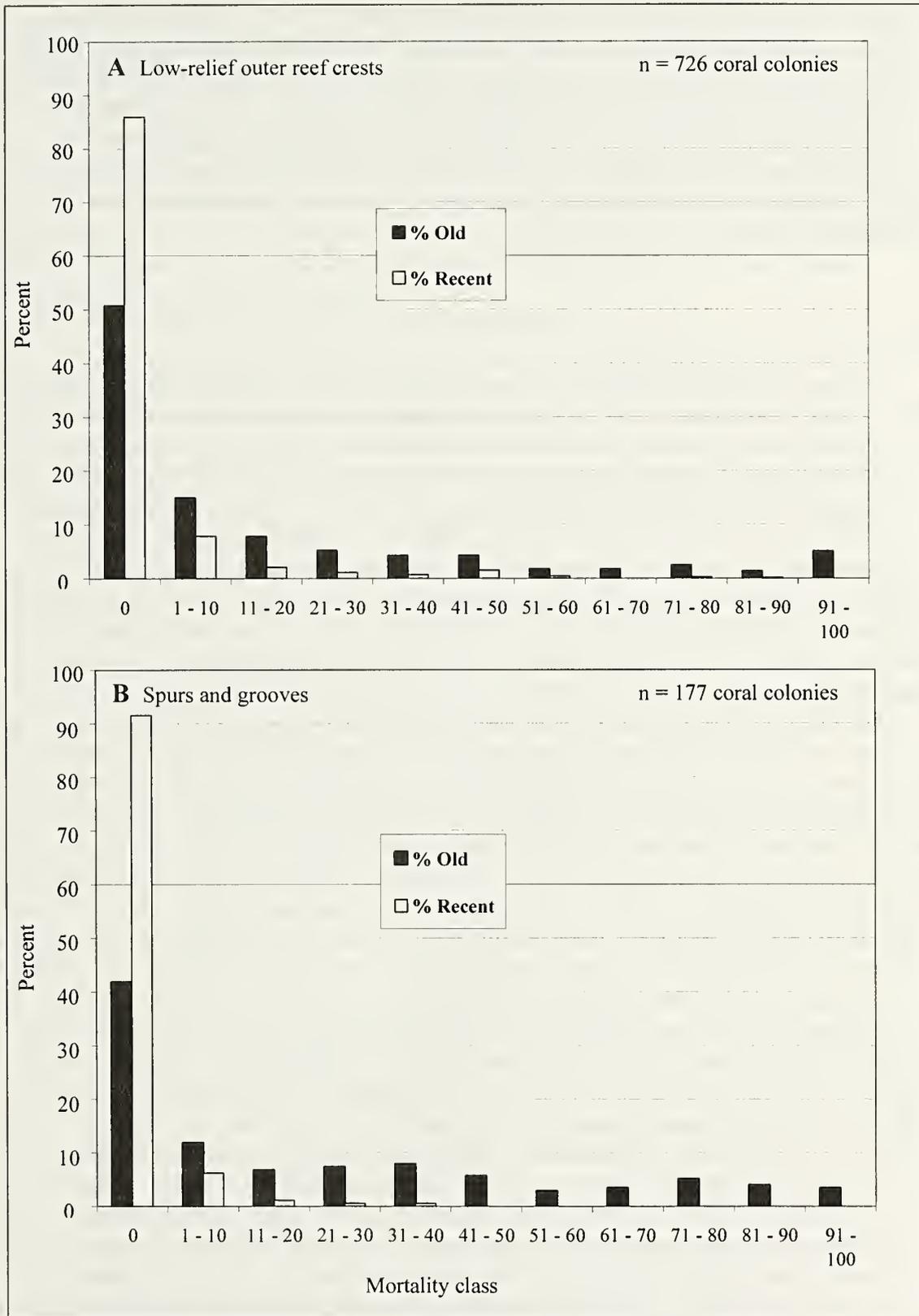
*Diseases.* The overall incidence of coral disease was low, with only 0-1% of all colonies being affected in 11/13 sites (Table 3). Diseased colonies were somewhat more abundant in the Fowl Cay pinnacles (2.5%) and the Sandy Cay fore reef (4%). The most commonly affected species included *Diploria strigosa*, *Montastraea annularis* and *Montastraea faveolata*. Black-band disease affected one surveyed colony each of *M. faveolata*, *Diploria clivosa* and *Colpophyllia natans* and it was noted on two colonies of *Siderastrea siderea* in areas not covered by the transects. Yellow-blotch disease (three colonies) and white plague (two colonies) were also observed. Several other surveyed colonies exhibited tissue sloughing or bands or borders of recently exposed skeleton surrounding live tissues. Due to the unclear etiology it was not possible for us to identify the putative disease agent in these cases and they were classified as “unknown.”

Because the transects did not intercept many of the colonies of *Diploria strigosa* that were sloughing tissues, a haphazard swimming survey was performed in two sites (Elbow Cay middle, outer reef crest and Lynyard Cay north, spurs and grooves). All colonies encountered that were greater than 20 cm in diameter (with some as large as 55 cm) were examined. Of 79 surveyed corals, 26 were dead, 34 were observed to be actively sloughing tissue or with recent partial mortality, and only 19 appeared “healthy.”

*Additional perturbations.* Other incidents of mortality observed in the transects included overgrowth of several coral species by the boring sponge *Cliona* (two colonies of *Porites astreoides* and one colony each of *Diploria labyrinthiformis*, *Montastraea cavernosa*, *Porites porites*, and *Siderastrea siderea*). The hydrocoral, *Millepora alvicornis* was observed to overgrow two colonies of *Porites astreoides* and parrotfish bite-marks were seen on a colony of *Acropora palmata*. Relatively few (0-3%) corals were bleached (Table 3), with entire colonies exhibiting very pale to white tissues.

#### Abundance of Algae and *Diadema antillarum*

Seven hundred fifty-four algal quadrats were examined during the survey. Macroalgal relative abundance values ranged from less than 30% in six sites to nearly 50% in the Elbow Cay north outer reef crest (Table 4). Mean macroalgal heights varied between about 2 cm in the Sandy Cay back reef and 5.5 cm in the Fowl Cay outer reef crest where macroalgal indices (macroalgal relative abundance x macroalgal height, a proxy for macroalgal biomass) were also lowest (64) and highest (184), respectively.



**Figure 5.** Frequency distribution of old and recent partial colony mortality of all stony corals ( $\geq 10$  cm diameter) in (A) Low-relief outer reef-crests (7 sites) and (B) Spurs and grooves (2 sites) off Abaco, Bahamas.

By a large margin, the highest relative abundance of crustose coralline algae (45%) was in the Sandy Cay fore reef (the only site in which they predominated), whereas the lowest was in the outer reef crest at the Man O' War south of south channel site (10%). Overall, turf algae constituted the most abundant algal functional group in the two spur-and-groove sites off Lynyard Cay and in many (5/7) of the low-relief outer reef crests.

Only a few (<5) individuals of the long-spine sea urchin, *Diadema antillarum*, were seen during underwater operations in Abaco and none was observed in the invertebrate belt transect surveys (Table 4).

## DISCUSSION

Coral cover in Abaco was generally similar to that observed further south in the Exuma Cays by Chiappone et al. (1997). The difference in latitude between Abaco and the Exumas is 3° or approximately 300 km. Chiappone et al. (1997) reported average live stony coral coverage of 22.8% for fringing reefs and 11.2% for windward hard-bottom habitats. In our surveys we found mean coverage of 18.5% in the Sandy Cay fringing reef and 13.5% for all other Abaco sites combined. The relative abundance of species in the studies are different, however, and one should note that the Exuma study included patch reefs (hosting *Montastraea annularis*, not widely documented in this survey) and channel reefs, neither of which type of habitat was examined in Abaco. A total of 39 species of scleractinians and *Millepora* were observed in the Exumas (Chiappone et al., 1997) compared with a similar number of taxa (38) in Abaco (this study).

The only previous examination of the reefs in the vicinity of the Hub of Abaco was carried out by Storr (1964) who examined the linear reefs offshore of the northern end of Elbow Cay between 1948 and 1961. His work was mostly qualitative in nature and his descriptions are of habitats that are landward of our outer reef crest site in Storr's Reef. Nevertheless, comparisons of general reef health between 1999 and the mid-twentieth century are interesting. Storr (1964) noted that *Porites astreoides* was abundant in all sites from offshore to inshore. *Acropora palmata* was abundant in the more seaward, shallow sites (reef flats), becoming patchy inshore. He described the appearance of *A. palmata* as healthy except at an inshore site where the corals were dead and covered with macroalgae. In contrast, *A. palmata* observed in this study showed high values of partial-and/or total- colony mortality but it is unknown when these colonies began to die. *Porites astreoides*, however, is still abundant. The most dramatic difference between the two time periods was the great abundance of *Diadema antillarum* found by Storr compared with the very low densities observed presently, a continuing result of the Caribbean-wide 1983 die-off (Lessios et al., 1984). Storr's (1964) observations of algal cover are too general for detailed comparison with our study (which otherwise might have allowed some correlation between abundance of *Diadema* and algal coverage).

Comparison of the study sites within protected (from fishing) and unprotected areas showed slightly higher coral cover in the protected sites. Moreover, the largest massive corals were found on the Fowl Cay pinnacles, which are protected, and in the Lynyard Cay spurs and grooves, which are distant from human development. Interpretation of these differences should be made cautiously, however, since the protected areas were chosen for their high quality (as observed by divers) and there are

relatively few replicates (four protected versus nine that are unprotected). A number of confounding natural factors, such as depth, substratum, slope, structural complexity and exposure to wave energy can affect coral populations so further study is warranted before the effect on the coral community of protecting these Abaco reefs can be evaluated.

The predominance of corals such as *Porites astreoides*, *Diploria clivosa*, *Diploria strigosa* and, in some cases, *Acropora palmata*, indicates that many of the surveyed sites are chronically exposed to relatively high wave energy and surge. The absence of large mounding corals, except in the deeper habitats on the Fowl Cay pinnacles (where small species, such as *Agaricia* spp., are also very common) and in the spur-and-groove formations at Lynyard Cay, is striking.

The high relative abundance of *Porites astreoides* recruits is consistent with their high frequency as adult colonies in the surveyed reefs and their brooding mode of larval development (Chornesky and Peters, 1987). Due to the ambiguities of coral recruit taxonomic identification, the data are difficult to interpret with respect to relative abundance of species of *Diploria*, *Montastraea*, etc. which, when small, are only identifiable to the genus level. The relatively few recruits encountered (Table 4) also precluded interpretation of patterns of recruitment at the level of site or reef type.

Many colonies of *Montastraea faveolata* and *Diploria strigosa* were observed with pale and dead tissue patches, perhaps due to environmental or biological stressors. Caribbean-wide warming occurred in the summer of 1999 and high-temperature exposure can result in bleaching and partial mortality (Williams and Bunkey Williams, 1990). Alternatively, coral diseases may have impacted these corals. It can be difficult to ascribe tissue necrosis to specific disease conditions (e.g., Richardson, 1998). Black-band disease has the clearest etiology; however, white-band disease, white plague, and other “white-line diseases” are harder to diagnose during snapshot field surveys.

Some of the lowest (both Man O’ War Cay sites) and highest (Elbow Cay north) values for partial-colony mortality and for standing dead were found within the same reef habitat type (low-relief outer, reef crests). There were no apparent differences in terms of anthropogenic impact, either direct or indirect, in any of these reefs, nor were there clear natural impacts that could account for the differences. As few replicates were present for the other habitat types, it is not possible to generalize about coral mortality distribution patterns.

Turf algae were very common but were trapping little sediment and appeared to have caused no stony coral mortality (personal observations). Macroalgal height and relative abundance trends may show some relationship to the degree of human impact in the region, as they are relatively low in the southern Sandy Cay fringing reefs, and some of the highest values occurred in the Elbow Cay reefs near Hopetown (Table 4). Insight into macroalgal biomass trends may be found by examining the role of herbivorous fishes in controlling macroalgae. Large groupers that were observed in the northern Fowl Cay pinnacles are protected from fishing and subjected to fish feeding by the local dive charter operators. Predation by carnivorous fishes off Fowl Cay may locally deplete the herbivorous fish populations leading to an increase in macroalgal biomass. Sandy Cay is similarly protected from fishing but not subjected to significant fish feeding and few large groupers were observed here, which may help to explain its lower macroalgal biomass values.

The AGRRA methods provide a good snapshot of the reefs of central Abaco and, as the results are compared to those of other Western Atlantic and Caribbean reefs (Kramer, this volume), may reveal large-scale trends in their stony coral, algal and fish communities. Management strategies on a local scale, however, should depend on studies that investigate the processes that affect reef communities. These investigations should include the impacts of anthropogenic factors such as water quality, fisheries and anchor damage, as well as the natural physical processes that may play an important role in controlling reef structure.

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Table 1. Site information for AGRRA stony coral and algal surveys in Abaco, Bahamas.

Site name	Site code/ status	Reef type	Latitude (°' N)	Longitude (°' W)	Survey date	Depth (m)	Benthic transects (#)	≥10cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)
<b><i>Fringing reef</i></b>									
Sandy Cay, Forereef	SCF/MPA <sup>1</sup>	Fore-reef slope	26 23.867	76 59.308	Aug 9 99	1-6	12	8.5	22.5 ± 11.5
Sandy Cay, Backreef	SCB/MPA	Back reef	26 23.867	76 59.308 <sup>2</sup>	Aug 9 99	2-7	15	8.5	14.0 ± 7.5
<b><i>Bank barrier reefs</i></b>									
Elbow Cay, South- <i>inner</i>	ECSI	Low relief outer reef crest	26 31.606	76 57.417	Aug 11 99	4-8	14	8.5	14.5 ± 4.5
Elbow Cay, Middle	ECM	Low relief outer reef crest	26 32.419	76 56.767	Aug 11 99	6-9	13	8	12.0 ± 9.0
Elbow Cay, North	ECN	Low relief outer reef crest	26 33.686	76 56.969	Aug 16 99	6-11	18	5.5	8.5 ± 4.5
Fowl Cay- <i>shallow</i>	FCS/MPA	Low relief outer reef crest	26 38.229	77 02.310	Aug 13 99	8-12	15	8	12.0 ± 4.5
Storr's Reef	SR	Low relief outer reef crest	26 34.654	76 57.538	Aug 16 99	2-6	15	8	14.5 ± 7.5
Man O' War Cay, N. of S. Channel	MOWSN	Low relief outer reef crest	26 36.197	76 58.987	Aug 12 99	6-9	14	5	10.5 ± 6.0
Man O' War Cay, S. of S. Channel	MOWSS	Low relief outer reef crest	26 35.835	76 58.583	Aug 12 99	2-5	14	8	15.0 ± 5.5
Lynyard Cay, North	LCN	Spur and groove	26 21.452	76 58.612	Aug 15 99	6-10	16	5.5	14.0 ± 12.5
Lynyard Cay, South	LCS	Spur and groove	26 21.212	76 58.680	Aug 15 99	6-11	15	6	13.0 ± 6.0
Elbow Cay, South- <i>outer</i>	ECSO	Shallow pinnacles	26 31.937	76 57.023	Aug 10 99	2-6	12	8	18.0 ± 5.5
Fowl Cay- <i>deep</i>	FCP/MPA	Deep pinnacles	26 38.244	77 02.129	Aug 13 99	10-16	13	6	17.0 ± 13.5
<b>All sites</b>						<b>1-16</b>	<b>13</b>	<b>7.1</b>	<b>14.3 ± 7.6</b>

<sup>1</sup>MPA designates that the site was in a marine protected area.

<sup>2</sup>Exact GPS coordinates were not obtained. The back-reef site was located approximately 150m ESE of the fore-reef site.

Table 2. Relative abundance of all stony corals seen during AGRRA surveys in Abaco.

Name	Relative abundance <sup>1</sup>	Name	Relative abundance <sup>1</sup>
<b>Milleporidae</b>			
<i>Millepora alcicornis</i>	Abundant		
<i>Millepora complanata</i>	Abundant		
<b>Scleractinia</b>			
<i>Acropora cervicornis</i>	Rare	<i>Montastraea annularis</i>	Abundant
<i>Acropora palmata</i>	Abundant	<i>Montastraea faveolata</i>	Abundant
<i>Agaricia agaricites</i> f. <i>agaricites</i>	Abundant	<i>Montastraea franksi</i>	Rare
<i>Agaricia agaricites</i> f. <i>danai</i>	Common	<i>Montastraea cavernosa</i>	Common
<i>Agaricia tenuifolia</i>	Few	<i>Manicina areolata</i>	Abundant
<i>Colpophyllia natans</i>	Common	<i>Meandrina meandrites</i>	Few
<i>Dendrogyra cylindrus</i>	Rare	<i>Mussa angulosa</i>	Few
<i>Dichocoenia stokesi</i>	Common	<i>Mycetophyllia danaana</i>	Few
<i>Diploria clivosa</i>	Abundant	<i>Mycetophyllia ferox</i>	Common
<i>Diploria labyrinthiformis</i>	Common	<i>Mycetophyllia lamarckiana</i>	Common
<i>Diploria strigosa</i>	Abundant	<i>Porites astreoides</i>	Abundant
<i>Eusmilia fastigiata</i>	Common	<i>Porites branneri</i>	Common
<i>Favia fragum</i>	Common	<i>Porites porites</i>	Common
<i>Isophyllastrea rigida</i>	Rare	<i>Scolymia</i> spp.	Common
<i>Isophyllia sinuosa</i>	Rare	<i>Stephanocoenia intersepta</i>	Common
<i>Leptoseris cucullata</i>	Common	<i>Siderastrea radians</i>	Common
<i>Madracis decactis</i>	Few	<i>Siderastrea siderea</i>	Abundant
<i>Madracis pharensis</i>	Few	<i>Tubastraea coccinea</i>	Rare

<sup>1</sup>Abundant = seen numerous times on more than half of the 13 dives, Common = seen on less than half of all dives but more than 10 times, Few = seen 4 to 10 times, Rare = seen 3 times or less. Relative abundance categories were determined during debriefings of all Abaco team members following each day of diving and at the end of the expedition.

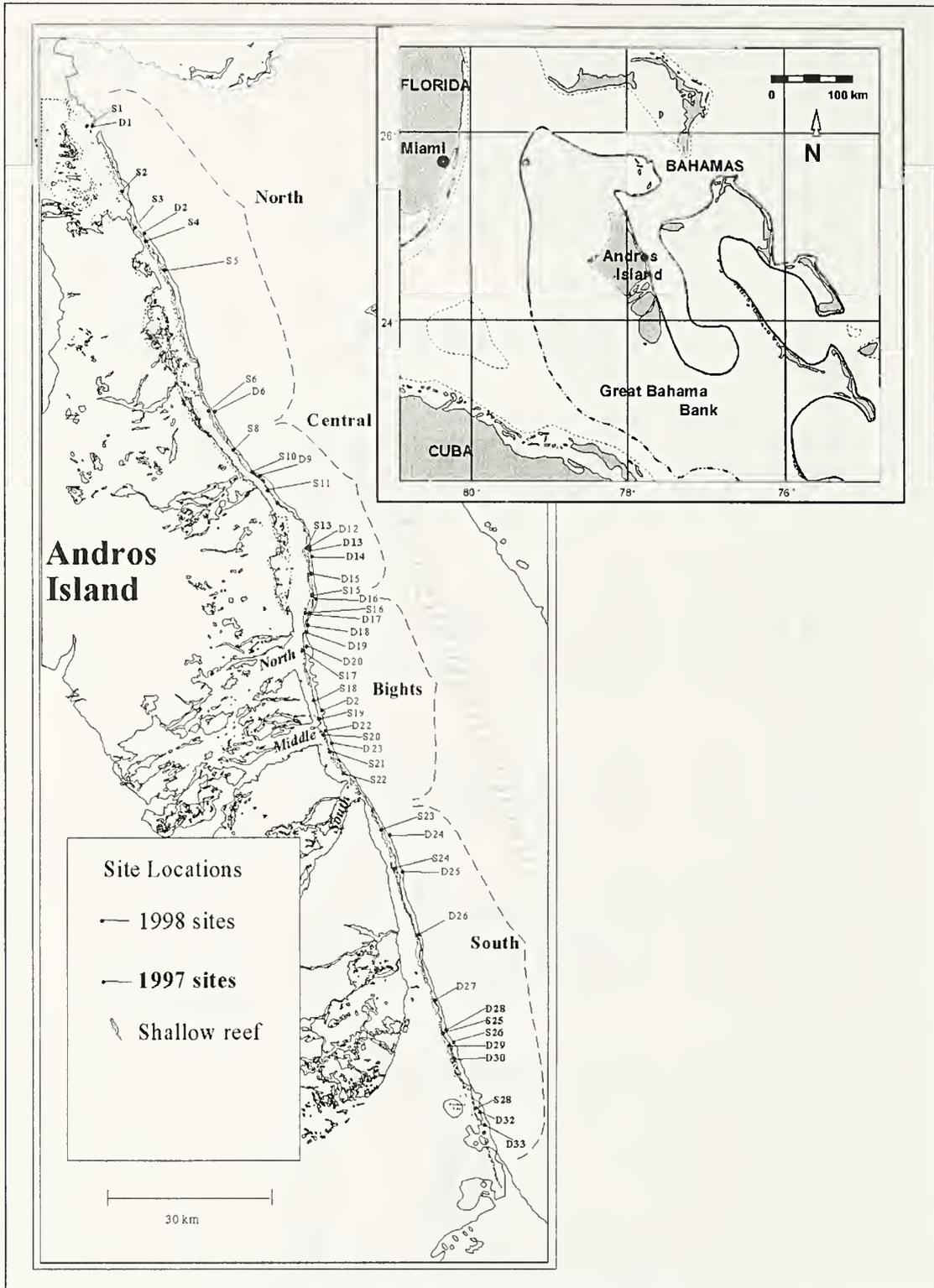
Table 3. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 10$  cm diameter) by site in Abaco.

Site name	Stony corals		Partial-colony surface mortality (% $\pm$ sd)			Stony corals (%)		
	(#)	Diameter (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased
<b><i>Fringing reef</i></b>								
Sandy Cay, Foreereef	103	62.0 $\pm$ 60.0	1.0 $\pm$ 3.5	22.0 $\pm$ 32.5	23.0 $\pm$ 32.5	6	0	4
Sandy Cay, Backreef	125	38.0 $\pm$ 33.0	<0.5 $\pm$ 1.0	15.5 $\pm$ 26.0	16.0 $\pm$ 26.0	1.5	1	1
<b><i>Bank barrier reefs</i></b>								
Elbow Cay, South-inner	118	37.0 $\pm$ 29.0	4.5 $\pm$ 14.0	22.0 $\pm$ 30.0	26.0 $\pm$ 32.5	3.5	2	0
Elbow Cay, Middle	105	39.0 $\pm$ 46.5	3.5 $\pm$ 11.5	22.5 $\pm$ 31.5	25.5 $\pm$ 33.5	5.5	1	0
Elbow Cay, North	97	41.0 $\pm$ 41.5	6.0 $\pm$ 14.0	25.0 $\pm$ 35.0	31.0 $\pm$ 37.5	8	0	0
Fowl Cay-shallow	121	33.5 $\pm$ 29.0	1.0 $\pm$ 6.0	19.0 $\pm$ 27.0	20.0 $\pm$ 28.5	6.5	0	1
Storr's Reef	121	43.5 $\pm$ 48.5	2.5 $\pm$ 9.0	19.5 $\pm$ 32.0	22.0 $\pm$ 33.5	7.5	1	0
Man O' War Cay, N. of S. Channel	71	32.5 $\pm$ 22.5	3.5 $\pm$ 10.0	12.5 $\pm$ 20.5	16.0 $\pm$ 23.5	0	0	0
Man O' War Cay, S. of S. Channel	109	35.0 $\pm$ 29.0	1.0 $\pm$ 4.5	8.0 $\pm$ 16.0	9.0 $\pm$ 17.0	0	0	0
Lynyard Cay, North	89	57.5 $\pm$ 72.0	1.0 $\pm$ 2.5	25.5 $\pm$ 31.5	26.5 $\pm$ 31.5	3.5	0	1
Lynyard Cay, South	88	47.5 $\pm$ 63.0	2.0 $\pm$ 6.5	25.5 $\pm$ 31.0	27.5 $\pm$ 31.5	3.5	0	1
Elbow Cay, South-outer	98	38.0 $\pm$ 39.5	1.0 $\pm$ 4.5	20.5 $\pm$ 29.0	21.0 $\pm$ 30.0	1	3	1
Fowl Cay-deep	77	49.0 $\pm$ 60.5	3.0 $\pm$ 13.0	18.0 $\pm$ 23.5	21.0 $\pm$ 26.0	0	1.5	2.5

Table 4. Algal characteristics, and density of stony coral recruits and of *Diadema antillarum* (mean  $\pm$  standard deviation) by site in Abaco.

Site name	Quadrats (#)	Relative abundance (%)			Macroalgal		Coral recruits (#/0.0625 m <sup>2</sup> )	<i>Diadema</i> (#/10 m)
		Macroalgae	Turf algae	Crustose coralline algae	Height (cm)	Index <sup>1</sup>		
<b><i>Fringing reef</i></b>								
Sandy Cay, Forereef	60	24.5 $\pm$ 28.0	30.5 $\pm$ 20.5	45.0 $\pm$ 26.5	3.0 $\pm$ 5.0	81	0.3 $\pm$ 0.8	0
Sandy Cay, Backreef	64	28.0 $\pm$ 23.0	51.0 $\pm$ 28.5	21.0 $\pm$ 24.0	2.0 $\pm$ 1.0	64	0.3 $\pm$ 1.2	0
<b><i>Bank barrier reefs</i></b>								
Elbow Cay, South-inner	45	39.0 $\pm$ 24.0	40.0 $\pm$ 22.0	21.0 $\pm$ 20.0	3.5 $\pm$ 2.0	133	0.2 $\pm$ 0.2	0
Elbow Cay, Middle	65	36.5 $\pm$ 24.0	46.0 $\pm$ 23.0	22.5 $\pm$ 19.0	3.5 $\pm$ 1.0	114	0.2 $\pm$ 0.5	0
Elbow Cay, North	60	47.0 $\pm$ 34.5	37.5 $\pm$ 30.5	15.5 $\pm$ 17.0	3.5 $\pm$ 1.0	155	0.2 $\pm$ 0.6	0
Fowl Cay-shallow	60	32.5 $\pm$ 23.0	50.0 $\pm$ 24.0	17.5 $\pm$ 19.5	5.5 $\pm$ 9.5	184	0.2 $\pm$ 0.5	0
Storr's Reef	55	29.0 $\pm$ 27.0	54.0 $\pm$ 30.5	17.5 $\pm$ 21.0	2.5 $\pm$ 1.0	72	0.3 $\pm$ 0.1	0
Man O' War Cay, N. of S. Channel	60	25.0 $\pm$ 21.0	57.0 $\pm$ 28.5	17.0 $\pm$ 22.0	2.5 $\pm$ 1.5	68	0.1 $\pm$ 0.3	0
Man O' War Cay, S. of S. Channel	45	28.5 $\pm$ 15.0	61.5 $\pm$ 21.5	10.0 $\pm$ 16.0	3.5 $\pm$ 1.5	106	0.2 $\pm$ 0.5	0
Lynyard Cay, North	60	29.5 $\pm$ 24.5	53.0 $\pm$ 26.0	17.5 $\pm$ 16.5	4.0 $\pm$ 3.0	124	0.3 $\pm$ 0.6	0
Lynyard Cay, South	60	34.5 $\pm$ 21.5	46.5 $\pm$ 25.0	19.0 $\pm$ 19.5	3.5 $\pm$ 2.0	125	0.5 $\pm$ 0.7	0
Elbow Cay, South-outer	60	44.5 $\pm$ 31.0	38.0 $\pm$ 27.5	17.5 $\pm$ 23.0	3.0 $\pm$ 1.5	130	0.1 $\pm$ 0.4	0
Fowl Cay-deep	60	44.5 $\pm$ 24.0	39.0 $\pm$ 22.0	16.5 $\pm$ 17.0	3.0 $\pm$ 1.0	141	0.3 $\pm$ 0.5	0

<sup>1</sup>Macroalgal index = relative abundance x macroalgal height



**Figure 1.** Location of Andros Island, the Bahamas (inset) and the 1997 and 1998 AGRRA survey sites in four areas (North, Central, Bights, South) Andros. See Tables 1A, 1B for site codes.

# ASSESSMENT OF THE ANDROS ISLAND REEF SYSTEM, BAHAMAS (PART 1: STONY CORALS AND ALGAE)

BY

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GINSBURG<sup>1</sup>

## ABSTRACT

Spatially extensive shallow reef crests with high coral cover (particularly *Acropora*) and fore reefs with high topographic complexity were documented during the 1997–1998 rapid assessments of coral and algal indicators in the relatively remote Andros reef system. Apart from the ecological effects of the *Diadema antillarum* die-off, evidence of major disturbance events (hurricanes, bleaching) were not apparent in 1997, although chronic levels of disease and predation were higher than in other Western Atlantic reef areas. High macroalgal abundances in fore reefs are attributed to low grazing pressures whereas the high densities of herbivorous fishes limit macroalgae in reef-crest habitats. During 1998, warm sea surface temperatures triggered localized bleaching in reef crests and appear to have facilitated widespread outbreaks of disease with resulting high mortality of massive corals in many fore reefs.

## INTRODUCTION

Recent declines in abundance and diversity of corals in the Western Atlantic are well documented in areas where local stressors such as overfishing, nutrification and sedimentation can be significant (see Ginsburg and Glynn, 1994). Less well known are the condition of coral reefs in more remote areas where anthropogenic influences are less important but where regional impacts caused by bleaching, diseases and hurricanes may be significant. To better understand the influences of regional versus local impacts, we examined the condition of a relatively remote area in the Bahamas, the Andros reef tract. Andros is separated both geographically and hydrographically from the greater Caribbean which may decrease the exchange of larvae and/or water-borne pathogens with other reefs of the region. Regional disturbances, such as the die-off of many acroporids from white-band disease (WBD) (Aronson and Precht, 1997, 2001) and detailed effects of the 1983 Caribbean-wide *Diadema* urchin die-off (Lessios et al., 1984), are poorly documented in Andros.

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The extent that anthropogenic impacts have affected the reefs off Andros is poorly quantified but is generally assumed to be minimal compared with many areas of the Caribbean. Andros has a very low population of about 8,155 (Bahamas Reef Environmental Education Foundation and MacAlister Elliot and Partners, 1998, unpublished report) with most of its populace concentrated in several small towns along the eastern coast of the island. Androsians rely on fishing, sponging and small-scale tourism for the majority of their income while additional employment comes from the Atlantic Underwater Test and Evaluation Centers (AUTEK), Androsia (a local batik factory), and small-scale farming. North Andros, used for timbering until 1975, now contains a small-scale agricultural industry (Sealey, 1990) but pollution and runoff are believed to be limited. The few small-scale resorts on Andros (<300 rooms total) cater towards tourism focused on guided bonefishing or diving the coral reefs.

Spatial trends in various indicators of the principal reef-building corals and algal populations that were surveyed during 1997 and 1998 using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) method are described in this paper. The condition of reef fish populations in Andros is presented in Kramer, Marks and Turnbull (this volume).

## METHODS

Andros Island has the largest land area (6,000 km<sup>2</sup>) of the Bahamas Archipelago and is located on the eastern margin of the western arm of Great Bahamas Bank where it abuts against the Tongue of the Ocean (Fig. 1). The extensive Andros reef tract parallels the eastern side of the island from Joulter Cays in the north to Saddleback Cays in the south (a total of 217 km). The reef is discontinuous in many places, particularly around the central and southern portion of Andros where several large channels (bights) cut across the entire island dividing it into three distinct landmasses. Smith (1948) and later Newell et al. (1951) both used the term "bank reef" to describe the Andros reef tract although it has also been described as a fringing reef, barrier reef, and fringing-barrier reef (which is probably the most accurate). The reef crest typically lies 1-2 km from the shoreline and is situated atop a Pleistocene ridge that is exposed in places in the form of small elongate offshore cays (e.g., Staniard Rock, Goat Cay, Long Rock, Long Bay Cays, Grassy Creek Cays). Fore reefs are best developed as high-relief features (e.g., pinnacles) composed of often dense aggregations of massive corals (primarily the *Montastraea annularis* species complex) and occur at intermediate depths between 7 m to 12 m, being most extensive in "low-to-intermediate" wave-energy environments. Well-developed reef crests and fore reefs often occur together in central and southern Andros. The outer reef slope contains a break at approximately 20 m followed by the shelf edge at 30 m that is marked by a near-vertical wall rich in sponges and other invertebrates.

For this study, the Andros reef tract was stratified into shallow crests (1-3 m deep) and intermediate-depth fore reefs (8-12 m deep), which were further subdivided into "well developed" and "poorly developed" categories for both depths using low altitude (200 m) oblique aerial photographs taken in July, 1997. Lagoonal patch reefs and deeper shelf-margin reefs were not investigated. Reconnaissance inspections or Manta tows were undertaken in each area to get an overview of reef type before haphazardly selecting representative well-developed sites to survey. In 1997, 26 sites (13 at 1-3 m depth, 13 at

8-12 m) located primarily in the northern and southern sections of Andros Island were examined. In 1998, 28 (15 deep and 13 shallow) sites along 75 km of central Andros were surveyed (Fig. 1; Tables 1A, 1B). Four sites overlapped between the 1997 and 1998 surveys (S6-S7; D6-D7; D9-D10; D12-D13). All surveyors were trained in the AGRRA method and consistency exercises were conducted regularly to minimize differences among observers. For the two survey periods, a total of six persons participated in the benthic surveys with three surveyors collecting approximately 75% of all the data.

The total number of 10-m benthic transects deployed at each site ranged from 6 to 24 and video was recorded on some transects for archival purposes. In 1997, when a modified Version 1 of the AGRRA protocols was used, live stony coral cover was not quantified. Visual estimates of old mortality did not factor out open spaces between columns for lobate colonies such as *M. annularis* and recent mortality estimates included sides of colonies. Maximum (rather than average) macroalgal heights were recorded within quadrats. In addition, after estimating their abundance in the uppermost canopy layer of the algal quadrats, macroalgae were removed prior to estimating the abundance of underlying turf and crustose coralline algae. We substituted the newly adopted Version 2 of the AGRRA protocols for the 1998 surveys (see Appendix One, this volume), expanding the size and health assessments to all stony corals that were at least 10 cm in diameter for approximately half of the sites. Coral sizes were measured to the nearest 5 cm in both years. Identification of corals was based on Humann (1992).

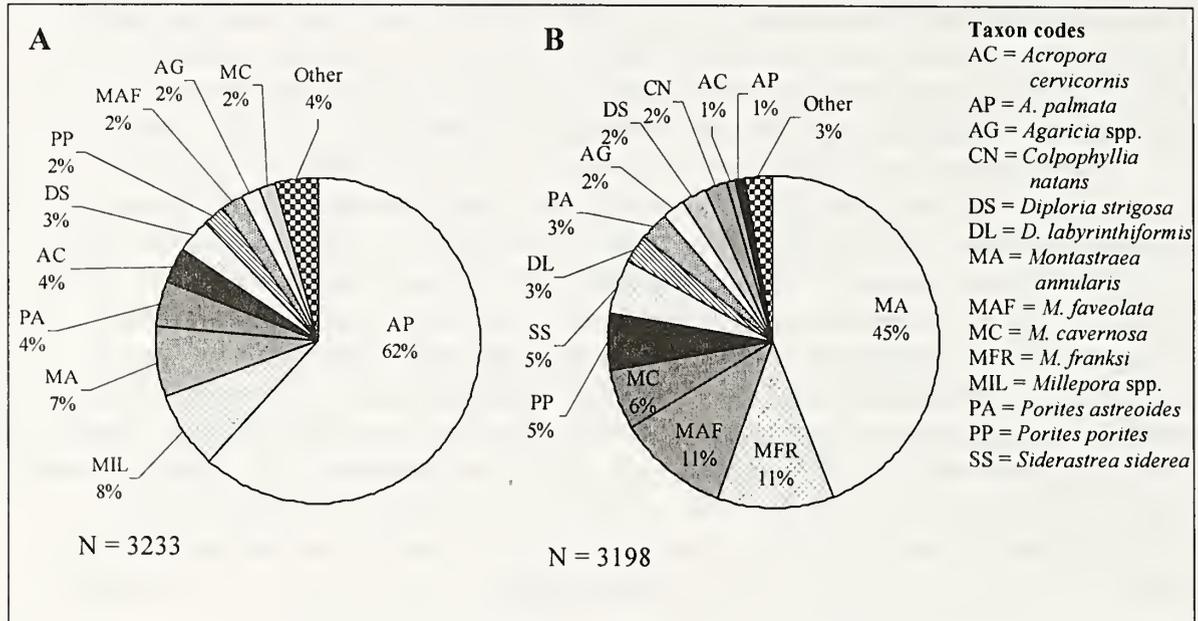
Statistical analysis was performed with the Statistica (version 5.1) program. To permit comparisons between years, all stony corals <25 cm in diameter were excluded from these analyses and the *Montastraea annularis* species complex was treated as a single entity. To statistically characterize variance for the benthic indicators, only the first 10 transects were analyzed from each site. Size data were log(X+1)-transformed and percentage data were arcsine-transformed. Parameters were analyzed by student's t-test and by 1-way and 2-way Analysis of Variance (ANOVA). For spatial analysis, the Andros sites were divided into four areas (North, Central, Bights, and South) (Fig. 1) and analyzed by ANOVA with sites hierarchically nested under areas as random factors.

## RESULTS

### Species Composition and Abundance of Stony Corals

A total of 3,221 stony corals were examined in 28 reef crests in Andros during the two years of field work. They were dominated by colonies of *Acropora palmata* and displayed varying degrees of development controlled, in part, by wave energy, reef aspect, and the presence of freshwater creeks. China Point (S10) was one of the few shallow sites that contained monotypic thickets of living *A. cervicornis*, although "standing dead" (colonies completely dead and still in growth position) thickets were encountered at several other sites. Live stony coral cover ranged from 20-54% (mean=36, sd=18) in 1998 in shallow reefs (Table 1A). A total of 16 "large" ( $\geq 25$  cm) scleractinian and one hydrozoan species were recorded in the shallow crests (all reefs combined). *Acropora palmata* comprised a mean of 62% of all these large corals (Fig. 2A). The relative abundance of subdominant taxa on reef crests (e.g., *M. annularis*, *Millepora*

*complanata*, *Porites astreoides*) was remarkably uniform across the reef areas (Fig. 3A). The most common size range for the colonies of *A. palmata* was 120-140 cm (mean diameter=137, sd=77) with the largest colonies exceeding 400 cm in diameter (Fig. 4A).

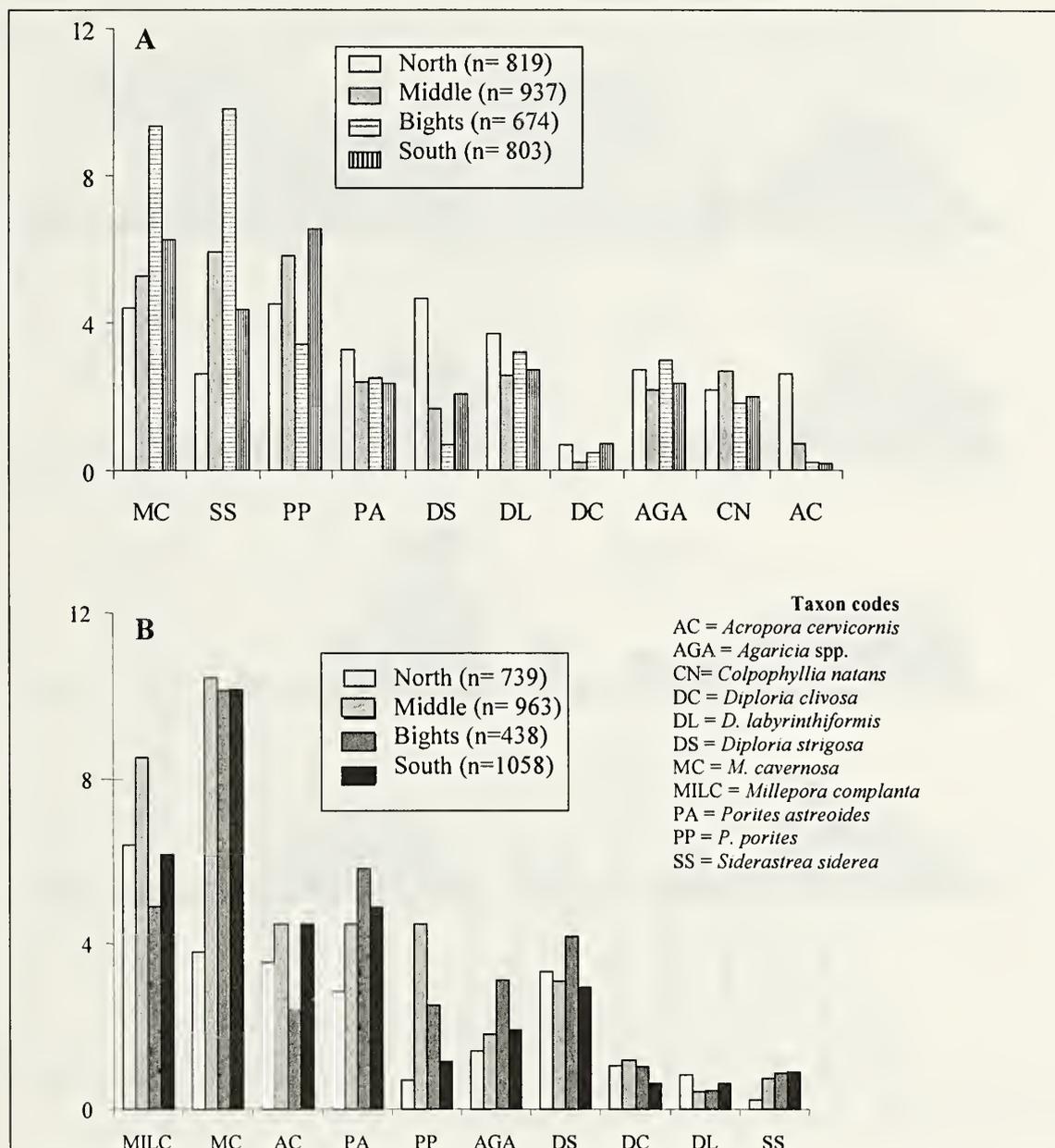


**Figure 2.** Species composition and mean relative abundance of all stony corals ( $\geq 25$  cm diameter) on (A) reef crests and (B) fore reefs in Andros.

Its colony diameters were significantly smaller in the bights area (mean=118 cm for S18-S22) compared with the other areas along Andros (mean=152 cm) (ANOVA df=1, MS=37, F=18.2,  $p < 0.0001$ ) (Fig. 4A). Average colony diameters overall were largest in the northern area and smallest in the bights area (Table 2A).

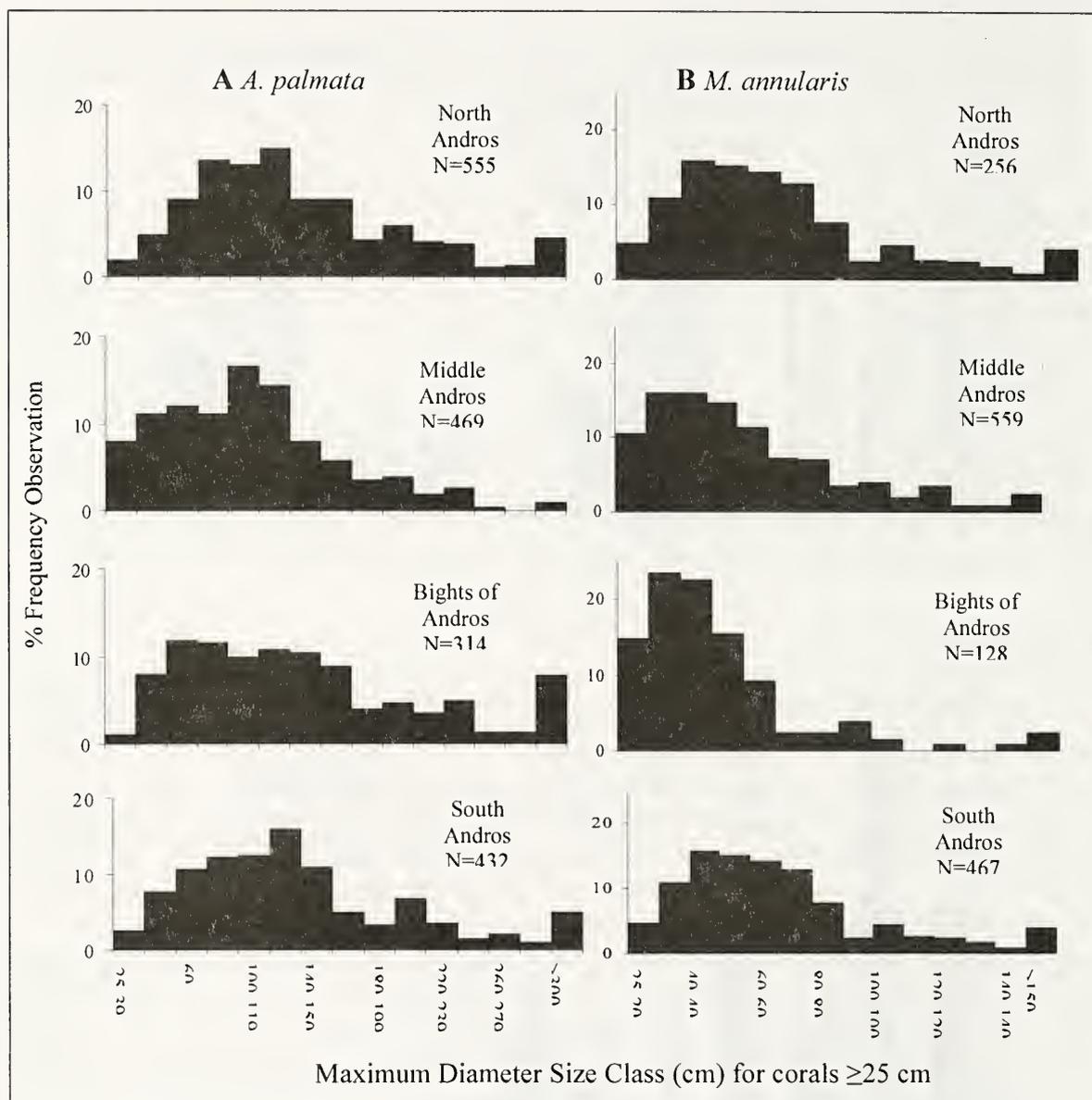
In the 32 intermediate-depth fore reefs that were surveyed, a total of 3,156 large corals were assessed. Fore reefs were mainly dominated by assemblages of *M. annularis* (Fig. 2B) and they were best developed around protected channels, cays, and seaward of reef crests. Live stony coral cover in 1998 ranged from 6 to 45% (mean=22.6, sd=13.8) in the fore reefs (Table 1B). Several sites in the bights area (e.g., D18, D19) had low coral cover ( $< 10\%$ ) and one site (D23) contained few corals greater than 25cm in diameter. A total of 18 species of large stony corals were counted within fore reef transects (all reefs combined), 67% of which were *M. annularis* complex (*M. annularis*, *M. faveolata*, *M. franksi*) (Fig. 2B). An exception to this pattern was the dominance of *S. siderea* rather than *M. annularis* at several sites in the bights area (Fig. 3B). Colonies of *M. annularis* were most common in the 40-50 cm size range (average=64, sd= 8), yet some individuals exceeded 250 cm in diameter (Fig 4B). Colony sizes were negatively correlated with water depth ( $r^2 = 0.03$ ,  $p < 0.0001$ ) and significantly smaller in the reefs surveyed in 1998 (mean diameter=54 cm) than in 1997 (mean diameter=70 cm) (t-test, df=1443,  $t = 10.1$ ,  $p < 0.0001$ ). As in the reef crests, colonies were largest in the northern area and smallest in the bights (Table 2A).

The number of species of stony coral recruits ( $\leq 2$  cm) was greater in fore reefs (26 species, all reefs combined) than in reef crests (16 species, all reefs combined). Small stony coral densities were also significantly higher in deeper (equivalent to 10.6 recruits/m<sup>2</sup>) than in shallower sites (equivalent to 2.9 recruits/m<sup>2</sup>) (t-test,  $t = 12.7$ , df = 544,



**Figure 3.** Relative species abundance of the 10 most abundant stony corals ( $\geq 25$  cm diameter) on (A) reef crests (here excluding *Acropora palmata*), (B) fore reefs (here excluding the *Montastraea annularis* complex) in four areas of Andros.

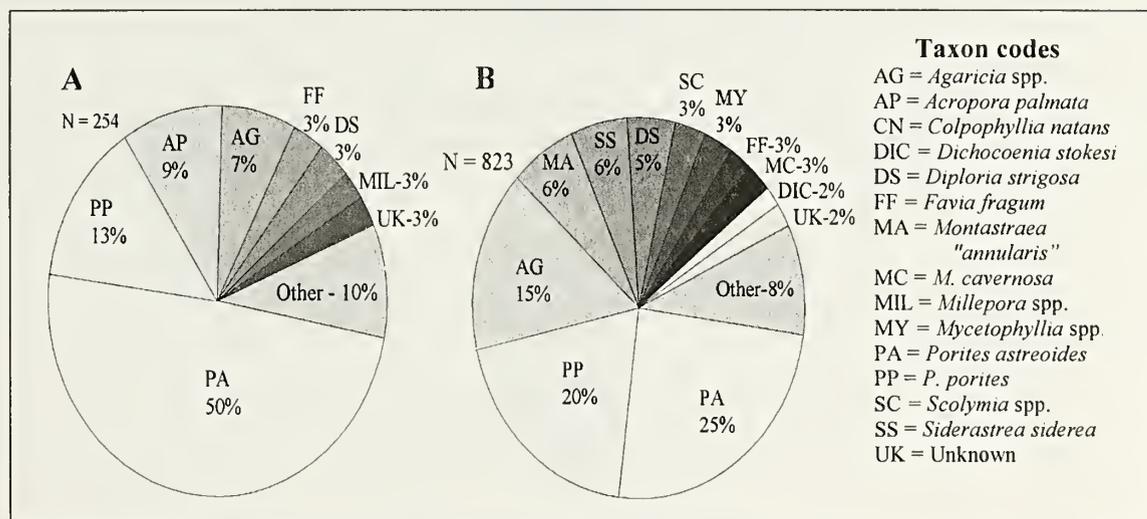
$p < 0.0001$ ) but showed a high degree of variability among reefs (Tables 3A,3B). *Porites astreoides* was the most abundant recruit in reef crests (50%) and fore reefs (25%) despite low abundances of large adults (~4% in reef crests, 3% on fore reefs) (Fig. 5). *Acropora palmata* was the third most abundant (9%) recruit in shallow reefs with the *M. annularis* complex as the fourth most abundant (6%) in deep reefs.



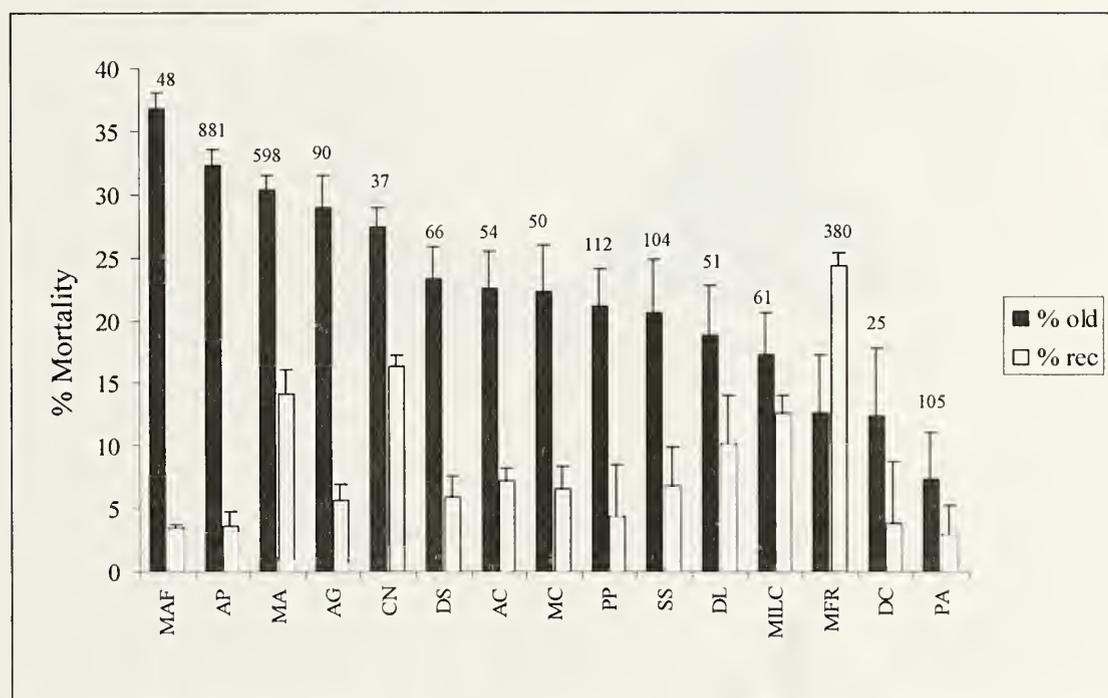
**Figure 4.** Size-frequency distribution of all colonies ( $\geq 5$  cm diameter) of (A) *Acropora palmata* on reef crests and (B) *Montastraea annularis* on fore reefs in four areas off Andros.

### Stony Coral Condition

Levels of old partial-colony mortality (hereafter old mortality) in large stony corals ranged from 12% (S5) to 52% (S25) (Table 2) and were higher in reef crests than fore reefs for both survey years (2-way ANOVA,  $df=581$ ,  $MS=1787$ ,  $F=12.5$ ,  $p<0.0001$ ). The frequency of 100% standing dead colonies was 10%. Reefs surveyed in 1997 had significantly higher old mortality (crests = 38%, fore reefs = 29%) than those surveyed in 1998 (crests = 29%, fore reefs = 22%) (2-way ANOVA,  $df=581$ ,  $MS=5665$ ,  $F=40.4$ ,  $p<0.0001$ ). Large differences in old mortality were apparent among species, with acroporids ( $n=1,764$ ) accounting for the highest levels of old mortality (48% for *A.*

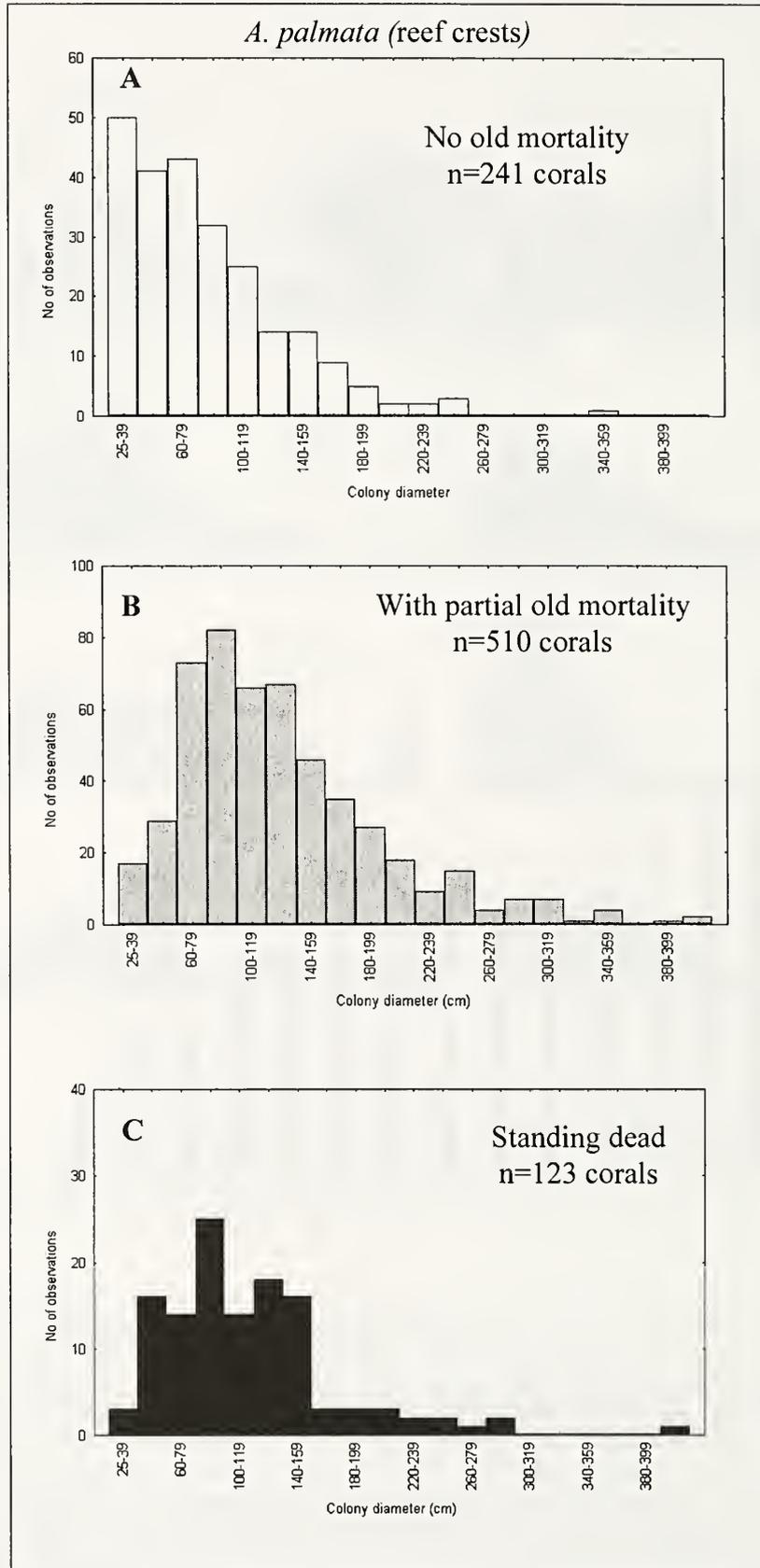


**Figure 5.** Species composition and mean relative abundance of stony coral recruits ( $\leq 2$  cm diameter) in (A) reef crests and (B) fore reefs off Andros.

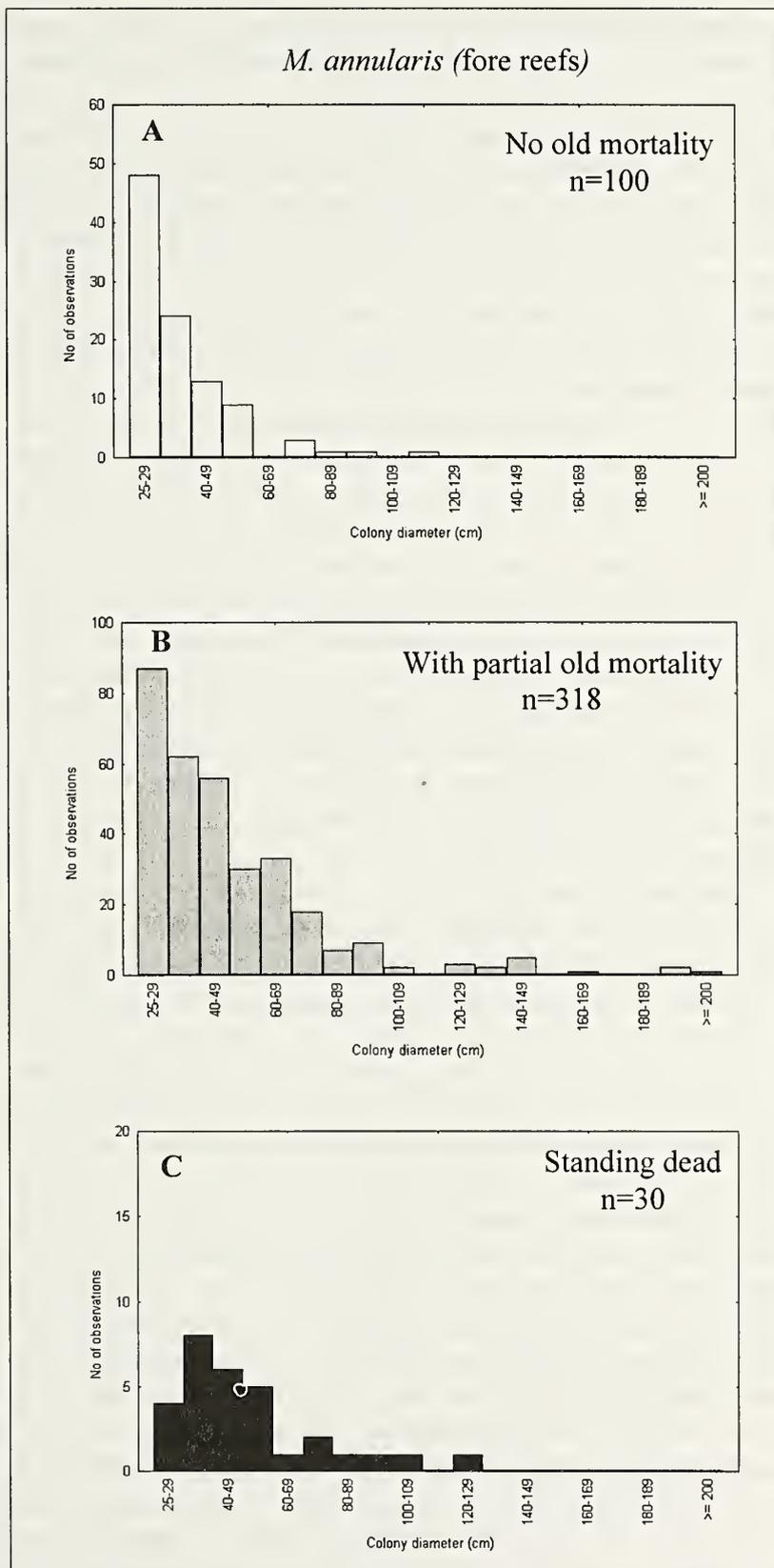


**Figure 6.** Percent old and recent (rec) partial colony mortality (mean  $\pm$  standard error) in 1998 for the 15 most abundant  $\geq 25$  cm stony coral species; both survey depths combined, off Andros. Species codes same as in Figure 5 except MAF = *Montastraea faveolata*, MA = *M. annularis*, MFR = *M. franksi*, AC = *Acropora cervicornis*, DL = *Diploria labyrinthiformis*, MILC = *Millepora complanata*, DC = *Diploria clivosa*.

*cervicornis*, 38% for *A. palmata*). These high values can be largely attributed to the presence of 100% standing dead colonies of *Acropora* which made up 1-24% (average 7%) of the assessed corals in shallow reef crests. Excluding 100% standing dead colonies, *Acropora* had an average old partial-mortality of 27%. Old mortality at fore reef sites ranged from 13% (D15) to 45% (D32) and the frequency of 100% standing



**Figure 7A.** Size frequency distribution of colony diameter for *Acropora palmata* for 1998 with (A) no, (B) partial (1-99%) or (C) 100% (=standing dead) old colony mortality in reef-crest sites off Andros.



**Figure 7B.** Size frequency distribution of colony diameter for *Montastraea annularis* for 1998 with (A) no, (B) partial (1-99%) or (C) 100% (= standing dead) old colony mortality in fore-reef sites off Andros.

dead colonies averaged 7% (Table 2B). *Montastraea faveolata* and *A. palmata* had the highest mean old mortality values (>30%) while *Porites astreoides* had the lowest (<10%) (Fig. 6 for 1998 data). In both *A. palmata* and *M. annularis*, old mortality values increased with colony size up to the mode of the population (120-140 cm-size classes for reef-crest colonies of *A. palmata*, 40-50 cm size classes for fore-reef colonies of *M. annularis*). Smaller corals tended to be either completely alive or completely dead (i.e., to have either no mortality or 100% mortality) whereas areas of old mortality were displayed on the surfaces of many of the larger colonies (Fig. 7A, B). Examination of three spatial scales (within reef, within areas, between areas) for each reef type showed that within-reef-variation explained approximately 80% of the variation in old mortality.

Recent partial-colony mortality (hereafter recent mortality) in the  $\geq 25$  cm stony corals ranged from 0.5-42.5% (Table 2) and was higher in fore reefs (mean 13%) than in reef crests (mean 6%) during both survey years (2-way ANOVA,  $df=580$ ,  $MS=3403$ ,  $F=34.6$ ,  $p<0.0001$ ). Recent mortality in the reef crests was lower during the 1998 surveys (average 4%) than in 1997 (average 7%) (t-test,  $df=270$ ,  $t=5$ ,  $p<0.0001$ ) but was not significantly different between years in the fore reefs (1997 mean=11%, 1998 mean=13%) (t-test,  $df=310$ ,  $t=0.4$ ,  $p=0.7$ ). Recent mortality showed no significant correlation with colony diameter in *A. palmata* and *M. annularis*. Very high levels of recent mortality (>20%) were recorded at several fore reefs in central and southern Andros during the 1998 survey (D10, D13, D24, D25). Significant differences in recent mortality also occurred between different coral species in 1998 (1-way ANOVA,  $df=40$ ,  $MS=3787$ ,  $F=7.7$ ,  $p<0.001$ ). Corals with the highest levels of recent mortality included *Montastraea franksi*, *Colpophyllia natans*, *M. annularis*, and *Diploria labyrinthiformis* (Fig. 6 for 1998 data).

Identifiable causes of recent stony coral mortality in reef crests included predation, diseases (Table 2A), physical damage associated with human-induced trash and boat groundings, and a bleaching event associated with warmer seawater temperatures observed in summer 1998 (see Kramer, this volume). Many colonies of *A. palmata* and *A. cervicornis* had elevated "chimneys" characteristic of repeated predation by the three-spot damselfish, *Stegastes planifrons*. Damselfish gardens were present on approximately 8% of all the surveyed stony corals and were significantly more common in reef crests than in fore reefs (t-test,  $df=584$ ,  $t=3.76$ ,  $p<0.0001$ ). Evidence of predation by parrotfishes (particularly the stoplight parrotfish, *Sparisoma viride*) in the form of excavations or small bite marks was low. The corallivorous snail, *Coralliophila abbreviata*, was observed singularly and in small groups (three to five per group), mainly on colonies of *Acropora*. Small white (<25 cm<sup>2</sup>) lesions were present on approximately 20% of all *A. palmata* surveyed (both years combined) and were attributed to either disease (patchy necrosis = white pox) or predation by *Coralliophila*. Although distinguishing recent mortality due to patchy necrosis from predation was often difficult, on average about 9% of the colonies of *A. palmata* were affected by diseases (all shallow sites combined). High (12-15% in S7, S9, S13, S15, S26) to alarmingly high (~25-30% in S11, S12, S28) incidences of disease were also observed. No large-scale bleaching was observed during 1997 (<2% of colonies overall affected); however, during August 1998, substantial bleaching (>10% of colonies affected) was observed in several shallow sites (e.g., S5, S21). Bleaching was very species-specific with *Millepora complanata*, *Agaricia tenuifolia*, *A. palmata*, and *A. cervicornis* being

most affected. At one site (S24), where 80% of the colonies were discolored (mostly being completely bleached), entire “very recent” colony mortality was observed in several instances.

Recent mortality of stony corals in the fore reefs was mainly attributed to diseases (Table 2B) and, to a lesser degree, to overgrowth by macroalgae. No significant predation or bleaching was noted either year in deeper water (> 5m). In 1997, macroalgal overgrowth by *Microdictyon marinum* was observed to cause limited bleaching and mortality around the edges of colonies, particularly the lobate *M. annularis*. Stony coral diseases were widespread during 1997, but at moderate levels (4% of colonies infected). During 1998, significantly higher levels of disease were recorded (18% infected colonies) (1-way ANOVA,  $df=1$ ,  $MS=1418$ ,  $F=21.7$ ,  $p<0.001$ ), with the most common being black-band disease (BBD) and white plague (WP). During 1998, active BBD and WP infected mainly faviid corals which included in order of decreasing infection *M. franksi*, *M. annularis*, *M. faveolata*, *M. cavernosa*, *D. labyrinthiformis* and *C. natans*. Within reefs, BBD occurred in clumped distributions with simultaneously active infections in colonies that were either touching or closely adjacent. At larger spatial scales, diseases were more prevalent in fore reefs with higher colony densities than in less well-developed reefs with lower colony densities. This pattern may be due in part to the abundance of *M. franksi* and *M. annularis* which were more susceptible. The incidence of WP varied among reefs and it was observed most commonly in the central and southern areas in reefs containing a high density of large corals. Here, sites with <5colonies/10m often had a lower prevalence of disease. Both BBD and WP affected all stony coral size classes. Approximately 6% of all fore-reef stony corals surveyed in 1998 (primarily *M. annularis* and *M. franksi*) had suffered 100% recent mortality. The average colony diameter of the  $\geq 25$  cm corals with 100% recent mortality was 44 cm with most being in the 25-40 cm size class (although corals <25cm were not surveyed in all sites). During 1998, the percentage of recent partial-colony mortality was strongly correlated with the percentage of stony corals showing signs of disease ( $r^2=0.63$ ,  $p <0.001$ ).

#### Algal Communities and *Diadema antillarum*

Algal populations off Andros were distinctly different between the two reef habitats but were remarkably consistent within each depth zone. Reef crests (Table 3A) tended to be dominated by crustose coralline algae (mean relative abundance = 44%) and turf algae and contained relatively few macroalgae (mean relative abundance =15%). Dominant macroalgal species in the reef crests were *Dictyota divaricata*, *Styopodium zonale* and *Turbinaria turbinata* with average canopy heights from 1-3 cm (mean = 1.8 cm) in 1998. In contrast, macroalgae (mean relative abundance = 47%) and crustose corallines (mean relative abundance = 35%) were codominants in fore reefs where turf algae were relatively scarce (Table 3B). The principal macroalgae observed in fore reefs were *Microdictyon marinum*, *Lobophora variegata*, *Dictyota pulchella*, *Dictyota divaricata*, *Sargassum hystrix*, and *Halimeda* spp. Macroalgal heights were significantly higher in fore reefs (averaging 3.3 cm in 1998) than in reef crests for both survey years (2-way ANOVA,  $MS=2.9$ ,  $df=534$ ,  $F=97$ ,  $p<0.0001$ ). When macroalgal abundance and height are considered together as a macroalgal index (Tables 3A, 3B), the index was four times greater in fore reefs (mean=204) than in reef crests (mean=46). *Diadema* was not

recorded in any of the sites during 1997 although a few individuals were counted outside of transects at several sites. *Diadema* densities were slightly higher at examined sites in 1998, averaging 0.4 individuals/100m<sup>2</sup> in both habitats.

## DISCUSSION

### Shallow Reef Crests

The reef crests off Andros presently contain extensive areas of *Acropora palmata*, a large percentage of which have colonies that are partially or even fully alive on their upward facing surfaces. Densities of live and partially live (i.e., <100% total partial-colony mortality) colonies were high (~4 colonies/10m). Evidence of the WB epizootic reported to have decimated populations of *A. palmata* populations in other areas of the Caribbean (Aronson and Precht, 1997, 2001) was minimal. The low-to-moderate levels of old mortality, large colony sizes, and fairly even size-distributions that we observed all suggest modest levels of disturbance to *A. palmata* in the last several decades. Smaller colony sizes were recorded in several sites, particularly near the bights area (e.g., S16-S22) where tidal creeks and channels discharge water having highly variable temperatures and salinities from Great Bahama Bank that probably inhibits *Acropora* growth.

When *A. cervicornis* was present on shallow reefs, it usually occurred as thickets, particularly in the back reef, although most of these colonies were long dead. The distribution and abundance of historic *A. cervicornis* populations are not known for Andros although anecdotal reports suggest it may have been more common in shallow-patch and back-reef habitats.

Low-to-moderate values (1-5%) for recent partial-colony mortality observed in the majority of the reef crest surveys is thought to represent background (chronic) mortality caused by predation, disease, competition, and other biotic interactions. The higher levels of recent mortality recorded during 1997 (e.g., S9, S12-14) are mainly attributed to the way mortality was scored (by including colony sides) rather than to higher levels of disturbance relative to 1998. Levels of recent mortality recorded during 1998 at non-bleached sites (~3.5%) are comparable to levels recorded in other western Atlantic reef crests not experiencing acute disturbances (Kramer, this volume). Most colonies of *A. palmata* had recently dead tissues; however, the small size of most of these lesions suggests either that routine injuries are of limited areal extent and/or tissue regeneration occurs fairly quickly (Meesters et al., 1996). Alternatively, if lesions were caused by disease, tissue loss is slow but may accumulate while the colony remains infected.

National Oceanographic and Atmospheric Administration (NOAA) hotspot maps indicated a 1-2 degree increase in sea surface temperatures beginning in late May, 1998 and extending through August, 1998. Observations during 1998 suggest that bleaching and bleaching-induced mortality off Andros mainly impacted the shallow reefs. Spatially, the extent of bleaching impacts to shallow reefs was variable with reefs immediately south of South Bight being the most affected during the time of our survey. The high value for recent mortality (>15%) observed in one reef crest (S24) surveyed during August, 1998 is attributed to bleaching-induced mortality. Species that suffered mortality

were mainly *Millepora complanata*, *Agaricia tenuifolia* and, to a lesser extent, *A. palmata*. Local reports from the Bahamas during 1998 suggest that region-wide bleaching reached a peak in late August/early September at least two weeks after our surveys ended (Wilkinson, 1998). Therefore, while our surveys indicated only localized impacts, the full effects of the 1998 bleaching event are difficult to ascertain since the immediate damage may have ensued for several weeks to months after our surveys and occurred beyond our survey area. However, the low relative abundance of macroalgae on reef crests, high abundance of living *A. palmata* colonies, and the presence of *A. palmata* recruits (mean=0.27/m<sup>2</sup>) suggest that recovery from low-to-moderate levels of bleaching-induced mortality on reef crests is favored.

### Fore Reefs

Patterns in coral abundance in the Andros fore reefs were evident at both small and large spatial scales. Well-developed fore reefs at intermediate depths (7-12 m) occurred mainly as dense patches of the *M. annularis* species complex often located off islands and along channels where either protection from large swells and/or underlying Pleistocene geomorphology favored reef growth. Grading away from these dense patches were stretches with lower relief, lower coral densities, and smaller coral sizes. Hence, the degree to which fore reefs had developed was largely determined by the abundance of the *M. annularis* complex. In poorly developed reefs, such as N. Mangrove (D23), the dominant coral was *S. siderea* (44%) while the *M. annularis* complex comprised only 17% of the large ( $\geq 25$  cm in diameter) coral population. The size and abundance of the colonies of *Montastraea* in any particular site presumably reflect both the suitability of local conditions (Bak and Meesters, 1998) and rates of coral growth. That fore reefs located in shallower water (7-9 m) had larger colonies of the *M. annularis* complex than those located slightly deeper (10-12m) is probably explicable by the observation that growth rates for *Montastraea* can decrease sharply in this depth range (Hubbard and Scaturo, 1985) and even small absolute increases of water depth (2-3 m) may shift colonies to smaller sizes. The smaller colony sizes found overall in the 1998 surveys can be mainly attributed to the relatively poor environmental conditions in many of the reefs near the bights. In addition, survey depths were slightly shallower in 1997 compared with those in 1998 because the zone of maximum development in the northern and southern areas was shallower than in the central and bights areas.

That we found a strong relationship between levels of old mortality in *M. annularis* and colony size up to the mode of the population (40-50 cm) in Andros (Fig. 7) is similar to the relationship described by Bak and Meesters (1998) in the Netherlands Antilles. In addition, levels of old mortality were strongly influenced by the species composition of the stony corals present in any given reef. Therefore, differences in mean colony size and species composition can explain some of the between-reef variability in levels of old mortality and these factors should be taken into account when comparisons are made among sites, reefs, areas, subregions, and regions.

During 1998, a mass mortality event affected many massive corals in the Andros fore reefs. This widespread disturbance is thought to have begun in June approximately one month after abnormally warm sea surface temperatures enveloped the area (T. Turnbull, personal communication). Local observations of "white coral" were first

reported in early July. One month later we found many massive colonies with either partial or complete recent tissue mortality and signs of disease (primarily WP and BBD) but surprisingly little evidence of bleaching (e.g. <5% colonies displayed discolored tissues). At severely impacted fore reefs, we estimated that up to 50% of the live stony coral cover had been recently lost. We believe severe outbreaks of one or more diseases must have occurred in late June/early July and moved through the entire area at a very rapid pace, perhaps in response to the increased seawater temperatures. [Increased temperatures can trigger or exacerbate the activity of microorganisms (bacteria, cyanobacteria, fungi, protozoans) and possibly viruses that are responsible for outbreaks of disease (e.g., Rutzler et al., 1983).] That the highest mortalities off Andros were observed at depths of 6-12 m in reefs with high coral densities and complex geomorphological structure suggests that diseases might be expected to spread most rapidly under conditions in which colonies (particularly monogeneric coral assemblages) have direct or close contact with one another. The unusually luxuriant growth of the *M. annularis* species complex that is typical of the Andros fore reefs may have made these communities very prone to outbreaks of disease. Recovery from this massive disturbance will take many years, possibly decades.

*Acropora cervicornis* was rarely seen in the deeper sites. Examination of sediments from several fore reefs, however, revealed only small amounts of *A. cervicornis* rubble indicating that it may never have been as common on Andros as in many other areas of the western Atlantic (e.g., Jamaica, Belize, Bonaire).

### Algae and *Diadema*

Historic populations of *Diadema* in the Bahamas, particularly at shallow depths, were very high according to descriptions by Storr (1964) and Newell and Rigby (1951). The low abundances observed in Andros (both shallow and deep sites) during our surveys reinforces local anecdotal accounts that the 1983-84 die-off of *Diadema antillarum* in the western Atlantic (Lessios et al., 1984) also decimated populations on Andros. At present, scarids (parrotfish), acanthurids (surgeonfish), and some pomacentrids (damselfish) are the principal herbivores in Andros' reefs (Kramer, Marks and Turnbull, this volume). As documented in other studies (Hay, 1984; Morrison, 1988), these herbivores are disproportionately distributed across the Andros shelf with higher numbers principally associated with well-developed shallow reef-crest habitats.

Macroalgal indices for the 1998 sites displayed a strong inverse relationship with herbivore biomass (Kramer, Marks and Turnbull, this volume). The relationship is unusually well pronounced, and may also reflect that our surveys took place when, due to seasonal blooms of *Microdictyon marinum*, the macroalgal index reaches a seasonal peak in the late summer (T. Turnbull, personal communication). Within individual habitat types (e.g., reef crest or fore reef), the herbivore-algae relationship was only weakly significant indicating other synergistic factors, such as wave energy and sedimentation, may also influence algal abundance patterns. However, the overall algal-herbivore relationship between depths suggests a strong top-down control on the distribution of macroalgae on Andros. Herbivorous fishes may have compensated for the loss of *Diadema* as an herbivore in reef crests but not in fore-reef sites. High macroalgal canopy heights and lack of grazed surfaces reflect a general lack of herbivory at greater (>6 m)

depths. The impact of excessive algae to fore reef corals is potentially severe, but not well characterized, off Andros. *Microdictyon*, which attaches to the sides of *Montastraea annularis* columns, was observed to grow over the edges of its live tissues and other small corals causing bleaching and, in some cases, mortality. Following the 1998 mass mortality of many massive stony corals, macroalgal abundances are expected to have further increased on fore reefs given the increased availability of substratum space. Macroalgal abundances will likely remain high in Andros fore reefs until levels of grazing by either *Diadema* or herbivorous fishes increase.

Even given the high relative abundance of macroalgae, mean densities of coral recruits in the Andros fore reefs were fairly high (equivalent to  $10.6/m^2$ ) compared with other areas of the wider Caribbean (Bak and Engel, 1979), suggesting that small corals may be able to survive despite overtopping by macroalgae. In addition, average recruit density for the shallow sites (equivalent to 3 juveniles/ $m^2$ ) was higher than at similar reef types in the Florida Reef Tract (maximum  $\sim 2.5$  juveniles of  $<4\text{cm}$  diameter/ $m^2$ ) (Chiappone and Sullivan, 1996). Juvenile species richness was also higher in the shallow reefs off Andros (16 species) than in shallow reefs off the Florida Keys ( $\sim 11$  species) (Chiappone and Sullivan, 1996). The presence and relatively high abundance of recruits of *A. palmata* in reef crests and of the *M. annularis* complex in fore reefs is surprising given that recruits of these species are generally absent or present in very low numbers in settlement studies (e.g., Bak and Engel, 1979; Hughes and Jackson, 1985; Rylaarsdam, 1983). High recruit abundance for these corals on Andros may be attributed to the high densities of adult colonies. The survivorship of the recruits we surveyed was not examined but may play an important role in the stability of these populations. Additional surveys focused on understanding the recruitment and survivorship of *A. palmata* and the *M. annularis* species complex on Andros are warranted.

Despite their relatively remote location, our results indicate that the reefs off Andros Island have been affected by some (e.g., loss of *Diadema antillarum*), but not many, of the regional disturbances documented in other parts of the Caribbean. Impacts of the regional WBD epizootic on *Acropora* and of the massive bleaching events prior to 1998 appear to have been lower off Andros, based on the dense thickets of live *A. palmata* and the large size of many stony corals at the time of our survey. However, a high abundance of macroalgae and the presence of coral diseases in these reefs are suspected in years prior to 1997. In 1998, the massive disease outbreak in fore-reef habitats, particularly in the central and bights areas, resulted in catastrophic tissue loss for many large corals (particularly the *M. annularis* complex). Bleaching in shallow reef crests during 1998 also caused acute localized loss of live stony corals often resulting in complete colony mortality for the most severely impacted reefs in southern Andros (personal observations). The longer-term trajectory with regard to the recovery of these reefs will depend on the frequency of future disturbance events, the status of their *Diadema* populations, and the recruitment of new, and survivorship of existing, stony corals.

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Table 1A. Reef crest site information for AGRRA stony coral and algal surveys off Andros Island, Bahamas (1997 sites are italicized).

Reef crest Site Name	Site Code	Latitude (N° ')	Longitude (W° ')	Survey Date	Depth (m)	Transects (#)	≥25 cm Stony Corals (#/10m) <sup>2</sup>	% Live Stony Coral Cover (mean ± sd) <sup>2</sup>
<i>N. Joulters</i>	<i>S1</i>	<i>25.31322</i>	<i>78.03433</i>	<i>Aug 15 97</i>	<i>2.5</i>	<i>13</i>	<i>13.5</i>	---
<i>Golding</i>	<i>S2</i>	<i>25.22392</i>	<i>78.08557</i>	<i>Aug 18 97</i>	<i>1.5</i>	<i>12</i>	<i>12.0</i>	---
<i>Morgan</i>	<i>S3</i>	<i>25.1601</i>	<i>78.0029</i>	<i>Aug 17 97</i>	<i>1.5</i>	<i>14</i>	<i>10.5</i>	---
<i>Coconut Point</i>	<i>S4</i>	<i>25.12885</i>	<i>77.98303</i>	<i>Aug 17 97</i>	<i>1.5</i>	<i>13</i>	<i>11.0</i>	---
Mahore	S5	25.06367	77.93783	Aug 21 98	1.0	10	9.5	43.0 ± 14.5
S. Staniard 2	S7	24.84493	77.86098	Aug 20 98	0.5	7	9.0	41.0 ± 22.5
<i>S. Staniard 1</i>	<i>S6</i>	<i>24.84217</i>	<i>77.8586</i>	<i>Oct 19 97</i>	<i>3.5</i>	<i>9</i>	<i>9.0</i>	---
<b>North Andros-all<sup>1</sup></b>						<b>78</b>	<b>10.6</b>	<b>42</b>
N. Love Hill	S8	24.77435	77.80772	Aug 18 98	1.0	9	9.5	41.5 ± 14.0
<i>S. Love Hill</i>	<i>S9</i>	<i>24.84425</i>	<i>77.8563</i>	<i>Aug 5 97</i>	<i>1.5</i>	<i>17</i>	<i>7.0</i>	---
China Point	S10	24.75133	77.80767	Aug 18 98	1.0	13	9.0	54.0 ± 31.0
Red Rock	S11	24.72917	77.77017	Aug 7 98	1.0	17	6.5	37.0 ± 15.5
<i>S. Autec</i>	<i>S12</i>	<i>24.69953</i>	<i>77.74343</i>	<i>Aug 6 97</i>	<i>1.5</i>	<i>11</i>	<i>6.5</i>	---
<i>S. Long Rock</i>	<i>S13</i>	<i>24.63067</i>	<i>77.691</i>	<i>Aug 9 97</i>	<i>2.5</i>	<i>15</i>	<i>9.0</i>	---
<i>Mid Long Rock</i>	<i>S14</i>	<i>24.6259</i>	<i>77.69318</i>	<i>Aug 8 97</i>	<i>1.5</i>	<i>24</i>	<i>8.0</i>	---
Sugar Rock	S15	24.5448	77.68372	Aug 10 98	2.0	8	7.0	20.0 ± 11.0
<b>Central Andros-all<sup>1</sup></b>						<b>114</b>	<b>7.8</b>	<b>38</b>
Autec2-South	S16	24.484	77.699	Aug 11 98	1.0	10	7.0	24.0 ± 11.5
N. Bight	S17	24.4395	77.69807	Aug 12 98	2.0	11	9.0	33.5 ± 9.0
Big Wood	S18	24.36703	77.68235	Aug 13 98	1.5	12	9.0	36.0 ± 11.0
Autec 3	S19	24.34315	77.67068	Aug 12 98	1.0	11	10.0	39.5 ± 15.5
Middle Bight	S20	24.3069	77.65638	Aug 13 98	1.5	11	9.5	35.5 ± 10.5
Mangrove C.	S21	24.29167	77.6462	Aug 13 98	1.5	15	8.5	25.0 ± 10.5
Mangrove S	S22	24.25315	77.62917	Aug 13 98	1.0	13	10.0	30.0 ± 13.5
<b>Bights-all<sup>1</sup></b>						<b>83</b>	<b>9.0</b>	<b>28.5</b>
Congo Town	S23	24.3013	77.64788	Aug 15 98	1.0	15	9.0	30.5 ± 14.5
Long Bay Cay	S24	24.09793	77.53703	Aug 16 98	1.0	12	10.5	35.5 ± 11.5
<i>North Rock</i>	<i>S25</i>	<i>23.79092</i>	<i>77.42637</i>	<i>Aug 14 97</i>	<i>2.0</i>	<i>15</i>	<i>9.5</i>	---
<i>North Grassy</i>	<i>S26</i>	<i>23.77822</i>	<i>77.41902</i>	<i>Aug 13 97</i>	<i>2.0</i>	<i>15</i>	<i>9.0</i>	---
<i>Delta</i>	<i>S27</i>	<i>23.70967</i>	<i>77.38237</i>	<i>Aug 13 97</i>	<i>2.0</i>	<i>13</i>	<i>9.5</i>	---
<i>Pigeon</i>	<i>S28</i>	<i>23.6965</i>	<i>77.377</i>	<i>Aug 11 97</i>	<i>1.5</i>	<i>14</i>	<i>8.5</i>	---
<b>South Andros-all<sup>1</sup></b>						<b>84</b>	<b>9.3</b>	<b>33.0</b>

<sup>1</sup>Transects (#) = sum; all other columns = means<sup>2</sup>Based on analysis of the first 10 transects/site.

Table 1B. Fore reef site information for AGRRA stony coral and algal surveys off Andros Island, Bahamas (*1997 sites are italicized*).

Deep Fore Reef Site Name	Site Code	Latitude (N° ')	Longitude (W° ')	Survey Date	Depth (m)	Transects (#)	≥25 cm Stony Corals (#/10m) <sup>2</sup>	% Live Stony Coral Cover (mean ± sd) <sup>2</sup>
<i>N. Joulters</i>	<i>D1</i>	<i>25.3132</i>	<i>78.0856</i>	<i>Aug 16 97</i>	<i>6.5</i>	<i>12</i>	<i>16.0</i>	---
<i>Nichols</i>	<i>D2</i>	<i>25.1438</i>	<i>72.9875</i>	<i>Aug 17 97</i>	<i>9.5</i>	<i>13</i>	<i>13.0</i>	---
<i>Conch</i>	<i>D3</i>	<i>25.1026</i>	<i>77.9639</i>	<i>Oct 17 97</i>	<i>5.5</i>	<i>9</i>	<i>8.5</i>	---
<i>Bucket</i>	<i>D4</i>	<i>24.8793</i>	<i>77.8784</i>	<i>Oct 20 97</i>	<i>10.0</i>	<i>6</i>	<i>11.0</i>	---
<i>N. Staniard</i>	<i>D5</i>	<i>24.8775</i>	<i>77.8791</i>	<i>Oct 18 97</i>	<i>7.5</i>	<i>13</i>	<i>6.5</i>	---
S. Staniard 2	D7	24.50630	77.56437	Aug 20 98	9.0	9	11.0	37.0 ± 13.0
<i>S. Staniard 1</i>	<i>D6</i>	<i>24.8438</i>	<i>77.9406</i>	<i>Oct 19 97</i>	<i>8.0</i>	<i>6</i>	<i>14.5</i>	---
<b>North Andros-all<sup>1</sup></b>						<b>68</b>	<b>11.5</b>	<b>37.0</b>
<i>S. Love Hill</i>	<i>D8</i>	<i>24.7678</i>	<i>77.7987</i>	<i>Aug 6 97</i>	<i>8.5</i>	<i>16</i>	<i>9.0</i>	---
<i>Coffee</i>	<i>D9</i>	<i>24.7452</i>	<i>77.7786</i>	<i>Aug 7 97</i>	<i>11.0</i>	<i>15</i>	<i>7.5</i>	---
West Klein	D10	24.7450	77.7847	Aug 7 98	10.5	10	9.0	24.5 ± 8.0
<i>S. Autec</i>	<i>D11</i>	<i>24.7015</i>	<i>77.7402</i>	<i>Aug 7 97</i>	<i>13.0</i>	<i>8</i>	<i>9.5</i>	---
<i>S. Long Rock</i>	<i>D12</i>	<i>24.6307</i>	<i>77.6910</i>	<i>Aug 9 97</i>	<i>8.0</i>	<i>12</i>	<i>11.5</i>	---
Long Rock	D13	24.6260	77.6910	Aug 8 98	8.7	10	11.0	22.5 ± 8.0
<i>Mid Long Rock</i>	<i>D14</i>	<i>24.37547</i>	<i>77.41587</i>	<i>Aug 9 97</i>	<i>9.5</i>	<i>11</i>	<i>10.0</i>	---
Green Cay	D15	24.5958	77.6933	Aug 9 98	11.0	14	3.0	11.0 ± 4.0
Sugar Rock	D16	24.5402	77.6821	Aug 10 98	11.5	13	6.5	18.0 ± 7.0
<b>Central Andros- all<sup>1</sup></b>						<b>109</b>	<b>8.5</b>	<b>19</b>
Bristol Galley	D17	24.5263	77.6892	Aug 10 98	11.5	12	7.0	17.0 ± 5.0
Autec 2	D18	24.5064	77.6970	Aug 11 98	12.5	15	3.5	9.0 ± 3.0
Autec 2-South	D19	24.4833	77.6965	Aug 11 98	10.5	12	7.5	15.0 ± 5.0
N. Bight	D20	24.4393	77.6962	Aug 12 98	9.5	11	11.5	31.0 ± 12.5
Autec 3	D21	24.3432	77.6707	Aug 14 98	10.5	10	9.0	25.5 ± 10.5
Middle Bight	D22	24.3097	77.6530	Aug 14 98	9.0	10	11.0	28.0 ± 8.0
Mangrove N.	D23	24.3011	77.6478	Aug 14 98	9.5	23	2.0	5.5 ± 3.0
<b>Bights-all<sup>1</sup></b>						<b>93</b>	<b>7.4</b>	<b>18.5</b>
Congo Town	D24	24.3012	77.6479	Aug 15 98	10.5	10	12.0	21.0 ± 15.5
Long Bay Cay	D25	24.0997	77.5334	Aug 16 98	10.5	10	11.0	19.0 ± 9.0
Oasis	D26	23.9476	77.3867	Aug 16 98	6.0	10	12.5	45.5 ± 21.0
High Point Cay	D27	23.4200	77.4600	Aug 17 98	9.0	9	13.5	35.0 ± 7.5
<i>North Rock</i>	<i>D28</i>	<i>23.7965</i>	<i>77.4222</i>	<i>Aug 13 97</i>	<i>7.5</i>	<i>11</i>	<i>11.5</i>	---
<i>North Grassy</i>	<i>D29</i>	<i>23.7803</i>	<i>77.4168</i>	<i>Aug 13 97</i>	<i>9.5</i>	<i>9</i>	<i>10.0</i>	---
<i>South Grassy</i>	<i>D30</i>	<i>23.7285</i>	<i>77.3974</i>	<i>Aug 12 97</i>	<i>7.5</i>	<i>10</i>	<i>12.5</i>	---
<i>Delta</i>	<i>D31</i>	<i>23.7097</i>	<i>77.3810</i>	<i>Aug 13 97</i>	<i>8.5</i>	<i>12</i>	<i>7.5</i>	---
<i>Pigeon</i>	<i>D32</i>	<i>23.6945</i>	<i>77.3742</i>	<i>Aug 11 97</i>	<i>9.0</i>	<i>12</i>	<i>8.0</i>	---
<i>Saddleback</i>	<i>D33</i>	<i>23.6767</i>	<i>77.3703</i>	<i>Aug 11 97</i>	<i>8.5</i>	<i>10</i>	<i>12.5</i>	---
<b>South Andros-all<sup>1</sup></b>						<b>103</b>	<b>11.2</b>	<b>30.0</b>

<sup>1</sup>Transects (#) = sum; all other columns = means<sup>2</sup>Based on analysis of the first 10 transects/site.

Table 2A. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 25$  cm diameter) in first 10 transects/site, by reef-crest sites off Andros (1997 sites are italicized).

Reef Crest Site Name	Site Code	Year	Stony Corals		Partial-Colony Mortality (%)			Stony Corals (%)		
			(#)	Diameter (cm)	Recent <sup>2</sup>	Old	Total	Standing dead	Bleached	Diseased
<i>N. Joulters</i>	S1	1997	177	114.5 $\pm$ 5.5	6.5 $\pm$ 0.5	21.5 $\pm$ 2.0	28.0 $\pm$ 2.0	3.5	1.5	8
<i>Golding</i>	S2	1997	144	117.5 $\pm$ 6.0	5.5 $\pm$ 0.5	33.0 $\pm$ 3.0	38.5 $\pm$ 3.0	10.5	2	5
<i>Morgan</i>	S3	1997	144	114.5 $\pm$ 7.0	7.0 $\pm$ 1.5	21.5 $\pm$ 2.0	28.0 $\pm$ 2.5	5.4	2	7
<i>Coconut Point</i>	S4	1997	138	134.0 $\pm$ 6.0	6.5 $\pm$ 1.0	37.0 $\pm$ 3.5	42.0 $\pm$ 3.0	20.5	0	3
<i>Mahore</i>	S5	1998	94	123.0 $\pm$ 9.5	2.0 $\pm$ 0.5	12.0 $\pm$ 2.0	13.5 $\pm$ 2.0	1	14	6.5
<i>S. Staniard 2</i>	S7	1998	64	135.5 $\pm$ 12.5	0.5 $\pm$ <0.5	24.0 $\pm$ 4.0	24.5 $\pm$ 4.0	8	0	15
<i>S. Staniard 1</i>	S6	1997	80	191.0 $\pm$ 12.5	4.5 $\pm$ 1.0	43.5 $\pm$ 4.5	47.0 $\pm$ 4.5	24.5	11	0
<b>North Andros-all<sup>1</sup></b>			<b>841</b>	<b>133 <math>\pm</math> 27</b>	<b>4.5 <math>\pm</math> 2.5</b>	<b>27.5 <math>\pm</math> 11.0</b>	<b>31.5 <math>\pm</math> 11.5</b>	<b>10.5 <math>\pm</math> 8.8</b>	<b>4.5 <math>\pm</math> 5.7</b>	<b>6.5 <math>\pm</math> 4.7</b>
<i>N. Love Hill</i>	S8	1998	85	101.5 $\pm$ 10.0	3.5 $\pm$ 1.5	24.0 $\pm$ 2.5	27.0 $\pm$ 3.0	0	3.5	1
<i>S. Love Hill</i>	S9	1997	116	116.0 $\pm$ 6.5	11.0 $\pm$ 1.5	38.0 $\pm$ 3.0	47.5 $\pm$ 3.0	8.5	6	12
<i>China Point</i>	S10	1998	109	105.5 $\pm$ 8.0	2.5 $\pm$ 0.5	25.0 $\pm$ 3.0	27.0 $\pm$ 3.0	3.5	10	4.5
<i>Red Rock</i>	S11	1998	108	97.0 $\pm$ 5.5	2.5 $\pm$ 0.5	32.0 $\pm$ 2.5	35.0 $\pm$ 3.0	1	7.5	25
<i>S. Autec</i>	S12	1997	71	128.5 $\pm$ 9.0	9.5 $\pm$ 1.5	41.5 $\pm$ 4.5	50.0 $\pm$ 4.0	14	1.5	25.5
<i>S. Long Rock</i>	S13	1997	130	126.0 $\pm$ 9.0	8.5 $\pm$ 1.0	37.5 $\pm$ 2.5	45.5 $\pm$ 3.0	8.5	1	14
<i>Mid Long Rock</i>	S14	1997	195	108.0 $\pm$ 4.5	8.5 $\pm$ 0.5	36.5 $\pm$ 2.0	44.0 $\pm$ 2.5	7.	3.5	9.5
<i>Sugar Rock</i>	S15	1998	57	117.5 $\pm$ 9.0	4.0 $\pm$ 1.0	27.5 $\pm$ 4.5	31.5 $\pm$ 4.0	9.	0	14
<b>Central Andros-all<sup>1</sup></b>			<b>871</b>	<b>112.5 <math>\pm</math> 11.5</b>	<b>6.0 <math>\pm</math> 3.5</b>	<b>33.0 <math>\pm</math> 6.5</b>	<b>38.5 <math>\pm</math> 9.5</b>	<b>6.5 <math>\pm</math> 4.6</b>	<b>4.1 <math>\pm</math> 3.5</b>	<b>13.2 <math>\pm</math> 8.7</b>
<i>Autec 2-South</i>	S16	1998	69	96.0 $\pm$ 6.5	1.0 $\pm$ 0.5	39.0 $\pm$ 4.5	39.5 $\pm$ 4.5	16	4.5	10
<i>N. Bight</i>	S17	1998	99	117.5 $\pm$ 6.0	2.0 $\pm$ 0.5	31.5 $\pm$ 3.5	33.0 $\pm$ 3.5	13	1	8
<i>Big Wood</i>	S18	1998	109	89.0 $\pm$ 5.5	4.5 $\pm$ 0.5	23.0 $\pm$ 3.0	27.0 $\pm$ 3.0	9	7.5	2
<i>Autec 3</i>	S19	1998	110	85.0 $\pm$ 5.0	3.5 $\pm$ 0.5	25.0 $\pm$ 3.0	28.5 $\pm$ 3.0	7.5	8	3.5
<i>Middle Bight</i>	S20	1998	103	88.0 $\pm$ 6.0	5.0 $\pm$ 1.0	35.0 $\pm$ 3.5	39.5 $\pm$ 3.5	12.5	5	1
<i>Mangrove C.</i>	S21	1998	124	84.0 $\pm$ 5.0	3.5 $\pm$ 1.0	38.5 $\pm$ 3.5	41.0 $\pm$ 3.5	17.5	13.5	4
<i>Mangrove S.</i>	S22	1998	130	73.0 $\pm$ 4.5	2.5 $\pm$ 0.5	25.0 $\pm$ 3.0	27.5 $\pm$ 3.0	9	3	6
<b>Bights-all<sup>1</sup></b>			<b>744</b>	<b>90.5 <math>\pm</math> 14.0</b>	<b>3.0 <math>\pm</math> 1.5</b>	<b>31.0 <math>\pm</math> 6.5</b>	<b>33.5 <math>\pm</math> 6.0</b>	<b>12.0 <math>\pm</math> 3.7</b>	<b>6.1 <math>\pm</math> 4.1</b>	<b>4.9 <math>\pm</math> 3.2</b>
<i>Congo Town</i>	S23	1998	138	93.5 $\pm$ 4.5	3.0 $\pm$ 0.5	30.0 $\pm$ 3.0	33.0 $\pm$ 3.0	13	8.5	3.5
<i>Long Bay Cay</i>	S24	1998	124	88.0 $\pm$ 5.0	15.0 $\pm$ 2.5	28.0 $\pm$ 3.0	41.0 $\pm$ 3.0	3	80	1.5
<i>North Rock</i>	S25	1997	146	119.0 $\pm$ 7.0	7.5 $\pm$ 1.0	52.0 $\pm$ 3.0	58.5 $\pm$ 2.5	18	0.5	7
<i>North Grassy</i>	S26	1997	132	109.5 $\pm$ 7.0	6.0 $\pm$ 1.0	47.5 $\pm$ 3.0	53.0 $\pm$ 3.0	12	2.5	13
<i>Delta</i>	S27	1997	123	119.0 $\pm$ 6.0	3.0 $\pm$ 0.5	40.0 $\pm$ 3.0	43.0 $\pm$ 3.0	4	1	4
<i>Pigeon</i>	S28	1997	117	131.5 $\pm$ 8.5	7.5 $\pm$ 1.0	41.0 $\pm$ 3.0	48.5 $\pm$ 3.0	7	0	31
<b>South Andros-all<sup>1</sup></b>			<b>780</b>	<b>110.0 <math>\pm</math> 16.5</b>	<b>7.0 <math>\pm</math> 4.5</b>	<b>40.0 <math>\pm</math> 9.5</b>	<b>46.0 <math>\pm</math> 9.0</b>	<b>9.5 <math>\pm</math> 5.8</b>	<b>15.4 <math>\pm</math> 31.8</b>	<b>10.0 <math>\pm</math> 11.0</b>

<sup>1</sup>Stony corals (#) = sum; all other columns = means  $\pm$  standard deviation.

<sup>2</sup>1997 values are inflated relative to 1998 due to changes in assessment methodology.

Table 2B. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 25$  cm diameter), in first 10 transects/site, by fore-reef sites off Andros (1997 sites are italicized).

Fore-reef Site name	Site Code	Year	Stony corals		Partial-colony mortality (%)			Stony corals (%)		
			(#)	Diameter (cm)	Recent <sup>2</sup>	Old	Total	Standing dead	Bleached	Diseased
<i>N. Joulters</i>	D1	1997	166	65.0 $\pm$ 2.5	11.0 $\pm$ 1.5	21.0 $\pm$ 2.0	32.0 $\pm$ 2.0	2.5	0.5	1
<i>Nichols</i>	D2	1997	168	72.0 $\pm$ 4.0	11.0 $\pm$ 1.5	33.0 $\pm$ 2.5	43.0 $\pm$ 2.5	7	3	2
<i>Conch</i>	D3	1997	77	57.5 $\pm$ 3.5	6.0 $\pm$ 1.5	19.0 $\pm$ 3.0	25.0 $\pm$ 3.0	1.5	2.5	1.5
<i>Bucket</i>	D4	1997	58	70.5 $\pm$ 7.0	2.0 $\pm$ 0.5	30.0 $\pm$ 3.5	32.0 $\pm$ 3.5	0	14	1.5
<i>N. Staniard</i>	D5	1997	77	60.5 $\pm$ 5.0	7.0 $\pm$ 1.5	39.5 $\pm$ 4.0	46.0 $\pm$ 4.0	6.5	4	1.5
S. Staniard 2	D6	1998	99	54.0 $\pm$ 4.0	4.0 $\pm$ 1.5	16.5 $\pm$ 2.0	20.5 $\pm$ 2.5	2	8	7
<i>S. Staniard 1</i>	D7	1997	87	59.5 $\pm$ 3.5	8.0 $\pm$ 1.5	33.5 $\pm$ 3.5	40.5 $\pm$ 3.5	5.5	10.5	1
<b>North Andros-all<sup>1</sup></b>			<b>732</b>	<b>62.5 <math>\pm</math> 6.5</b>	<b>7.0 <math>\pm</math> 3.5</b>	<b>27.5 <math>\pm</math> 8.5</b>	<b>34.0 <math>\pm</math> 9.5</b>	<b>3.5 <math>\pm</math> 2.7</b>	<b>6.0 <math>\pm</math> 4.9</b>	<b>2.2 <math>\pm</math> 2.1</b>
<i>S. Love Hill</i>	D8	1997	98	69.0 $\pm$ 3.5	12.0 $\pm$ 2.0	37.5 $\pm$ 3.5	48.5 $\pm$ 3.5	2	1	12
<i>Coffee</i>	D9	1997	113	55.5 $\pm$ 3.0	16.5 $\pm$ 2.0	31.0 $\pm$ 3.0	46.5 $\pm$ 3.0	4.5	2.5	9
West Klein	D10	1998	89	47.5 $\pm$ 2.0	24.5 $\pm$ 3.5	16.0 $\pm$ 3.0	39.0 $\pm$ 3.5	3.5	13.5	42.5
<i>S. Autec</i>	D11	1997	74	66.5 $\pm$ 4.0	19.0 $\pm$ 2.5	30.0 $\pm$ 3.0	48.0 $\pm$ 3.5	2.5	4	2.5
<i>S. sLong Rock</i>	D12	1997	135	59.5 $\pm$ 2.5	14.0 $\pm$ 1.5	25.0 $\pm$ 2.0	39.0 $\pm$ 2.5	0	5	8
Long Rock	D13	1998	110	58.5 $\pm$ 3.0	39.0 $\pm$ 4.0	27.0 $\pm$ 3.5	58.5 $\pm$ 3.5	7.5	11	27.5
<i>Mid Long Rock</i>	D14	1997	110	55.0 $\pm$ 3.0	13.5 $\pm$ 1.5	22.5 $\pm$ 2.0	36.0 $\pm$ 2.5	1	5.5	7.5
Green Cay	D15	1998	37	41.0 $\pm$ 3.5	15.5 $\pm$ 5.0	13.0 $\pm$ 3.5	28.0 $\pm$ 5.5	2.5	2.5	13.5
Sugar Rock	D16	1998	78	34.0 $\pm$ 1.0	8.5 $\pm$ 2.5	20.5 $\pm$ 3.0	28.5 $\pm$ 3.5	1.5	5	16.5
<b>Central Andros-all<sup>1</sup></b>			<b>844</b>	<b>54.0 <math>\pm</math> 11.5</b>	<b>18.0 <math>\pm</math> 9.0</b>	<b>24.5 <math>\pm</math> 7.5</b>	<b>41.5 <math>\pm</math> 10.0</b>	<b>2.7 <math>\pm</math> 2.2</b>	<b>5.5 <math>\pm</math> 4.1</b>	<b>15.4 <math>\pm</math> 12.4</b>
Bristol Galley	D17	1998	74	38.5 $\pm$ 2.0	15.0 $\pm$ 3.5	24.0 $\pm$ 4.0	37.5 $\pm$ 4.5	7	4	23
Autec 2	D18	1998	46	39.0 $\pm$ 3.0	8.5 $\pm$ 4.0	22.5 $\pm$ 5.5	29.0 $\pm$ 5.5	11	11	6.5
Autec 2-South	D19	1998	81	46.5 $\pm$ 3.5	11.0 $\pm$ 3.0	29.0 $\pm$ 4.0	38.0 $\pm$ 4.0	8.5	0	11
N. Bight	D20	1998	115	54.5 $\pm$ 3.5	12.5 $\pm$ 2.5	20.0 $\pm$ 2.5	30.5 $\pm$ 3.0	5	2.5	15.5
Autec 3	D21	1998	89	39.5 $\pm$ 2.0	14.5 $\pm$ 3.0	20.5 $\pm$ 3.0	34.0 $\pm$ 3.5	2	8	21.5
Middle Bight	D22	1998	106	51.5 $\pm$ 3.5	16.0 $\pm$ 3.0	22.5 $\pm$ 3.0	37.5 $\pm$ 3.5	4.	2	22.5
Mangrove N.	D23	1998	47	34.0 $\pm$ 1.5	7.0 $\pm$ 2.0	17.5 $\pm$ 3.5	24.5 $\pm$ 3.5	0	6.5	6.5
<b>Bights-all<sup>1</sup></b>			<b>558</b>	<b>43.5 <math>\pm</math> 7.5</b>	<b>12.0 <math>\pm</math> 3.5</b>	<b>22.5 <math>\pm</math> 3.5</b>	<b>33.0 <math>\pm</math> 5.0</b>	<b>5.4 <math>\pm</math> 3.8</b>	<b>4.8 <math>\pm</math> 3.8</b>	<b>15.2 <math>\pm</math> 7.3</b>
Congo Town	D24	1998	122	53.5 $\pm$ 2.5	42.5 $\pm$ 3.5	25.5 $\pm$ 3.0	62.0 $\pm$ 3.0	2.5	2.5	32
Long Bay Cay	D25	1998	110	43.0 $\pm$ 2.0	20.5 $\pm$ 3.5	23.0 $\pm$ 3.0	40.0 $\pm$ 3.5	5.5	4.5	25.5
Oasis	D26	1998	127	72.5 $\pm$ 5.0	2.0 $\pm$ 0.5	26.0 $\pm$ 2.5	28.0 $\pm$ 2.5	2.5	1.5	3
High Point Cay	D27	1998	114	45.5 $\pm$ 2.0	5.0 $\pm$ 1.5	20.5 $\pm$ 2.5	25.5 $\pm$ 2.5	2	2.5	8
<i>North Rock</i>	D28	1997	124	63.5 $\pm$ 3.5	9.0 $\pm$ 1.5	29.5 $\pm$ 2.5	38.0 $\pm$ 2.5	1.5	0	1.5
<i>North Grassy</i>	D29	1997	92	54.0 $\pm$ 4.0	8.5 $\pm$ 1.5	33.5 $\pm$ 3.0	42.0 $\pm$ 3.0	1	2	2
<i>South Grassy</i>	D30	1997	125	69.0 $\pm$ 3.5	10.0 $\pm$ 1.0	39.0 $\pm$ 2.5	49.0 $\pm$ 2.5	5	1	4
<i>Delta</i>	D31	1997	90	53.0 $\pm$ 2.5	11.0 $\pm$ 2.5	31.5 $\pm$ 3.0	42.0 $\pm$ 3.5	3.5	1	3.5
<i>Pigeon</i>	D32	1997	97	68.5 $\pm$ 4.5	13.5 $\pm$ 2.5	45.0 $\pm$ 4.0	56.0 $\pm$ 3.5	12.5	1	6
<i>Saddleback</i>	D33	1997	123	63.5 $\pm$ 3.0	11.0 $\pm$ 1.5	29.5 $\pm$ 2.5	40.5 $\pm$ 2.5	2.5	4	12
<b>South Andros-all<sup>1</sup></b>			<b>1124</b>	<b>58.5 <math>\pm</math> 10.0</b>	<b>13.5 <math>\pm</math> 11.5</b>	<b>30.5 <math>\pm</math> 7.5</b>	<b>42.5 <math>\pm</math> 11.0</b>	<b>3.8 <math>\pm</math> 3.3</b>	<b>2 <math>\pm</math> 1.4</b>	<b>9.8 <math>\pm</math> 10.6</b>

<sup>1</sup>Stony corals (#) = sum; all other columns = means  $\pm$  standard deviation.

<sup>2</sup>1997 values are inflated relative to 1998 due to changes in assessment methodology.

Table 3A. Algal characteristics, and density of stony coral recruits and of *Diadema antillarum* (mean  $\pm$  standard deviation), by reef-crest sites off Andros (1997 sites are italicized).

Reef Crest Site Name	Site Code	Year	Quadrats (#)	Relative Abundance (%)			Macroalgal		Recruits (#/0.0625 m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )
				Macroalgae	Turf algae	Crustose coralline algae	Height <sup>2</sup>	Index <sup>3</sup>		
<i>N. Joulters</i>	S1	1997	53	6.5 $\pm$ 10.0	43.0 $\pm$ 11.0	50.5 $\pm$ 6.0	---	---	0.2 $\pm$ 0.4	0
<i>Golding</i>	S2	1997	40	7.5 $\pm$ 12.0	39.5 $\pm$ 13.5	53.0 $\pm$ 11.5	---	---	0.0 $\pm$ 0.2	0
<i>Morgan</i>	S3	1997	44	8.0 $\pm$ 8.5	51.5 $\pm$ 16.5	40.5 $\pm$ 18.0	---	---	0.3 $\pm$ 0.5	0
<i>Coconut Point</i>	S4	1997	41	8.5 $\pm$ 15.0	44.0 $\pm$ 16.5	47.5 $\pm$ 14.5	---	---	0.1 $\pm$ 0.3	0
Mahore	S5	1998	45	12.0 $\pm$ 16.0	49.5 $\pm$ 18.5	38.5 $\pm$ 17.0	3	82	0.1 $\pm$ 0.4	1
S. Staniard 2	S7	1998	34	13.0 $\pm$ 7.5	64.5 $\pm$ 22.0	22.5 $\pm$ 19.5	2	41	0.1 $\pm$ 0.3	0
<i>S. Staniard 1</i>	S6	1997	36	3.5 $\pm$ 6.0	49.5 $\pm$ 16.0	47.0 $\pm$ 15.5	---	---	0.2 $\pm$ 0.3	0
<b>North Andros-all<sup>1</sup></b>			<b>293</b>	<b>8.0 <math>\pm</math> 3.0</b>	<b>49 <math>\pm</math> 8.0</b>	<b>43.0 <math>\pm</math> 10.5</b>	<b>2.5 <math>\pm</math> 0.5</b>	<b>61.5 <math>\pm</math> 29.0</b>	<b>0.2 <math>\pm</math> 0.1</b>	<b>0.1 <math>\pm</math> 0.4</b>
N. Love Hill	S8	1998	40	17.0 $\pm$ 23.0	39.5 $\pm$ 25.0	43.5 $\pm$ 27.0	2.5	79	0.4 $\pm$ 0.8	0
<i>S. Love Hill</i>	S9	1997	60	18.5 $\pm$ 15.5	41.0 $\pm$ 27.5	40.5 $\pm$ 29.5	---	---	0.3 $\pm$ 0.7	0
China Point	S10	1998	44	10.5 $\pm$ 9.5	37.0 $\pm$ 25.5	52.5 $\pm$ 26.5	2	28	0.1 $\pm$ 0.5	0
Red Rock	S11	1998	18	13.5 $\pm$ 21.5	57.0 $\pm$ 26.0	29.5 $\pm$ 27.0	1	15	0.1 $\pm$ 0.2	0
<i>S. Autec</i>	S12	1997	42	26.0 $\pm$ 21.0	38.0 $\pm$ 22.5	35.5 $\pm$ 24.0	---	---	0.3 $\pm$ 0.5	0.9
<i>S. Long Rock</i>	S13	1997	46	14.5 $\pm$ 14.0	40.5 $\pm$ 14.5	45.0 $\pm$ 19.0	---	---	0.2 $\pm$ 0.4	0
<i>Mid Long Rock</i>	S14	1997	92	9.5 $\pm$ 10.5	50.5 $\pm$ 24.5	39.5 $\pm$ 26.0	---	---	0.4 $\pm$ 0.6	0
Sugar Rock	S15	1998	37	8.0 $\pm$ 8.5	50.0 $\pm$ 20.5	42.0 $\pm$ 22.0	1.5	23	0.2 $\pm$ 0.8	0
<b>Central Andros-all<sup>1</sup></b>			<b>379</b>	<b>14.0 <math>\pm</math> 5.5</b>	<b>44.0 <math>\pm</math> 7.0</b>	<b>42 <math>\pm</math> 8.0</b>	<b>1.5 <math>\pm</math> 1.0</b>	<b>36.5 <math>\pm</math> 29.0</b>	<b>0.3 <math>\pm</math> 0.1</b>	<b>0.1 <math>\pm</math> 0.3</b>
Autec 2-South	S16	1998	46	10.5 $\pm$ 11.0	44.5 $\pm$ 23.0	44.5 $\pm$ 25.0	1	24	0.0 $\pm$ 0.2	0
N. Bight	S17	1998	55	13.5 $\pm$ 20.0	45.0 $\pm$ 21.5	39.0 $\pm$ 18.0	1.5	26	0.1 $\pm$ 0.6	0
Big Wood	S18	1998	54	26.0 $\pm$ 20.5	37.0 $\pm$ 23.5	36.5 $\pm$ 26.0	2.5	99	0.1 $\pm$ 0.4	4
Autec 3	S19	1998	54	9.0 $\pm$ 12.0	46.5 $\pm$ 25.0	44.5 $\pm$ 28.5	1.5	22	0.1 $\pm$ 0.7	0
Middle Bight	S20	1998	54	13.5 $\pm$ 20.0	40.0 $\pm$ 21.0	46.0 $\pm$ 22.5	1.5	39	0.1 $\pm$ 0.3	2.5
Mangrove C.	S21	1998	65	10.5 $\pm$ 13.5	40.5 $\pm$ 21.0	48.5 $\pm$ 22.0	1.5	47	0.2 $\pm$ 0.5	0
Mangrove S	S22	1998	64	20.5 $\pm$ 17.5	42.0 $\pm$ 18.0	37.5 $\pm$ 18.0	2.5	88	0.2 $\pm$ 0.5	0
<b>Bights-all<sup>1</sup></b>			<b>392</b>	<b>15.0 <math>\pm</math> 6.0</b>	<b>42.0 <math>\pm</math> 3.5</b>	<b>42.5 <math>\pm</math> 4.5</b>	<b>1.5 <math>\pm</math> 0.5</b>	<b>49.0 <math>\pm</math> 31.5</b>	<b>0.1 <math>\pm</math> 0.1</b>	<b>1 <math>\pm</math> 1.5</b>
Congo Town	S23	1998	69	11.5 $\pm$ 15.5	13.5 $\pm$ 19.0	44.0 $\pm$ 19.0	1.5	26	0.1 $\pm$ 0.4	2.5
Long Bay Cay	S24	1998	54	11.0 $\pm$ 14.5	14.0 $\pm$ 19.0	41.0 $\pm$ 19.5	2	43	0.1 $\pm$ 0.3	0
<i>North Rock</i>	S25	1997	50	14.5 $\pm$ 19.0	29.5 $\pm$ 21.0	56.0 $\pm$ 16.0	---	---	0.1 $\pm$ 0.4	0
<i>North Grassy</i>	S26	1997	51	10.5 $\pm$ 11.5	43.0 $\pm$ 19.5	47.0 $\pm$ 17.5	---	---	0.4 $\pm$ 0.7	0
<i>Delta</i>	S27	1997	44	11.5 $\pm$ 12.0	30.5 $\pm$ 15.0	58.0 $\pm$ 16.5	---	---	0.3 $\pm$ 0.5	0
<i>Pigeon</i>	S28	1997	44	18.0 $\pm$ 15.0	40.5 $\pm$ 15.0	41.5 $\pm$ 18.0	---	---	0.2 $\pm$ 0.4	0
<b>South Andros-all<sup>1</sup></b>			<b>312</b>	<b>12.5 <math>\pm</math> 3.0</b>	<b>28.5 <math>\pm</math> 12.5</b>	<b>48.0 <math>\pm</math> 7.5</b>	<b>1.5 <math>\pm</math> 0.5</b>	<b>34.5 <math>\pm</math> 12.0</b>	<b>0.2 <math>\pm</math> 0.1</b>	<b>0.5 <math>\pm</math> 1.0</b>

<sup>1</sup>Quadrats (#) = sum; all other columns = means  $\pm$  standard deviation.

<sup>2</sup>Heights measured as maximum canopy heights in 1997 are not included.

<sup>3</sup>Macroalgal index = relative abundance of macroalgae  $\times$  macroalgal height.

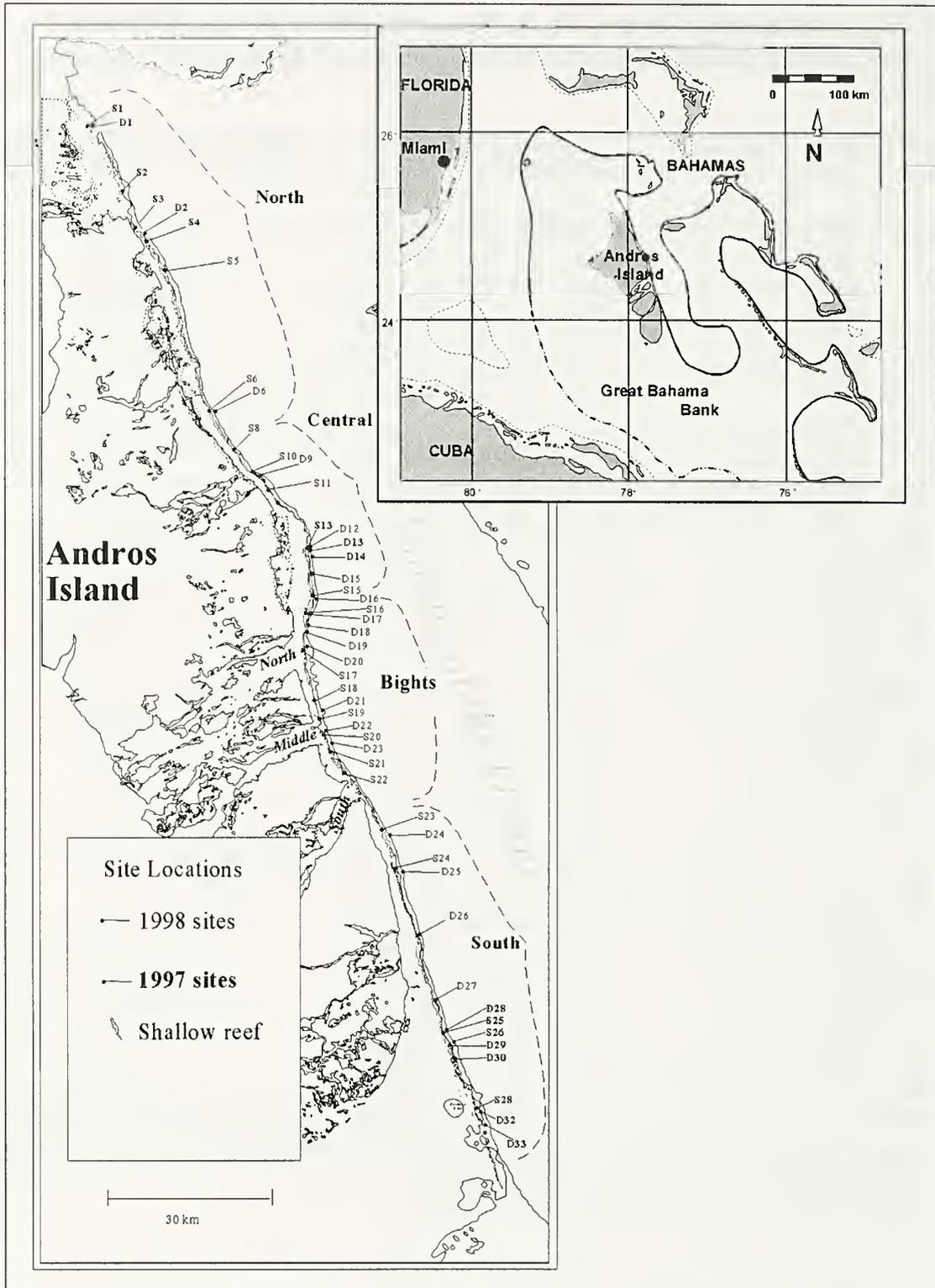
Table 3B. Algal characteristics, and density of stony coral recruits and of *Diadema antillarum* (mean  $\pm$  standard deviation), by fore-reef sites off Andros (1997 sites are italicized).

Fore Reef Site Name	Site Code	Year	Quadrat (#)	Relative Abundance (%)			Macroalgal		Recruits (#/0.625 m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )
				Macroalgae	Turf algae	Crustose coralline algae	Height <sub>2</sub>	Index <sup>3</sup>		
<i>N. Joulters</i>	D1	1997	35	49.0 $\pm$ 18.0	10.0 $\pm$ 13.5	41.0 $\pm$ 18.5	---	---	0.3 $\pm$ 0.6	0
<i>Nichols</i>	D2	1997	42	35.5 $\pm$ 15.0	17.0 $\pm$ 13.0	47.5 $\pm$ 14.0	---	---	0.5 $\pm$ 0.6	0
<i>Conch</i>	D3	1997	30	60.0 $\pm$ 32.0	26.5 $\pm$ 26.5	14.0 $\pm$ 14.0	---	---	0.3 $\pm$ 0.6	0
<i>Bucket</i>	D4	1997	24	25.5 $\pm$ 8.5	45.0 $\pm$ 22.0	29.5 $\pm$ 18.5	---	---	0.7 $\pm$ 0.8	0
<i>N. Staniard</i>	D5	1997	21	33.5 $\pm$ 18.5	44.0 $\pm$ 26.0	22.5 $\pm$ 8.0	---	---	0.5 $\pm$ 0.6	0
S. Staniard 2	D6	1998	45	35.0 $\pm$ 14.0	27.5 $\pm$ 18.5	37.5 $\pm$ 19.0	3.5	162	0.4 $\pm$ 1.1	1
<i>S. Staniard 1</i>	D7	1997	24	36 $\pm$ 10.0	46.0 $\pm$ 18.0	22.5 $\pm$ 18.0	---	---	0.8 $\pm$ 1.0	0
<b>North Andros-all<sup>1</sup></b>			<b>221</b>	<b>39.0 <math>\pm</math> 11.5</b>	<b>31.0 <math>\pm</math> 14.0</b>	<b>30.5 <math>\pm</math> 12.0</b>	<b>3.5</b>	<b>162</b>	<b>0.5 <math>\pm</math> 0.2</b>	<b>0.2 <math>\pm</math> 0.4</b>
<i>S. Love Hill</i>	D8	1997	37	61.0 $\pm$ 19.5	25.0 $\pm$ 14.5	14.5 $\pm$ 12.0	---	---	0.9 $\pm$ 1.2	0
<i>Coffee</i>	D9	1997	51	56.5 $\pm$ 17.0	13.5 $\pm$ 14.5	30.0 $\pm$ 14.0	---	---	0.7 $\pm$ 0.9	0
West Klein	D10	1998	45	48.0 $\pm$ 19.5	28.0 $\pm$ 18.0	24.0 $\pm$ 11.5	3.5	177	0.5 $\pm$ 0.9	0
<i>S. Autec</i>	D11	1997	33	74.0 $\pm$ 17.5	6.5 $\pm$ 9.5	19.0 $\pm$ 16.0	---	---	0.7 $\pm$ 1.0	0
<i>S. Long Rock</i>	D12	1997	38	62.5 $\pm$ 20.5	10.5 $\pm$ 11.5	27.0 $\pm$ 16.0	---	---	0.9 $\pm$ 0.9	0
Long Rock	D13	1998	46	48.0 $\pm$ 20.0	30.0 $\pm$ 19.5	22.0 $\pm$ 13.0	3.0	145	0.1 $\pm$ 0.4	0
<i>Mid Long Rock</i>	D14	1997	33	59.0 $\pm$ 15.0	9.5 $\pm$ 12.0	31.5 $\pm$ 10.5	---	---	0.7 $\pm$ 0.8	0
Green Cay	D15	1998	62	54.5 $\pm$ 20.0	15.5 $\pm$ 19.0	30.0 $\pm$ 14.0	3	212	0.7 $\pm$ 0.8	0
Sugar Rock	D16	1998	56	41.5 $\pm$ 12.5	21.0 $\pm$ 16.5	37.5 $\pm$ 15.0	3.5	215	0.9 $\pm$ 1.0	0
<b>Central Andros all<sup>1</sup></b>			<b>401</b>	<b>56.0 <math>\pm</math> 9.5</b>	<b>18.0 <math>\pm</math> 9.0</b>	<b>26.0 <math>\pm</math> 7.0</b>	<b>3.2 <math>\pm</math> 0.3</b>	<b>187 <math>\pm</math> 33</b>	<b>0.7 <math>\pm</math> 0.2</b>	<b>0</b>
Bristol Galley	D17	1998	48	45.5 $\pm$ 21.0	18.0 $\pm$ 20.5	36.5 $\pm$ 15.0	4.0	238	0.8 $\pm$ 1.1	0
Autec 2	D18	1998	62	51.5 $\pm$ 13.0	11.0 $\pm$ 9.0	37.5 $\pm$ 10.5	3.5	281	0.7 $\pm$ 1.1	2
Autec 2-South	D19	1998	44	40.0 $\pm$ 15.5	24.0 $\pm$ 24.0	36.0 $\pm$ 18.5	3.0	178	0.4 $\pm$ 0.7	0
N. Bight	D20	1998	47	45.0 $\pm$ 15.5	21.0 $\pm$ 22.5	33.5 $\pm$ 16.0	3.0	202	0.8 $\pm$ 1.2	0
Autec 3	D21	1998	49	44.0 $\pm$ 16.0	14.0 $\pm$ 17.5	42.0 $\pm$ 18.0	3.0	185	0.6 $\pm$ 0.7	0
Middle Bight	D22	1998	48	42.0 $\pm$ 13.0	14.0 $\pm$ 15.5	44.0 $\pm$ 14.0	3.7	255	0.6 $\pm$ 1.0	0
Mangrove N.	D23	1998	70	50.5 $\pm$ 19.0	15.5 $\pm$ 17.0	34.0 $\pm$ 22.0	3.0	238	0.2 $\pm$ 0.6	0
<b>Bights-all<sup>1</sup></b>			<b>368</b>	<b>45.5 <math>\pm</math> 4.0</b>	<b>17.0 <math>\pm</math> 5.0</b>	<b>37.5 <math>\pm</math> 4.0</b>	<b>3.3 <math>\pm</math> 0.4</b>	<b>226 <math>\pm</math> 38</b>	<b>0.6 <math>\pm</math> 0.2</b>	<b>0.3 <math>\pm</math> 0.7</b>
Congo Town	D24	1998	48	50.0 $\pm$ 24.0	13.5 $\pm$ 19.0	36.5 $\pm$ 23.0	4.0	218	0.3 $\pm$ 0.7	0
Long Bay Cay	D25	1998	50	50.0 $\pm$ 16.5	14.0 $\pm$ 19.0	36.0 $\pm$ 17.5	3.5	223	0.4 $\pm$ 0.7	4
Oasis	D26	1998	50	31.5 $\pm$ 19.5	29.0 $\pm$ 25.5	39.5 $\pm$ 17.5	3.0	140	0.6 $\pm$ 1.2	0
High Point Cay	D27	1998	35	41.5 $\pm$ 16.0	16.0 $\pm$ 21.5	42.5 $\pm$ 19.5	3.5	190	0.9 $\pm$ 1.2	2
<i>North Rock</i>	D28	1997	33	49.5 $\pm$ 12.5	3.5 $\pm$ 6.0	47.5 $\pm$ 13.0	---	---	1.1 $\pm$ 1.5	0
<i>North Grassy</i>	D29	1997	26	43.0 $\pm$ 25.0	10.0 $\pm$ 16.0	47.0 $\pm$ 24.0	---	---	0.9 $\pm$ 1.8	0
<i>South Grassy</i>	D30	1997	31	49.0 $\pm$ 15.0	4.0 $\pm$ 7.0	46.5 $\pm$ 12.0	---	---	0.6 $\pm$ 0.8	0
<i>Delta</i>	D31	1997	43	41.0 $\pm$ 15.0	4.0 $\pm$ 6.5	55.0 $\pm$ 16.0	---	---	1.1 $\pm$ 1.3	0
<i>Pigeon</i>	D32	1997	39	48.5 $\pm$ 15.5	9.0 $\pm$ 11.0	42.5 $\pm$ 12.5	---	---	1.7 $\pm$ 1.7	0
<i>Saddleback</i>	D33	1997	37	48.0 $\pm$ 16.0	5.5 $\pm$ 9.0	46.5 $\pm$ 15.0	---	---	0.9 $\pm$ 1.1	0
<b>South Andros-all<sup>1</sup></b>			<b>392</b>	<b>45.5 <math>\pm</math> 6.5</b>	<b>11.0 <math>\pm</math> 8.0</b>	<b>38.5 <math>\pm</math> 3.0</b>	<b>3.3 <math>\pm</math> 0.4</b>	<b>193 <math>\pm</math> 38</b>	<b>0.9 <math>\pm</math> 0.4</b>	<b>0.6 <math>\pm</math> 1.4</b>

<sup>1</sup>Quadrats (#) = sum; all other columns = means  $\pm$  standard deviation.

<sup>2</sup>Heights measured as maximum canopy heights in 1997 are not included.

<sup>3</sup>Macroalgal index = relative abundance of macroalgae x macroalgal height.



**Figure 1.** Location of Andros Island, the Bahamas (inset) and the 1997 and 1998 AGRRA survey sites in four areas (North, Central, Bights, South) off eastern Andros. See Tables 1A, 1B for site codes.

# ASSESSMENT OF ANDROS ISLAND REEF SYSTEM, BAHAMAS (PART 2: FISHES)

BY

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and TIMOTHY L. TURNBULL<sup>3</sup>

## ABSTRACT

Coral reef fish assemblages were surveyed at 48 reef-crest and fore-reef habitats along approximately 200 km of reefs on the eastern side of Andros Island in August of 1997 and 1998. A total of 164 species were recorded in roving diver surveys, averaging 55 species per site. Select species density averaged 37.4 individuals/100m<sup>2</sup> in belt transects and was significantly more abundant in reef crests than fore reefs. The select fish assemblages were dominated by scarids, haemulids, and acanthurids, while serranids were ubiquitous but present in low densities (<0.5/100m<sup>2</sup>). Small differences in the community structure of four geographic areas (north, central, bights, south) are indicative of well-mixed populations. Species richness and abundance were comparatively low, particularly in fore-reef habitats, although mean size and biomass were relatively high. The Andros reef fish assemblages may be naturally limited by low recruitment, lack of nursery habitat, or possibly by high levels of predation. The entire reef system may be at high risk to even modest increases in fishing.

## INTRODUCTION

Intact fish assemblages are integral to the functioning of coral reefs and patterns in their diversity, abundance, and size can be used to understand underlying ecological processes such as recruitment, predation, and herbivory. These patterns vary at different spatial and temporal scales and are likely to be influenced by habitat variables such as topographic complexity (e.g., Connell and Kingsford, 1998; Núñez-Lara and Arias-González, 1998), live coral cover (e.g., Bell and Galzin, 1984), depth (e.g., Lewis and Wainwright, 1985), wave energy (McGehee, 1994; Mejia and Garzón-Ferreira, 2000),

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and cross shelf position (Lindeman et al., 1998) as well as species interactions (e.g., predation, competition), suitability of substratum (e.g., algal cover, Lawson et al., 1999), and larval availability and recruitment patterns (e.g., Cowen et al., 2000). The application of comprehensive habitat-based sampling programs (e.g., Ault et al., 2001) has proven effective in gaining insight into complex patterns of fish assemblages.

The spatial variation in the abundance and distribution of fish communities off Andros Island, Bahamas, one of the most extensive reef systems in the Western Atlantic, is poorly known. The reef tract parallels the eastern side of the island extending 217 km from Joulter's Cays in the north to Saddleback Cays in the south. Shallow reef crests and outer slope reefs are the principal reef types although lagoonal patch reefs are also present. The presence of extensive, topographically complex reef crests and fore reefs (Kramer et al., this volume) would tend to suggest that suitable habitat is sufficiently available to support abundant and diverse fish populations off Andros Island.

Historically, fishing has been mostly local, artisanal-level fishing concentrated in the central and southern regions of Andros, with relatively little commercial activity. Harvesting has probably altered fish communities less significantly than reported for other areas in the Caribbean (e.g., Roberts, 1995). However, in the past decade commercial fishing has increased, with larger operations particularly active south of the bights in Mangrove Cay and South Andros. Significant fishing effort is concentrated in December and January around several large Nassau grouper (*Epinephelus striatus*) spawning aggregations located in central and southern Andros. Little data are available on fishing pressures, since statistics on landing sites and total fishing effort are not recorded (Bahamas Reef Environmental Education Foundation and Macalister Elliot and Partners, 1998, unpublished report).

In this paper, we present the results from a large-scale (>100 km) habitat-based assessment of the fishes off Andros Island reefs using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) methodology and examine how elements of species richness, density, and biomass vary spatially and between habitat types. Information on the status of fish communities can provide an essential baseline critical for management and conservation efforts of fishes, particularly targeted species. Spatial trends and the condition of principal reef-building corals and algal populations are presented in Kramer et al. (this volume).

## METHODS

Andros Island is located in the central Bahamas where the Great Bahamas Bank meets the Tongue of the Ocean (Fig. 1). An extensive yet discontinuous fringing bank barrier reef parallels the eastern side of the island. Reef crests are dominated by colonies of *Acropora palmata* (live and standing dead) and display varying degrees of development controlled, in part, by wave energy, reef aspect, and the presence of freshwater creeks. Fore reefs range from hard-bottom assemblages dominated by gorgonians to reefs composed of dense coral growth dominated by the three morphotypes of the *Montastraea annularis* species complex, which reach heights of 2-3 m off the

bottom. Structurally developed fore reefs are often associated with well-developed reef crests and occur at "intermediate" depths of 7-12 m.

The Andros reef tract (Fig. 1) was divided into four geographic areas: north (N); central (C); bights (B); and south (S). Fish surveys were conducted at the same locations as the benthic surveys (Kramer et al., this volume) and are representative of the better-developed reefs within two stratified habitat types (shallow reef crest and intermediate-depth fore reef). In 1997, 17 sites (8 at 1-3 m depth, 9 at 8-12 m) located mainly in the northern and southern areas were surveyed. In 1998, the 31 surveyed sites (15 shallow and 16 deep) were located in all four areas (Tables 1A, 1B). At each site, a combination of belt transects and roving diver surveys were used to assess the fish community structure using the AGRRA methodology at the same time that benthic characteristics [including live stony coral cover, density of "large" ( $\geq 25$  cm diameter) stony corals, relative algal cover] were evaluated.

In 1997, the AGRRA Version 1.0 fish protocol was employed, except that only 2-5 belt transects (each 50 x 2 m) were made at each site because our underwater time was limited. All species and all sizes of haemulids (grunts), scarids (parrotfishes), and serranids (groupers) present in the belt transects were counted during these surveys. In 1998, when AGRRA Version 2.0 (see Appendix One, this volume) was used, 10 belt transects, each 30 x 2 m, were deployed at each site except at four locations (S5, S11, S20, D10 having 6, 4, 5, and 9 transects, respectively). In 1998, counts of serranids were restricted to species of *Epinephelus* and *Mycteroperca*, while scarids and haemulids less than 5 cm in length were not tallied, but the number of fish species quantified in the belt transects was expanded to include bar jack (*Caranx ruber*), yellowtail damsel (*Microspathodon chrysurus*), barracuda (*Sphyraena barracuda*), hogfish (*Lachnolaimus maximus*), Spanish hogfish (*Bodianus rufus*) and all balistids. In 1998, swimming speeds were 6-8 minutes per 30 m transect, while in 1997 they were 8-10 minutes per 50 m transect. Two of the authors (Marks, Turnbull) conducted the fish transects both years, and all roving diver species richness counts were done by the same surveyor (Marks). Fish identification was based on Humann (1994).

Statistical analyses were performed with the program Statistica (Version 5.1). Transect averages of fish density were calculated for each site based on the number of transects deployed and represented as the mean number per transect adjusted to a common unit area of 100 m<sup>2</sup>. Parameters were analyzed by student's t-test and by 1- and 2-way Analysis of Variance (ANOVA). The four geographic areas (N, C, B, S) were analyzed by ANOVA with sites hierarchically nested under areas as random factors. Density and biomass data were checked to ensure that variances met assumptions of homoscedasticity. Regression analysis was used to examine relationships between fish data and benthic habitat variables (coral cover, coral size, coral frequency, depth).

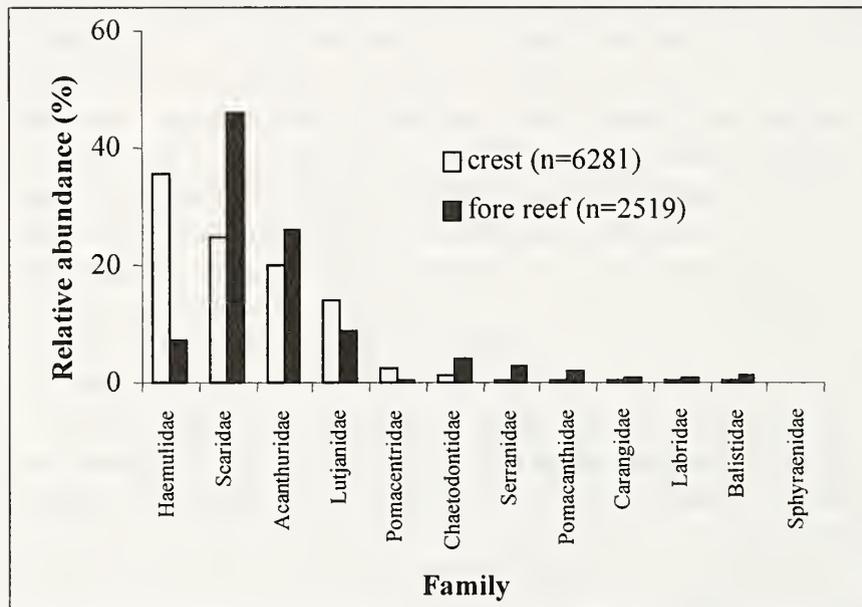
## RESULTS

A total of 164 species, plus several other unidentified species of silversides, herrings, and anchovies, were documented during the roving diver surveys for the entire Andros reef tract (<http://www.reef.org>). For the reef-crest sites, a total of 126 species were

observed during ~30 hours of roving bottom time (Table 1A). In the fore reefs, 144 species were observed during ~33 hours of bottom time (Table 1B). The average number of species encountered per site was 54 in the reef crests (average roving bottom time =78 minutes, n=23 sites) and 56 in the fore reefs (average roving bottom time=79 minutes, n=25 sites). Total species richness within an area (N, C, B, S) ranged from 113-128; approximately 60% of all recorded species were seen in each geographic area. Differences were found in the presence or absence of rare species with low sighting frequencies. For example, several species only seen in northern Andros included black durgon (*Melichthys niger*), trunkfish (*Lactophyrus trigonus*), and spotted trunkfish (*L. bicaudalis*) whereas the diamond blenny (*Malacoctenus boehlkei*) and porkfish (*Anisotrematus virginicus*) were seen in the bights and southern areas but not in the central or northern areas.

A total of 50 fish species were counted (out of a possible 72 AGRRA species) within the belt transects. Forty-four transect species were counted in the reef crests compared to 47 in the fore reefs. Ten AGRRA species not recorded in transects were seen during roving diver surveys at low (<30%) sighting frequencies. In 1998, an average of 21.5 fish species was recorded within transects at each site (n=31 sites). Reef crests had slightly more species (mean=22.5, n=15 sites) on average than fore reefs (mean=20.5, n=16 sites). Species similarity between the four areas of Andros was high with over 75% of the AGRRA-listed transect species counted within at least three of the four areas.

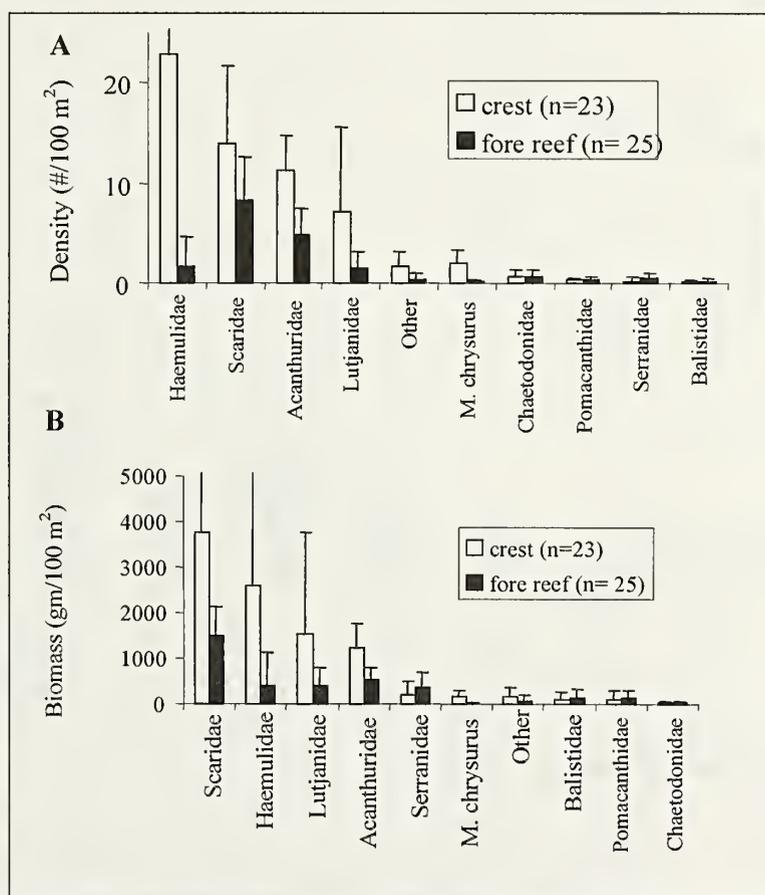
The total number of adult AGRRA fishes in the belt transects was 8,800 with 6,281 individuals counted in reef-crest habitats and 2,519 in fore-reef habitats (all sites and both years combined). Reef-crest communities were dominated by haemulids (37%), scarids (25%), acanthurids (20%), and lutjanids (14%), while fore reefs contained a higher proportion of scarids (46%) and substantially fewer haemulids (7%) (Fig. 2).



**Figure 2.** Relative abundance of AGRRA fishes in reef crests and fore reefs pooled for both years off Andros, Bahamas.

Mean fish density for surveyed transect species was 37.4 individuals/100 m<sup>2</sup> (both habitats and both years combined). Mean family density and biomass for the surveyed

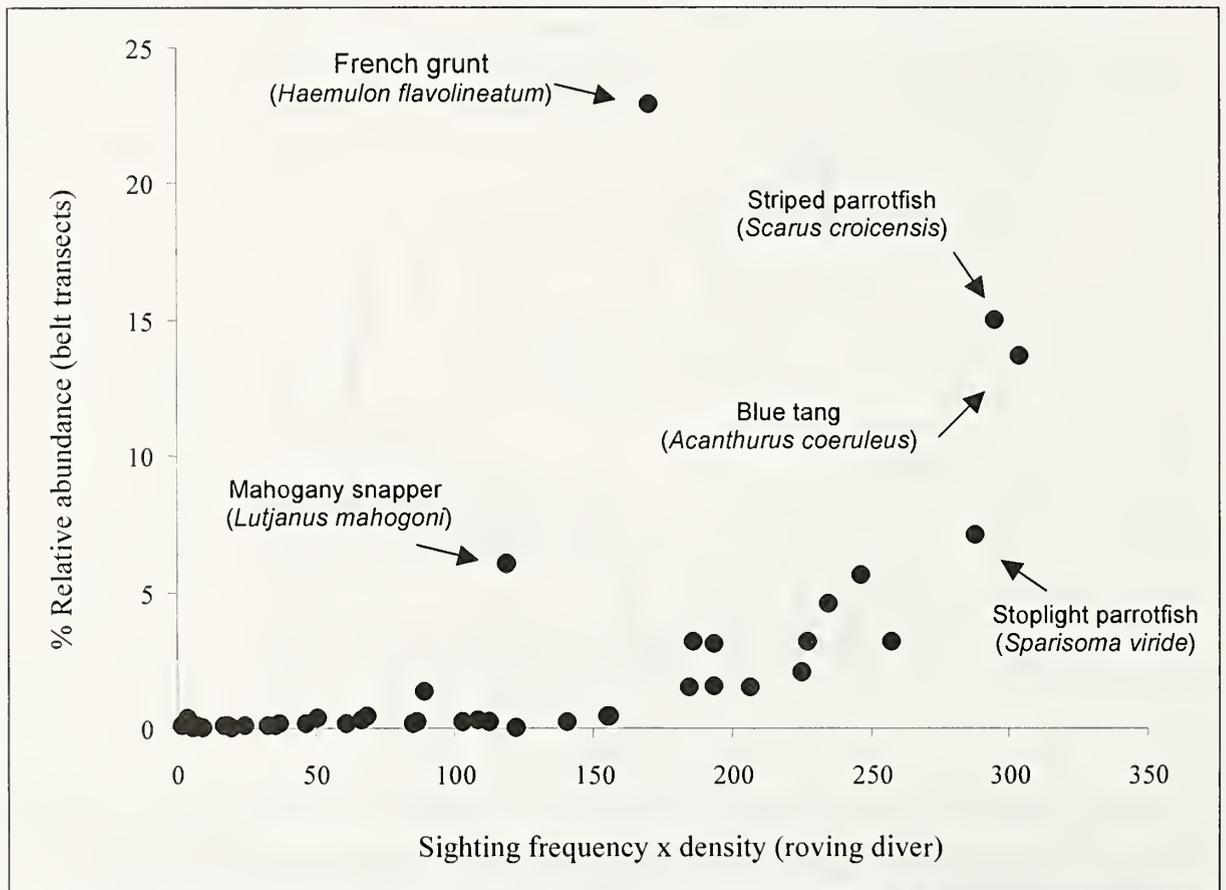
species are shown in Figure 3. Higher fish densities were recorded in the 1998 surveys (overall mean=47.7/100m<sup>2</sup>) than in 1997 (overall mean=19.2/100m<sup>2</sup>). Statistically significant differences were detected for both herbivorous fishes (scarids  $\geq 5$  cm, acanthurids, *M. chrysurus*) (t-test, df=46, t=-3.7, p<0.001) and for carnivores (haemulids  $\geq 5$  cm, lutjanids, select serranids) (t-test, df=46, t=-3.44, p<0.01). When the data for all sites and both years are combined, large, free-ranging herbivores (scarids  $\geq 5$  cm, acanthurids) had relatively high densities (19.5/100 m<sup>2</sup>), more than three times that of major piscivores (select serranids, lutjanids) (4.6/100 m<sup>2</sup>) (Tables 2A,B). For both survey years (all densities in numbers/100m<sup>2</sup>), the reef crests contained significantly more fish (mean=58.1, sd=45.3, n=23 sites) than were found in the fore reefs (mean=18.3, sd=7.8, n=25 sites) (t-test, df=46, t=-4.3, p<0.0001). Moreover, the highest recorded fish densities (>100/100m<sup>2</sup>) consistently occurred in reef crests (e.g., S11, S18, S23) while the lowest densities (<8/100m<sup>2</sup>) were in fore reefs (e.g., D6, D12, D16). Total AGRRA fish density patterns are strongly weighted by scarid, acanthurid, and haemulid densities, all of which were significantly higher in reef-crest habitats (scarids  $\geq 5$  cm, p<.001; acanthurids, p<0.00001; haemulids  $\geq 5$  cm, p<0.001). Families with fewer sightings, including the Chaetodontidae Serranidae, and Balistidae, all had higher densities in the fore reefs (Fig. 3).



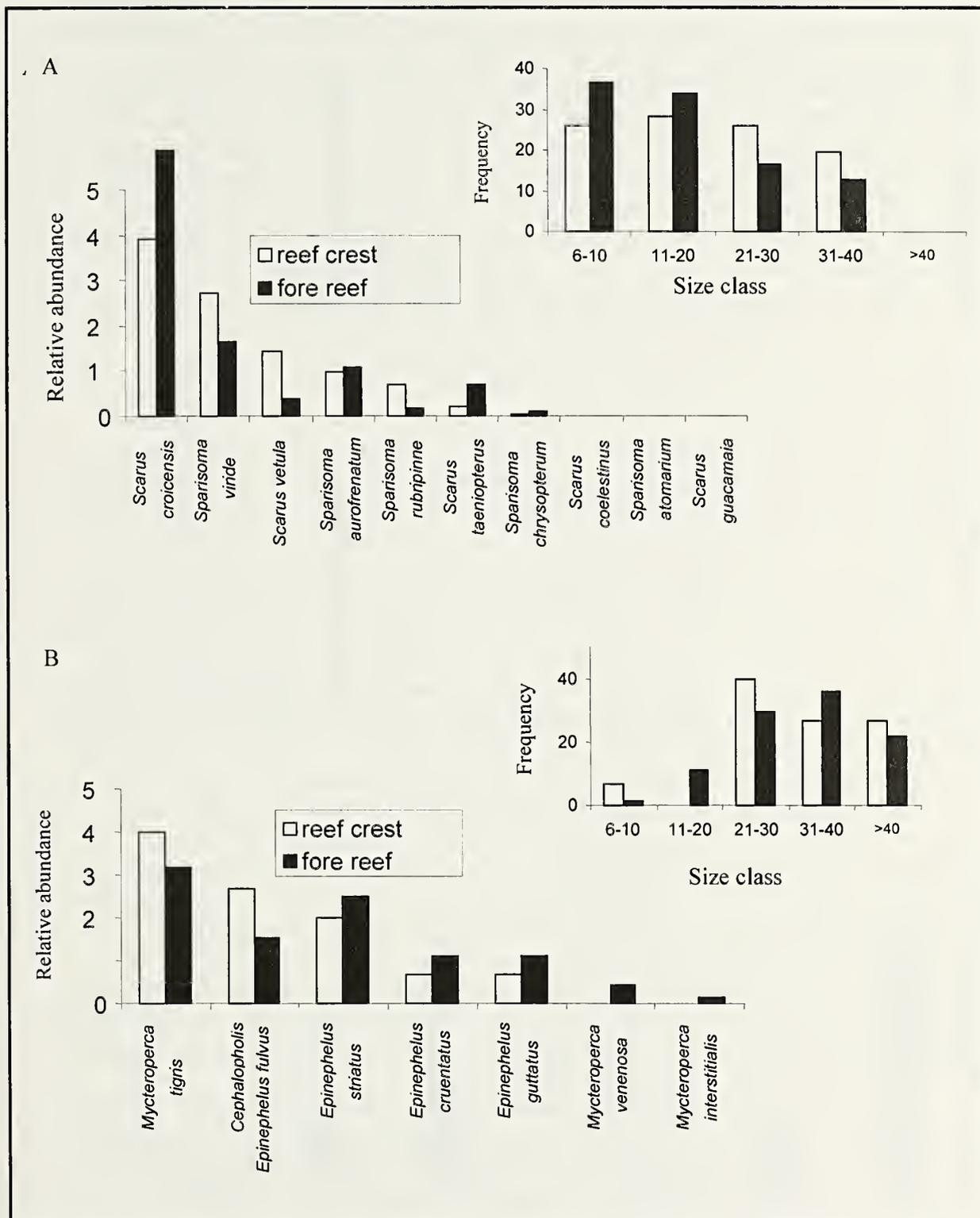
**Figure 3.** (A) Density (mean no. fish/100 m<sup>2</sup>  $\pm$  standard deviation) and (B) biomass (mean g/100 m<sup>2</sup>  $\pm$  standard deviation) for AGRRA fishes, pooled for both habitats (reef crests and fore reefs) and both years (1997 and 1998) off Andros, Bahamas. Other = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Sphyraena barracuda*.

A comparison between the relative abundance of species observed in transects versus their abundance (calculated as the sum of density  $\times$  sighting frequency) in the roving diver surveys is shown in Figure 4 for the 15 most commonly observed AGRRA species for both years combined. Several species of haemulids (e.g., French grunt, *Haemulon flavolineatum*) and lutjanids (e.g., mahogany snapper, *Lutjanus mahogoni*) had clumped distributions with high concentrations only in several shallow sites.

Scarids were well represented with a total of 10 species seen in belt transects in all surveys combined and a higher species richness in fore-reef sites. The two most abundant species were striped parrotfish (*Scarus croicensis*) and stoplight parrotfish (*Sparisoma viride*) (Fig. 5A). The sizes of most adult parrotfishes were considered "average" for their species with approximately 20% in the 31-40 cm class range (Fig. 5A). All three species of acanthurids were present off Andros (average site density = 7.9/100 m<sup>2</sup>, n = 48 sites), with nearly twice as many encountered in reef crests (n = 1,258 in 23 sites) as in fore reefs (n = 652 in 25 sites). Blue tangs (*Acanthurus coeruleus*) made up 61% of the acanthurids seen in the belt transects followed by ocean surgeons (*A. bahianus*) (25%), and doctorfish (*A. chirurgus*) (14%). Most acanthurids were in the 11-20 cm size class. Another potentially important herbivore, the Bermuda chub (*Kyphosus sectatrix*) was rare off Andros (sighting frequency of 31% in the roving diver surveys) and usually occurred as isolated individuals.



**Figure 4.** Relationship between relative abundance in belt transects and roving diver surveys ( $\Sigma$  density  $\times$  sighting frequency in 1997 + 1998) for the 15 most commonly observed AGRRA fish species off Andros, Bahamas.



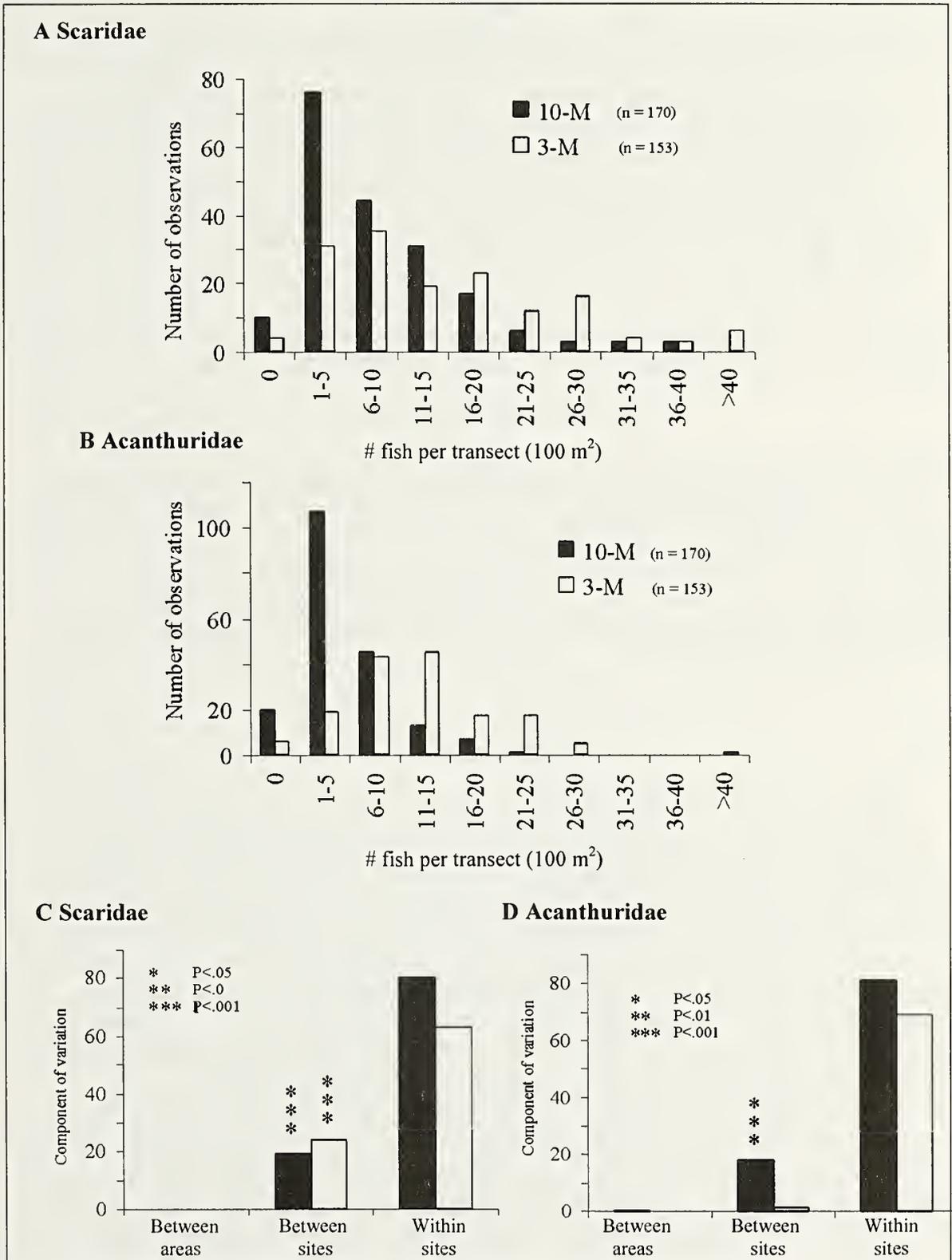
**Figure 5.** Relative species abundance and size frequency distribution in cm of (A) scarids  $\geq 5$  cm and (B) serranids in reef crests and fore reefs off Andros, Bahamas.

The yellowtail damselfish (*M. chrysurus*) was observed at 96% of reef-crest sites (Table 3A, roving diver sighting frequency) at belt transect densities of 1.9/100 m<sup>2</sup>, but at only 26% of fore reef sites at densities of 0.1/100 m<sup>2</sup>. Other territorial damselfish including (in order of sighting frequency in the roving diver surveys) bicolor (*Stegastes partitus*), threespot (*S. planifrons*), dusky (*S. fucus*), longfin (*S. diencaeus*), and cocoa (*S. variabilis*) were similarly most abundant in reef-crest habitats. The roving diver sighting frequency for serranids was high on Andros (e.g., they were observed at nearly all sites). Within belt transects, serranids composed 0.5% of the AGRRA fishes counted in reef crests and 3% in fore-reef sites (Fig. 2). The most frequently encountered serranid species in the belt transects, in order of abundance, were tiger grouper (*Mycteroperca tigris*), coney (*Epinephelus fulvus*), Nassau grouper (*E. striatus*), graysby (*E. cruentatus*), and red hind (*E. guttatus*) (Fig. 5B). Yellowmouth grouper (*M. interstitialis*) and yellowfin grouper (*M. venosa*) were present in fore-reef habitats only, and other serranids, including black grouper (*M. bonaci*) and rock hind (*E. adscensionis*), were only seen during roving diver surveys. The density of serranids within transects was quite low, averaging only 0.39 individuals/100 m<sup>2</sup> for all years and sites combined. Significant differences in serranid density were detected between the reef crests and fore reefs (t-test, df=46, t=-2.4 p<0.05) but not among the four areas (N, C, B, S) (1-way ANOVA, df=3, MS=0.38, F=2.6, p=0.06). The sizes of adult groupers counted in transects were large, with nearly 30% of all groupers being greater than 40 cm in length (Fig. 5B).

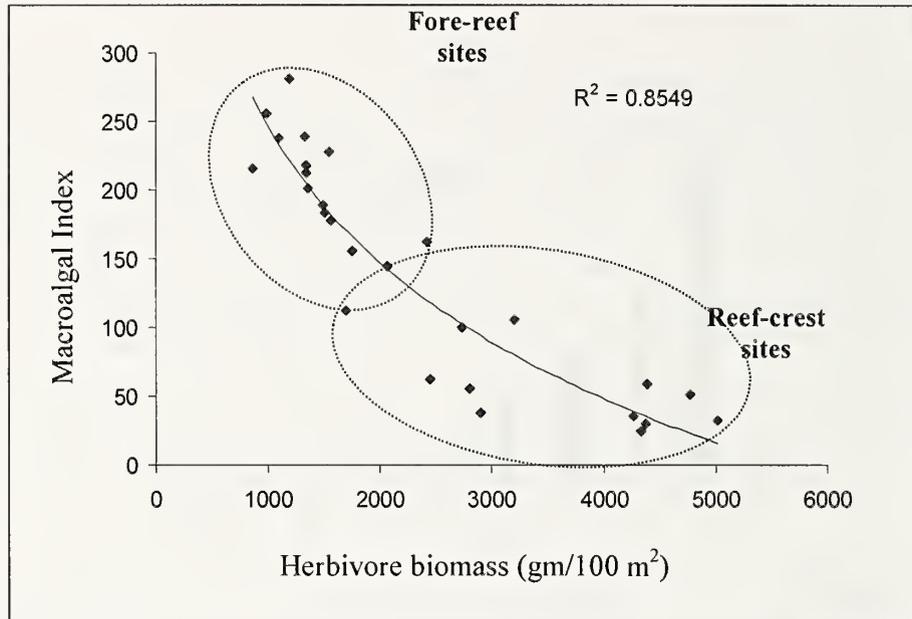
Eleven species of grunts were observed (roving diver surveys), with greater mean densities in belt transects at reef crests (22.8/100 m<sup>2</sup>) than in fore reefs (1.7/100 m<sup>2</sup>). Ten species of snapper were observed during roving diver surveys; schoolmasters (*Lutjanus apodus*) and yellowtail snappers (*Ocyurus chrysurus*) were seen most frequently (Table 3A, B). Snapper densities in belt transects averaged 7.1/100 m<sup>2</sup> on reef crests and 1.5/100 m<sup>2</sup> on fore reefs. Mean snapper density for the shallow sites was significantly (p<0.05) higher in the bights (14.8/100 m<sup>2</sup>) than in the other geographic areas.

Within each of the four areas, variation in total fish density was higher at the within- and between-site scale, particularly for reef-crest habitats, than among the geographic areas. For example, Figure 6 shows the variability in transect abundance at three spatial scales for scarids and acanthurids. Most of the variation (~80%) occurred at the within-site scale, with the remainder at the between-site scale. No significant differences were detected among the four areas of Andros (N, C, B, S) for either total species density (1-way ANOVA, df=3, MS=826, F=0.6, p=0.63) or total fish biomass (1-way ANOVA, df=3, MS=390259, F=1.4, p=0.25). Only grouper biomass was statistically different among the four areas (1-way Anova, df=3, MS=373576, F=4.2, p<0.05) mainly because few were seen on northern reef crests.

A significant positive relationship was found between chaetodontid density and live stony coral cover in fore reefs (p<0.01), but not in reef crests. Analysis of herbivore-macroalgal index relationships within each of the habitat types was significant (p<0.05) only among fore reefs and not reef crests. However, when the two habitats were combined a significant inverse relationship (p<0.001) was found between the macroalgal index (a proxy for macroalgal biomass) and herbivore biomass in 1998 (Fig. 7). The best fit line shown in the figure is a log-fit with reef-crest and fore-reef sites separately circled. A significant negative relationship also existed for depth and herbivore biomass



**Figure 6.** Frequency distribution of fish density recorded within 100 m<sup>2</sup> transects for (A) scarids  $\geq 5$  cm and (B) acanthurids for both reef-crest and fore-reef habitats. Breakdown of the components of variation in fish transect densities at three spatial scales (within a site, between sites, and between areas) for (C) scarids  $\geq 5$  cm and (D) acanthurids in both reef-crest and fore-reef habitats. \*\*\* = level of significance is indicated by number of stars.



**Figure 7.** Regression of mean herbivore biomass (acanthurids, scarids  $\geq 5$  cm, and *Microspathodon chrysurus*) and mean macroalgal index, for 27 sites (all depths combined) assessed during 1998, excluding sites with insufficient ( $n < 10$ ) belt transects.

( $p < 0.001$ ) when all 98 sites were combined. A general inverse pattern between the piscivore density and herbivores was observed within shallow habitats. A weakly significant positive relationship existed between herbivore and piscivore density within only deep sites ( $p = 0.034$ ).

## DISCUSSION

Species richness and relative family and species dominance documented during our surveys are similar to those reported in other western Atlantic reefs (Turnigan and Acosta, 1989; also see [www.reef.org](http://www.reef.org)). However, fish densities along Andros Island are lower, particularly on fore reefs, than in some other areas of the Bahamas (Sluka et al., 1996) and Caribbean (Schmitt, unpublished data; Lewis and Wainwright, 1985; Kramer, this volume). Given the presence and extent of well-developed reef-crest and fore-reef habitats and the relatively modest fishing pressures, a greater abundance of fishes was expected. The lack of strong spatial differences in community structure or abundance among the four geographic areas suggest that the processes governing the structure of the Andros fish populations are fairly uniform along the entire eastern reef tract.

Low fish abundance in the fore reefs may be a result of limited larval supply, unsuccessful pre- and post-settlement processes, or lack of nearby juvenile habitat. No scientific surveys have been conducted on larval abundances, recruitment patterns, nursery habitat, and current patterns along Andros. The majority of larval recruitment is likely to be from local sources since the shallow, extensive carbonate platform banks surrounding the Tongue of the Ocean probably form natural barriers restricting entry of external larvae. Flow through Providence Channel is relatively low (Busby et al., 1966) and may not play a significant role in recruitment dynamics. Thus, fish populations may be influenced primarily by local recruitment processes rather than by input from external

larval sources (Cowen et al., 2000) and local gyres, or eddies may play a significant role in their abundance.

In addition to larval supply, low fish abundances on Andros may be influenced by the availability of adjacent juvenile habitat. Nagelkerken et al. (2001) found that proximity of nearby mangroves and seagrass influenced the abundance of *Ocyurus chrysurus* and *Scarus croicensis* in Curaçao, while the presence of available seagrass nursery habitat affected *Sparisoma chrysopterum*, *Sphyræna barracuda*, and several species of *Lutjanus* and *Haemulon*. Along the eastern coast of Andros, dense seagrass beds are uncommon (total area <6 km<sup>2</sup>) (Kramer, unpublished data) and their paucity may be a limiting factor affecting fish abundance. However, the numerous creeks and bights along the mainland would appear to be ideal nursery habitats particularly as many are lined by mangroves. The relative importance of local versus external recruitment and the role of nursery habitat availability in structuring fish communities off Andros warrant further investigation.

Based on our data, variance in the total density and biomass of the AGRRA fishes is mainly attributable to intrinsic habitat characteristics such as coral cover and structural complexity (Kramer et al., 1999), habitat type and depth, and sampling biases. Differences in fish assemblages between sites of similar habitat type may also result from species interaction variables (not discussed here).

The significant positive relationship between chaetodontid density and live stony coral cover observed on Andros' fore reefs may be associated with high habitat dependency or specialized microhabitat use (Robertson, 1996). The distribution of groupers has also been correlated to habitat features, particularly topographic complexity (Connell and Kingsford, 1998; Sluka et al., 1996, 2001), which may explain the relatively high abundance of *M. tigris* and other large-body-size groupers in the network of tunnels and overhangs between the 1-3 m tall columns of the *Montastraea annularis* species complex in the fore reefs. Roberts and Ormond (1987) found the availability of shelter holes to influence the abundance of some fishes (scarids, acanthurids, labrids, and pomacentrids) while live coral cover was only important for chaetodontids. Although not examined in this study, other parameters such as adjacent habitat diversity, patch size, proximity to tidal channels, and distance from mainland may also influence fish distribution on Andros. Distinguishing which habitat factors are most important in structuring fish communities is difficult since many coexist or are additive, thus a more experimental approach isolating specific variables is needed to better understand factors governing these spatial patterns.

Although acanthurids are known to prefer shallow habitats, the high abundance of scarids in reef crests off Andros is surprising since other studies have typically found their densities to be greater in deeper water (Lewis and Wainwright, 1985; also see Horn, 1989). The high abundance of acanthurids and scarids in the shallow reef crests, as well as the large size of scarids, probably result in significantly greater herbivory here than in the fore reefs. Algal communities in each habitat reflect these presumed differences in herbivory, and the strong inverse relationship between macroalgal index and herbivore biomass (Fig. 7) implies top-down control over algal assemblages. At shallow depths, the algal community is predominantly crustose corallines and turf algae, both of which are indicative of well-grazed surfaces (Steneck and Dethier, 1994). In contrast, the fore reefs

have few grazed surfaces and are dominated by fleshy algae (e.g., *Dictyota* spp., *Microdictyon marinum*) (Kramer et al., this volume). Reduction of herbivory with depth has been related to decreased trophic carrying capacity (Hay and Goertemiller, 1983; Steneck, 1988). Physical factors such as sedimentation and wave energy may also contribute to algal composition in particular habitats (e.g., Fabricius and De'ath, 2001). In addition, the lack of preferred food and the increased risk of predation may also influence herbivore movements and levels of grazing.

The scarcity of the important grazing sea urchin, *Diadema antillarum*, (Kramer et al., this volume) is also a major factor in explaining the dominance of macroalgae in the fore reefs off Andros. Historically, *Diadema* was the dominant herbivore off Andros (Miner, 1933; Newell and Rigby, 1951), but populations severely declined during 1983 as in other areas of the Caribbean (e.g., Lessios et al., 1984). Following the *Diadema* die-off, Morrison (1988) found that erect, resistant macroalgal species increased at intermediate depths on heavily fished Jamaican reefs. Robertson (1991) found that the abundance of two herbivorous acanthurids, *Acanthurus coeruleus* and *A. chirurgus*, increased significantly on Panama patch reefs after the die-off of *Diadema*. It appears that herbivorous fishes along Andros may have filled the trophic niche of *Diadema* as dominant grazers on high-relief reef crests but not in fore reefs. These results emphasize the importance of herbivory on Andros' coral and algal communities. The recovery of *Diadema* to its former population densities may be a critical first step towards reducing macroalgae in the fore reefs.

The risk of predation may influence the spatial patterns of prey fishes (e.g., Reinthal and Macintyre, 1994), and may also partially explain the lower densities of scarids and acanthurids on Andros' fore reefs. The high relief (2-4 m) and structurally complex arrangement of corals and pinnacles associated with intermediate depth fore-reef zones would appear to be ideal for predators because of the abundance of ambush locations. However, a weakly significant positive relationship was found between the densities of key herbivores and piscivores in the fore reefs off Andros, suggesting that other factors are also important. It is also possible that the distribution of predators determined from snapshot daytime surveys are not indicative of overall levels of predation. If fish predation on Andros' fore reefs is unusually high, it must still be explained why densities of predators (serranids, lutjanids) are low compared to other Bahamian island groups (Sluka et al., 1996). Our survey protocol may have systematically underestimated their abundance as transect methods have been found to underreport the density of fish species with wide ranges or low abundances (Thresher and Gunn, 1986). Furthermore, the size and number of transects can also greatly influence density estimates (Sale and Sharp, 1983). Significantly fewer fish were recorded in nearly all families (including serranids) in the 1997 surveys compared to the 1998 surveys which is thought to be a direct result of the low number of transects employed at each site. However, temporal and spatial differences in the fish community for the two sample periods cannot be ruled out as additional factors.

Relief and structural complexity of the habitat, by providing hiding places, may also directly influence reported fish abundances for predatory and sedentary species. It is possible that for both survey years rare and cryptic species off Andros, including most groupers, may have been systematically undersampled in fore reefs because of the

unusually high relief (2-4 m) of the columnar corals. A pilot comparison between 30 m fish transects swum in 7 minutes (standard) versus 15 minutes (extended) revealed that significantly greater numbers of serranids were seen when observers had more time to look beneath overhangs and into tunnels. Thus, observed densities of groupers may be related to the structural complexity of the substratum; the reason for the low abundance of other ecologically important (e.g., herbivores, corallivores) or commercially significant fish (e.g., snappers) in fore reefs remains unexplained.

Low fish abundance, particularly of commercially significant species, on Andros fore reefs may be an indication that the entire reef tract is overfished (Roberts, 1995). Fishing pressures on Andros are thought to be light-to-moderate in comparison to many other areas of the Caribbean and are mainly targeted towards large-bodied groupers (e.g., *M. tigris*, *E. striatus* and *M. interstitialis*) and snappers (e.g., *Lutjanis analis*, *L. synagris* and *O. chrysurus*). Commercial and subsistence fishers also target species such as barracuda, triggerfish, hogfish, and some grunts. Although fish such as angelfish, parrotfish, and surgeonfish are not yet targeted, there may be significant bycatch associated with trap fishing that extends to lower trophic levels (T. Turnbull, personal observation).

The result of targeted fishing often leads to a decrease in the size and abundance of harvested species or, in more severe cases, to the loss of those species (e.g., Roberts, 1995; Koslow et al., 1988). Were the Andros reef tract as overfished as other reefs in the Caribbean, we might expect targeted guilds such as serranids to be dominated by species having small adult body sizes (*Epinephelus fulvus*, *E. cruentatus* and *E. guttatus*). In fact, large-bodied targeted species (*M. tigris* and *E. striatus*) were two of the most frequently seen serranids on Andros and nearly 30% of all groupers that were observed within transects were very large (>40 cm length). In addition, Andros is one of the few locations in the Bahamas to have at least two well-documented grouper spawning aggregations (at High Cay and Tinker Rocks, respectively). These aggregations are estimated to contain hundreds to thousands of fishes during spawning periods, although historic numbers are suspected to have been much higher (T. Turnbull, personal observations).

Commercial fishing intensity has increased in the last several years, driven by higher market prices (Bahamas Reef Environmental Education Foundation and Macalister Elliot and Partners, 1998, unpublished report), particularly in southern Andros where illegal fishing by non-Bahamian residents occurs. Given the low abundance of fishes and the growing demand for targeted fish species, its fish populations are likely to be vulnerable to even modest increases in fishing intensity. Several large marine protected areas have been proposed along Andros, and at least two are in the process of being implemented (G. Larson, personal communication). In addition, in 1999 and 2000, the Bahamas Fisheries Department prohibited fishing at the High Cay aggregation site during critical spawning periods (five days around the full moon in December-February) in an effort to reduce fishing pressures. However, the aggregation was not closed in 2001. The implementation of adaptive management strategies, such as a seasonal closure on the grouper fishery, is essential for maintaining sustainable fish populations along Andros.

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Table 1A. Reef-crest site information for AGGRA fish surveys off Andros Island, Bahamas (1997 sites are italicized).

Reef crest site name	Site code	Latitude (N°')	Longitude (W°')	Survey date	Depth (m)	% live stony coral cover (%± se)	Macroalgal relative abundance (%± se)	Bottom time (min.)	Fish transects (#/m)	Roving diver fish species (#)
<i>N. Joulters</i>	S1	25.31322	78.03433	Aug 15 97	2.5	---	6.5 ± 10.0	125	4/50	74
<i>Golding</i>	S2	25.22392	78.08557	Aug 18 97	1.5	---	7.5 ± 12.0	95	4/50	57
<i>Morgan</i>	S3	25.1601	78.0029	Aug 17 97	1.5	---	8.0 ± 8.5	90	3/50	62
<i>Coconut Point</i>	S4	25.12885	77.98303	Aug 17 97	1.5	---	8.5 ± 15	100	5/50	60
Mahore	S5	25.06367	77.93783	Aug 21 98	1	43.0 ± 14.5	12.0 ± 16	80	6/30	70
S. Standard 2	S7	24.84493	77.86098	Aug 20 98	0.5	41.0 ± 22.5	13.0 ± 7.5	60	10/30	45
<b>North Andros-all</b>								<b>550</b>		<b>98</b>
N. Love Hill	S8	24.77435	77.80772	Aug 18 98	1	41.5 ± 14.0	17.0 ± 23.0	70	10/30	47
China Point	S10	24.75133	77.80767	Aug 18 98	1	54.0 ± 31.0	10.5 ± 9.5	60	10/30	54
Red Rock	S11	24.72917	77.77017	Aug 7 98	1	37.0 ± 15.5	13.5 ± 21.5	60	4/30	63
<i>S. Long Rock</i>	S13	24.63067	77.691	Aug 9 97	3	---	14.5 ± 14.0	185	2/50	47
Sugar Rock	S15	24.5448	77.68372	Aug 10 98	2	20.0 ± 11.0	8.0 ± 8.5	60	10/30	50
<b>Central Andros-all</b>								<b>435</b>		<b>90</b>
Autec2-South	S16	24.484	77.699	Aug 11 98	1	24.0 ± 11.5	10.5 ± 11.0	90	10/30	56
N. Bight	S17	24.4395	77.69807	Aug 12 98	2	33.5 ± 9.0	13.5 ± 20.0	65	10/30	48
Big Wood	S18	24.36703	77.68235	Aug 13 98	1.5	36.0 ± 11.0	26.0 ± 20.5	60	10/30	47
Autec 3	S19	24.34315	77.67068	Aug 12 98	1	39.5 ± 15.5	9.0 ± 12.0	60	10/30	44
Middle Bight	S20	24.3069	77.65638	Aug 13 98	1.5	35.5 ± 10.5	13.5 ± 20.0	60	5/30	49
Mangrove C.	S21	24.29167	77.6462	Aug 13 98	1.5	25.0 ± 10.5	10.5 ± 13.5	60	10/30	55
Mangrove S.	S22	24.25315	77.62917	Aug 13 98	1	30.0 ± 13.5	20.5 ± 17.5	60	10/30	56
<b>Bights-all</b>								<b>455</b>		<b>78</b>
Congo Town	S23	24.3013	77.64788	Aug 15 98	1	30.5 ± 14.5	11.5 ± 15.5	60	10/30	53
Long Bay	S24	24.09793	77.53703	Aug 16 98	1	35.5 ± 11.5	11.0 ± 14.5	60	10/30	48
<i>North Rock</i>	S25	23.79092	77.42637	Aug 14 97	2	---	14.5 ± 19.0	80	3/50	52
<i>North Grassy</i>	S26	23.77822	77.41902	Aug 15 97	2	---	10.5 ± 11.5	80	3/50	61
<i>Pigeon</i>	S28	23.6965	77.377	Aug 11 97	1.5	---	18.0 ± 15.0	75	4/50	54
<b>South Andros</b>								<b>355</b>		<b>85</b>
<b>Reef crests- all</b>								<b>1795</b>		<b>126</b>

Table 1B. Fore-reef site information for AGGRA fish surveys off Andros Island, Bahamas (1997 sites are italicized).

Fore reef site name	Site code	Latitude (N°')	Longitude (W°')	Survey date	Depth (m)	% live stony coral cover (mean ± se)	Macroalgal rel. abundance (mean ± se)	Bottom time (min.)	Fish transects (#/m)	Roving diver fish species (#)
<i>N. Joutlers</i>	D1	25.3132	78.0856	Aug 16 97	6.5	---	49.0 ± 18.0	90	4/50	70
<i>Nichols</i>	D2	25.1438	72.9875	Aug 17 97	9.5	---	35.5 ± 15.0	85	5/50	52
S. Staniard 2	D7	24.50630	77.56437	Aug 20 98	9	37.0 ± 13.0	35.0 ± 14.0	60	10/30	49
<b>North Andros-all</b>								<b>235</b>		<b>84</b>
West Klein	D10	24.7450	77.7847	Aug 7 98	10.5	24.5 ± 8.0	48.0 ± 19.5	215	9/30	59
<i>S. Long Rock</i>	D12	24.6307	77.6910	Aug 9 97	8	---	62.5 ± 20.5	95	2/50	61
Long Rock	D13	24.6260	77.6910	Aug 8 98	8.5	22.5 ± 8.0	48.0 ± 20.0	---	10/30	---
<i>Mid Long Rock</i>	D14	24.37547	77.41587	Aug 9 97	9.5	---	59.0 ± 32.0	185	5/50	89
Green Cay	D15	24.5958	77.6933	Aug 9 98	11	11.0 ± 4.0	54.5 ± 20.0	90	10/30	58
Sugar Rock	D16	24.5402	77.6821	Aug 10 98	11.5	18.0 ± 7.0	41.5 ± 12.5	90	10/30	58
<b>Central Andros-all</b>								<b>675</b>		<b>104</b>
Bristol Galley	D17	24.5263	77.6892	Aug 10 98	11.5	17.0 ± 5.0	45.5 ± 21.0	85	10/30	54
Autec 2	D18	24.5064	77.6970	Aug 11 98	12.5	9.0 ± 3.0	51.5 ± 13.0	70	10/30	61
Autec 2-South	D19	24.4833	77.6965	Aug 11 98	10.5	15.0 ± 5.0	40.0 ± 15.5	70	10/30	53
N. Bight	D20	24.4393	77.6962	Aug 12 98	9.5	31.0 ± 12.5	45.0 ± 15.5	70	10/30	56
Autec 3	D21	24.3432	77.6707	Aug 14 98	10.5	25.5 ± 10.5	44.0 ± 16.0	60	10/30	50
Middle Bight	D22	24.3097	77.6530	Aug 14 98	9	28.0 ± 8.0	42.0 ± 13.0	60	10/30	53
Mangrove N.	D23	24.3011	77.6478	Aug 14 98	9.5	5.5 ± 3.0	50.5 ± 19.0	50	10/30	35
<b>Bights-all</b>								<b>415</b>		<b>102</b>
Congo Town	D24	24.3012	77.6479	Aug 15 98	10.5	21.0 ± 15.5	50.0 ± 24.0	70	10/30	70
Long Bay Cay	D25	24.0997	77.5334	Aug 16 98	10.5	19.0 ± 9.0	50.0 ± 16.5	60	10/30	63
Oasis	D26	23.9476	77.3867	Aug 16 98	6	45.5 ± 21.0	31.5 ± 19.5	75	10/30	72
High Point Cay	D27	23.4200	77.4600	Aug 17 98	9	35.0 ± 7.5	41.5 ± 16.0	60	10/30	54
<i>North Rock</i>	D28	23.7965	77.4222	Aug 13 97	7.5	---	49.5 ± 12.5	75	5/50	43
<i>North Grassy</i>	D29	23.7803	77.4168	Aug 13 97	9.5	---	43.0 ± 25.0	75	4/50	55
<i>South Grassy</i>	D30	23.7285	77.3974	Aug 12 97	7.5	---	49.0 ± 15.0	80	4/50	41
<i>Pigeon</i>	D32	23.6945	77.3742	Aug 11 97	9	---	48.5 ± 15.5	80	3/50	36
<i>Saddleback</i>	D33	23.6767	77.3703	Aug 11 97	8.5	---	48.0 ± 16.0	95	4/50	52
<b>South Andros-all</b>								<b>660</b>		<b>116</b>
<b>Fore reefs - all</b>								<b>1985</b>		<b>144</b>

Table 2A. Density (mean  $\pm$  sd) of AGRRA fishes by site in reef crests off Andros Island (1997 sites are italicized).

Reef crest site name	Site code	Year	Herbivores (#/100m <sup>2</sup> )				Carnivores (#/100m <sup>2</sup> )			Total AGRRA fishes (#/100m <sup>2</sup> )
			Acanthuridae	Scaridae ( $\geq 5$ cm)	<i>Microspathodon chrysurus</i>	Haemulidae ( $\geq 5$ cm)	Lutjanidae	Serranidae <sup>2</sup>		
<i>N. Joulters</i>	S1	1997	7 $\pm$ .5	11 $\pm$ 6	---	15 $\pm$ 19.5	0.5 $\pm$ 0.5	0	33 $\pm$ 21.5	
<i>Golding</i>	S2	1997	13.5 $\pm$ 7.5	10.5 $\pm$ 10.5	---	10.5 $\pm$ 9	3 $\pm$ 4	0	37 $\pm$ 18	
<i>Morgan</i>	S3	1997	12 $\pm$ 12.5	7.5 $\pm$ 4	---	27.5 $\pm$ 13.5	0.5 $\pm$ 0.5	0	47.5 $\pm$ 12	
<i>Coconut Point</i>	S4	1997	7 $\pm$ 4	5 $\pm$ 6	---	0.5 $\pm$ 0.5	0.5 $\pm$ 1	0	13.5 $\pm$ 9	
Mahore	S5	1998	14.5 $\pm$ 6.5	39.5 $\pm$ 21.5	2.5 $\pm$ 3.5	3.5 $\pm$ 5.5	0.5 $\pm$ 0.5	0	62.5 $\pm$ 24.5	
S. Standard 2	S7	1998	14 $\pm$ 7.5	23 $\pm$ 24.5	1 $\pm$ 1.5	0.5 $\pm$ 0.5	8.5 $\pm$ 12	0	48 $\pm$ 27.5	
<b>North Andros<sup>1</sup></b>			<b>11.3 <math>\pm</math> 3</b>	<b>15.9 <math>\pm</math> 13</b>	<b>1.9 <math>\pm</math> 1</b>	<b>9.5 <math>\pm</math> 10</b>	<b>2.1 <math>\pm</math> 3</b>	<b>0</b>	<b>40.4 <math>\pm</math> 17</b>	
N. Love Hill	S8	1998	12 $\pm$ 6.5	8.5 $\pm$ 8.5	0.5 $\pm$ 1.5	2 $\pm$ 3.5	2 $\pm$ 2.5	0	26.5 $\pm$ 10.5	
China Point	S10	1998	8 $\pm$ 4	9.5 $\pm$ 4.5	0	48.5 $\pm$ 70	4 $\pm$ 5.5	0	72.5 $\pm$ 72	
Red Rock	S11	1998	9.5 $\pm$ 2	14 $\pm$ 5	3.5 $\pm$ 3.5	179.5 $\pm$ 107.5	13 $\pm$ 21.5	0	221.5 $\pm$ 104	
<i>S. Long Rock</i>	S13	1997	11.5	8	0	0	0	0	20	
Sugar Rock	S15	1998	12 $\pm$ 8	23.5 $\pm$ 13.5	1 $\pm$ 1.5	4.5 $\pm$ 5.5	2 $\pm$ 3.5	1.5 $\pm$ 1.5	45.5 $\pm$ 27	
<b>Central Andros<sup>1</sup></b>			<b>10.7 <math>\pm</math> 2</b>	<b>12.6 <math>\pm</math> 3</b>	<b>1.1 <math>\pm</math> 2</b>	<b>46.9 <math>\pm</math> 84</b>	<b>4.1 <math>\pm</math> 6</b>	<b>0.3 <math>\pm</math> 0</b>	<b>77.2 <math>\pm</math> 94</b>	
Autec2-South	S16	1998	10.5 $\pm$ 4.5	13.5 $\pm$ 12.5	1 $\pm$ 1	14 $\pm$ 11.5	32.5 $\pm$ 65.5	1 $\pm$ 1.5	74 $\pm$ 74	
N. Bight	S17	1998	12.5 $\pm$ 5	20 $\pm$ 13	2 $\pm$ 2.5	13 $\pm$ 14	9.5 $\pm$ 10	0.5 $\pm$ 0.5	60.5 $\pm$ 22	
Big Wood	S18	1998	9.5 $\pm$ 8	9 $\pm$ 7	3.5 $\pm$ 5.5	72 $\pm$ 102	20.5 $\pm$ 23	0.5 $\pm$ 1	117 $\pm$ 117	
Autec 3	S19	1998	16 $\pm$ 7	18 $\pm$ 18	4 $\pm$ 3	4.5 $\pm$ 6	8 $\pm$ 8	0.5 $\pm$ 1	53 $\pm$ 19	
M. Bight	S20	1998	15.5 $\pm$ 2	15 $\pm$ 9.5	1 $\pm$ 1.5	14.5 $\pm$ 20	14.5 $\pm$ 19.5	0.5 $\pm$ 1	63.5 $\pm$ 38.0	
Mangrove C.	S21	1998	11.5 $\pm$ 8.5	16 $\pm$ 10	1.5 $\pm$ 1.5	23.5 $\pm$ 31.5	11.5 $\pm$ 16	0.7 $\pm$ 1	66.5 $\pm$ 40.5	
Mangrove S.	S22	1998	14 $\pm$ 7	11 $\pm$ 9	0.5 $\pm$ 1	12 $\pm$ 15.5	7.5 $\pm$ 10.5	0 $\pm$ 0.5	47.5 $\pm$ 23.3	
<b>Bights<sup>1</sup></b>			<b>12.7 <math>\pm</math> 3</b>	<b>14.7 <math>\pm</math> 4</b>	<b>2.0 <math>\pm</math> 1</b>	<b>21.9 <math>\pm</math> 28</b>	<b>14.8 <math>\pm</math> 10</b>	<b>0.5 <math>\pm</math> 0</b>	<b>68.7 <math>\pm</math> 25</b>	
Congo Town	S23	1998	18 $\pm$ 13	14 $\pm$ 7.5	4 $\pm$ 3	61 $\pm$ 38	19 $\pm$ 17.5	0	118.5 $\pm$ 53.0	
Long Bay	S24	1998	13.5 $\pm$ 8.5	22 $\pm$ 9	2.5 $\pm$ 2.5	7.5 $\pm$ 10.5	7.5 $\pm$ 13	0 $\pm$ 0.5	53.5 $\pm$ 22.5	
<i>NorthRock</i>	S25	1997	5.5 $\pm$ 1.5	7.5 $\pm$ 8.5	---	1 $\pm$ 1	0.5 $\pm$ 0.5	0 $\pm$ 0.5	15.5 $\pm$ 8.5	
<i>North Grassy</i>	S26	1997	7 $\pm$ 1	8 $\pm$ 7	---	10.5 $\pm$ 9.5	0	0.5 $\pm$ 0.5	27 $\pm$ 13	
<i>Pigeon</i>	S28	1997	5.5 $\pm$ 3.5	6 $\pm$ 4.5	---	0	0	0	13.5 $\pm$ 7.5	
<b>South Andros<sup>1</sup></b>			<b>9.9 <math>\pm</math> 5</b>	<b>11.5 <math>\pm</math> 6</b>	<b>3.3 <math>\pm</math> 1</b>	<b>16.0 <math>\pm</math> 23</b>	<b>5.3 <math>\pm</math> 7</b>	<b>0.2 <math>\pm</math> 0</b>	<b>52.9 <math>\pm</math> 35</b>	

<sup>1</sup>Mean  $\pm$  standard error; <sup>2</sup>*Epinephelus* spp. and *Myceteroperca* spp.

Table 2B. Density (mean  $\pm$  sd) of AGRRA fishes by site in fore reefs off Andros Island (1997 sites are italicized).

Reef crest site name	Site code	Year	Herbivores (#/100m <sup>2</sup> )			<i>Microspathodon chrysurus</i>	Carnivores (#/100m <sup>2</sup> )		Total AGRRA fishes (#/100m <sup>2</sup> )
			Acanthuridae	Scaridae ( $\geq$ 5 cm)	Haemulidae ( $\geq$ 5 cm)		Lutjanidae	Serranidae <sup>2</sup>	
<i>N. Joulters</i>	D1	1997	3 $\pm$ 1.5	10.5 $\pm$ 5	---	1.5 $\pm$ 16.5	1 $\pm$ 0	1 $\pm$ 1	30 $\pm$ 12.5
<i>Nichols</i>	D2	1997	2.5 $\pm$ 1	2.5 $\pm$ 2	---	0 $\pm$ 0.5	0	0	6 $\pm$ 2
S. Staniard 2	D6	1998	12 $\pm$ 5.5	11 $\pm$ 7	0.5 $\pm$ 1.5	1 $\pm$ 1.5	2 $\pm$ 4	0.5 $\pm$ 1	29 $\pm$ 12.5
<b>North Andros<sup>1</sup></b>	D10	1998	<b>5.7 <math>\pm</math> 5.4</b>	<b>8.1 <math>\pm</math> 4.8</b>	<b>0.7 <math>\pm</math> 0</b>	<b>5.4 <math>\pm</math> 8.1</b>	<b>0.9 <math>\pm</math> 0.9</b>	<b>0.4 <math>\pm</math> 0.4</b>	<b>21.7 <math>\pm</math> 13.7</b>
West Klein	D12	1997	6.5 $\pm$ 4.5	10 $\pm$ 6.5	0	1.5 $\pm$ 1	1.5 $\pm$ 1.5	1 $\pm$ 1	23.5 $\pm$ 10
<i>S. Long Rock</i>	D13	1998	2	1.5	---	3.5	0.5	0	7.5
Long Rock	D14	1997	6.5 $\pm$ 4.5	14 $\pm$ 10.5	0	2.5 $\pm$ 2.5	2 $\pm$ 3	1 $\pm$ 1.5	29 $\pm$ 14.5
<i>Mid Long Rock</i>	D15	1998	2.5 $\pm$ 1	2.5 $\pm$ 2	---	0 $\pm$ 0.5	0	0	6 $\pm$ 2
Green Cay	D16	1998	2.5 $\pm$ 2.5	15.5 $\pm$ 9	0	0.5 $\pm$ 0.5	1 $\pm$ 1.5	1 $\pm$ 1	21 $\pm$ 11
Sugar Rock	D17	1998	4 $\pm$ 4	5 $\pm$ 4.5	0	0.3 $\pm$ 1	2.5 $\pm$ 4	1.5 $\pm$ 1.5	14.5 $\pm$ 10.5
<b>Central Andros<sup>1</sup></b>	D18	1998	<b>4 <math>\pm</math> 2</b>	<b>8.1 <math>\pm</math> 6</b>	<b>0</b>	<b>1.3 <math>\pm</math> 1</b>	<b>1.2 <math>\pm</math> 1</b>	<b>0.7 <math>\pm</math> 1</b>	<b>16.9 <math>\pm</math> 9</b>
Bristol Galley	D19	1998	3.5 $\pm$ 2.5	12 $\pm$ 10.5	0	1.5 $\pm$ 2	2 $\pm$ 4	0.5 $\pm$ 1	21 $\pm$ 15
Autec 2	D20	1998	4.5 $\pm$ 2.5	6.5 $\pm$ 6.5	0	0.5 $\pm$ 1	2 $\pm$ 2	1 $\pm$ 1	16 $\pm$ 6.5
Autec 2-South	D21	1998	3.5 $\pm$ 3	5.5 $\pm$ 4.5	0	0	4.5 $\pm$ 10.5	0.5 $\pm$ 1	15 $\pm$ 10.5
N. Bight	D22	1998	6 $\pm$ 3	5.5 $\pm$ 4.5	0.5 $\pm$ 0.5	1 $\pm$ 1	2 $\pm$ 2.5	1 $\pm$ 1	20 $\pm$ 8.5
Autec 3	D23	1998	5 $\pm$ 3	17.5 $\pm$ 13.5	0	0 $\pm$ 0.5	2 $\pm$ 2.5	0 $\pm$ 0.5	27 $\pm$ 13.5
Middle Bight	D24	1998	4 $\pm$ 3	11.5 $\pm$ 7	0	0 $\pm$ 0.5	2 $\pm$ 3	0 $\pm$ 0.5	19.5 $\pm$ 8.5
Mangrove N.	D25	1998	6.5 $\pm$ 6	3.5 $\pm$ 2.5	0	0 $\pm$ 0.5	0	0.5 $\pm$ 1	11.5 $\pm$ 6.5
<b>Bights<sup>1</sup></b>	D26	1998	<b>4.7 <math>\pm</math> 1</b>	<b>8.9 <math>\pm</math> 5</b>	<b>0</b>	<b>0.5 <math>\pm</math> 0</b>	<b>2.0 <math>\pm</math> 1</b>	<b>0.6 <math>\pm</math> 0</b>	<b>18.5 <math>\pm</math> 5</b>
Congo Town	D27	1998	3.5 $\pm$ 3	9 $\pm$ 7.5	0 $\pm$ 0.5	1 $\pm$ 2	1 $\pm$ 1.5	0.5 $\pm$ 1	17.5 $\pm$ 10
Long Bay Cay	D28	1998	3.5 $\pm$ 4	12.5 $\pm$ 8.5	0	1 $\pm$ 1.5	0 $\pm$ 0.5	0.5 $\pm$ 1	19 $\pm$ 10
Oasis	D29	1998	13.5 $\pm$ 6.5	7 $\pm$ 4.5	0.5 $\pm$ 1.5	3.5 $\pm$ 3.5	7.5 $\pm$ 8.5	0.5 $\pm$ 0.5	35.5 $\pm$ 15.5
High Point Cay	D30	1998	5 $\pm$ 3	8 $\pm$ 6	0	1.5 $\pm$ 1.5	3 $\pm$ 5	0.5 $\pm$ 0.5	19.5 $\pm$ 8
<i>North Rock</i>	D31	1997	5 $\pm$ 3	4 $\pm$ 4.5	---	0.5 $\pm$ 0.5	0.4 $\pm$ 0.5	0	10 $\pm$ 6.5
<i>North Grassy</i>	D32	1997	4 $\pm$ 3.5	8 $\pm$ 7.5	---	2 $\pm$ 2	0	0.5 $\pm$ 0.5	15 $\pm$ 13.5
<i>South Grassy</i>	D33	1997	5 $\pm$ 2	5 $\pm$ 2	---	0.5 $\pm$ 0.5	0	0.5 $\pm$ 0.5	11 $\pm$ 2.5
<i>Pigeon</i>	D34	1997	2.5 $\pm$ 1.5	7 $\pm$ 6.5	---	1.5 $\pm$ 1.5	0.5 $\pm$ 0.5	0	13.5 $\pm$ 12
<i>Saddleback</i>	D35	1997	3.5 $\pm$ 4	11.5 $\pm$ 6	---	1.5 $\pm$ 0.6	1 $\pm$ 1.5	1 $\pm$ 1.5	20.5 $\pm$ 12
<b>South Andros<sup>1</sup></b>	D36	1997	<b>5 <math>\pm</math> 3</b>	<b>8.1 <math>\pm</math> 3</b>	<b>0.2 <math>\pm</math> 0</b>	<b>1.4 <math>\pm</math> 1</b>	<b>1.4 <math>\pm</math> 2</b>	<b>0.4 <math>\pm</math> 0</b>	<b>17.9 <math>\pm</math> 8</b>

<sup>1</sup>Mean  $\pm$  standard error; <sup>2</sup>*Epinephelus* spp. and *Myceroperca* spp.

Table 3A. Twenty-five most frequently sighted fish species during roving diver surveys for all reef-crest sites combined off Andros Island, with density (mean  $\pm$  sd) for species counted in belt transects.

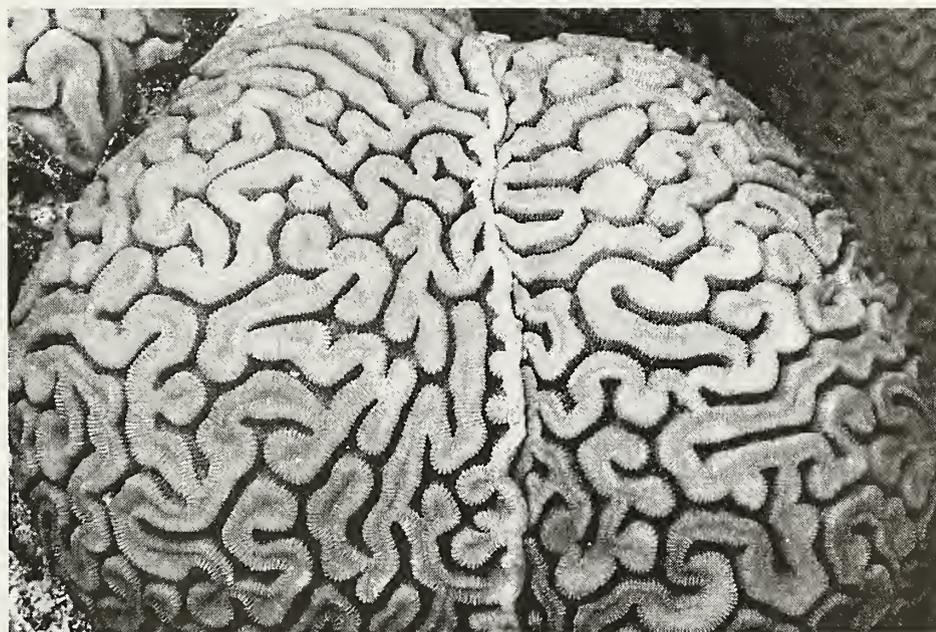
Scientific name	Common name	Roving Diver			Belt transect 1998 density (#/100m <sup>2</sup> )
		Sighting Frequency (SF)	Density (1-4)	SF x Density	
<i>Reef-crest sites</i>					
<i>Thalassoma bifasciatum</i>	Bluehead	100	3.5	350	----
<i>Acanthurus coeruleus</i>	Blue Tang	100	3.3	330	7.2 $\pm$ 6.7
<i>Abudefduf saxatilis</i>	Sergeant Major	100	3	300	----
<i>Halichoeres garnoti</i>	Yellowhead Wrasse	100	2.8	280	----
<i>Acanthurus bahianus</i>	Ocean Surgeonfish	100	2.8	280	3.6 $\pm$ 4.2
<i>Lutjanus apodus</i>	Schoolmaster	100	2.7	270	3.5 $\pm$ 7.4
<i>Aulostomus maculatus</i>	Trumpetfish	96	1.9	182	----
<i>Sparisoma viride</i>	Stoplight Parrotfish	96	3	287	4.7 $\pm$ 5.4
<i>Microspathodon chrysurus</i>	Yellowtail Damselfish	96	2.9	278	1.9 $\pm$ 2.7
<i>Scarus vetula</i>	Queen Parrotfish	96	2.8	268	2.4 $\pm$ 3.2
<i>Haemulon sciurus</i>	Bluestriped Grunt	96	2.5	240	2.4 $\pm$ 4.9
<i>Ophioblennius atlanticus</i>	Redlip Blenny	96	2.5	240	----
<i>Scarus croicensis</i>	Striped Parrotfish	92	3.2	293	6.4 $\pm$ 9.0
<i>Sparisoma aurofrenatum</i>	Redband Parrotfish	92	2.9	266	1.5 $\pm$ 2.1
<i>Sparisoma rubripinne</i>	Redfin Parrotfish	92	2.8	256	1.2 $\pm$ 2.2
<i>Ocyurus chrysurus</i>	Yellowtail Snapper	92	2.6	238	0.9 $\pm$ 3.0
<i>Haemulon plumieri</i>	White Grunt	92	2.5	229	0.7 $\pm$ 1.6
<i>Haemulon flavolineatum</i>	French Grunt	88	3.2	280	21.0 $\pm$ 49.9
<i>Chromis cyanea</i>	Blue Chromis	88	3.1	271	----
<i>Stegastes partitus</i>	Biolor Damselfish	88	2.8	245	----

Table 3B. Twenty-five most frequently sighted fish species during roving diver surveys for all fore-reef sites combined off Andros Island, with density (mean  $\pm$  sd) for species counted in belt transects.

Scientific name	Common name	Roving Diver			Belt transect 1998 density (#/100m <sup>2</sup> )
		Sighting Frequency (SF)	Density (1-4)	SF x Density	
<i>Fore-reef sites</i>					
<i>Acanthurus coeruleus</i>	Blue Tang	97	3	290	3.5 $\pm$ 4.0
<i>Sparisoma viride</i>	Stoptlight Parrotfish	97	2.9	280	1.6 $\pm$ 2.6
<i>Thalassoma bifasciatum</i>	Bluehead	94	3.4	318	----
<i>Chromis cyanea</i>	Blue Chromis	94	3.1	290	----
<i>Scarus croicensis</i>	Striped Parrotfish	94	3	281	5.6 $\pm$ 7.3
<i>Stegastes partitus</i>	Bicolor Damselfish	94	3	281	----
<i>Aulostomus maculatus</i>	Trumpetfish	94	1.9	178	----
<i>Gramma loreto</i>	Fairy Basslet	90	3	271	----
<i>Sparisoma aurofrenatum</i>	Redband Parrotfish	90	2.7	244	1.0 $\pm$ 1.7
<i>Ocyurus chrysurus</i>	Yellowtail Snapper	90	2.5	226	1.1 $\pm$ 3.4
<i>Canthigaster rostrata</i>	Sharpnose Puffer	90	2.4	217	----
<i>Caranx ruber</i>	Bar Jack	90	2.3	208	0.2 $\pm$ 0.9
<i>Lutjanus apodus</i>	Schoolmaster	90	2.3	208	0.8 $\pm$ 2.8
<i>Stegastes planifrons</i>	Threespot Damselfish	90	2.2	199	----
<i>Mycteroperca tigris</i>	Tiger Grouper	90	1.9	172	0.2 $\pm$ 0.5
<i>Clepticus parrae</i>	Creole Wrasse	87	3.7	322	----
<i>Coryphopterus personatus</i>	Masked Goby	87	3.5	305	----
<i>Halichoeres garnoti</i>	Yellowhead Wrasse	87	2.7	235	----
<i>Haemulon plumieri</i>	White Grunt	87	2.1	183	0.3 $\pm$ 0.7
<i>Chaetodon capistratus</i>	Foureye Butterflyfish	87	2.1	183	0.7 $\pm$ 1.3



**Plate 4A.** The need for large-scale comparable data on coral reef condition in the Western Atlantic led to the initiation of the AGRR Program and the development of its key reef health indicators. Given their importance in constructing the three-dimensional framework of coral reefs, the condition of scleractinian and hydrozoan corals, like this *Montastraea annularis*, is a primary focus of the AGRR benthos protocol. (Photo Robert S. Steneck)



**Plate 4B.** Distinguishing colony boundaries before estimating size or partial mortality is essential. A colony is defined on the basis of common skeletal or live tissue connections and/or by polyp size and color. Two closely adjacent colonies of *Diploria labyrinthiformis* are recognized by a thin lip of raised skeleton and live tissues. Common skeleton at the bases of the lobes of *Montastraea annularis* (Plate 3A), help in the recognition of individual colonies. (Photo Robert W. Steneck)

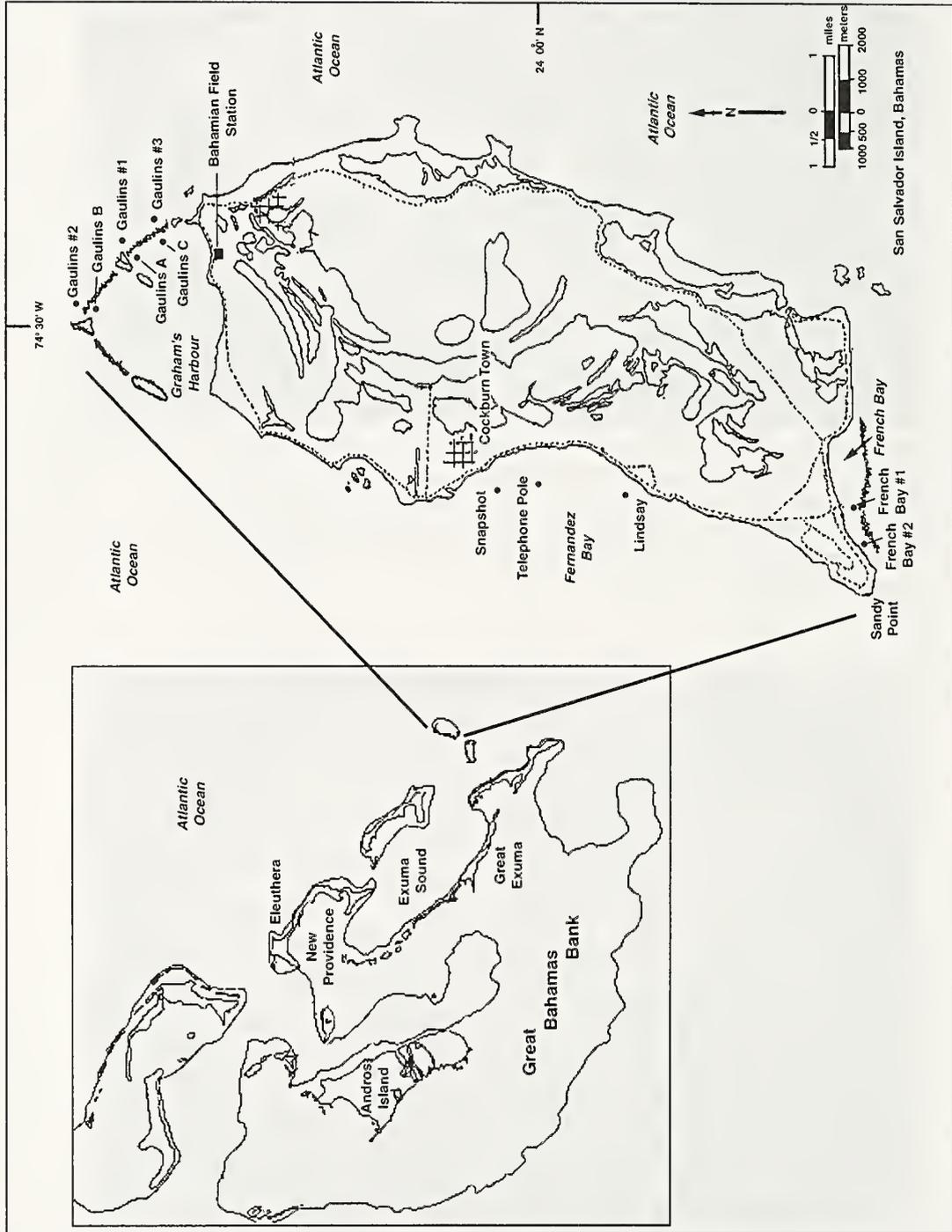


Figure 1. AGRRA survey sites at San Salvador Island, Bahamas.

# ASSESSMENT OF CORAL REEFS OFF SAN SALVADOR ISLAND, BAHAMAS (STONY CORALS, ALGAE AND FISH POPULATIONS)

BY

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## ABSTRACT

During assessments at 11 shallow reef sites on San Salvador Island, Bahamas in June 1998 we found low prevalence of disease, bleaching, and recent partial-colony mortality among stony corals (10 cm minimum diameter). Old partial-colony mortality was >50% in *Acropora palmata*; however, recent tissue losses were low and it had recruits at several sites. Total (recent + old) partial-colony mortality of the *Montastraea annularis* species complex exceeded 30% on leeward patch reefs and back reefs. Groupers (serranids), snappers (lutjanids), and grunts (haemulids) were rare. Parrotfishes (scarids) were uncommon at most sites and surgeonfishes (acanthurids) were the dominant herbivores. Macroalgae, particularly browns that are seldom grazed by surgeonfishes, were the dominant algal functional group. The green macroalga *Microdictyon marinum* was extremely abundant and overgrowing *Porites porites* on leeward patch reefs. To facilitate their conservation, San Salvador Island's reef resources should be designated as a marine reserve.

## INTRODUCTION

The condition of coral reefs worldwide is in decline (Wilkinson, 2000) and the reefs of the greater Caribbean region have emerged as one area of particular concern (Ginsburg, 1994). Reductions in coral cover and diversity, with concomitant increases in macroalgal abundances in the Western Atlantic, have been related to the well-documented mass mortality of the herbivorous sea urchin *Diadema antillarum*

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(Lessios, 1988; Hughes et al., 1999), to increases in coral diseases (Bruckner and Bruckner, 1997; Santavy and Peters, 1997), to overfishing (Hughes, 1994; Koslow et al., 1994), and to widespread elevated sea surface temperatures resulting in coral bleaching (Brown, 1997; Wilkinson et al., 1999). While the greatest degradation of Caribbean reefs is associated with large human populations (Hughes, 1994; Causey et al., 2000), fewer studies document the condition of coral reefs in areas experiencing less human disturbance as has been until quite recently the case off San Salvador, Bahamas.

San Salvador Island (24°N, 74°30'W), located 600 km ESE of Miami on an isolated carbonate platform (Fig. 1), is bordered by a narrow shelf with an abrupt shelf-edge break leading to a very steep slope. Marine water quality is excellent with deep water close offshore and no immediate sources of concentrated pollutants. Its eastern and southeastern coasts typically are windward to the prevailing trade winds. A well-developed, *Acropora palmata*-dominated, bank-barrier reef lies off the northern coast and smaller bank-barrier reefs occur along the southeast and southern coasts. Hundreds of small patch reefs dot the island's eastern shelf; larger patch reefs occur in the broad coastal embayments on the leeward western shelf.

Relatively little has been published on the ecological health and short-term changes to San Salvador's coral reefs. Curran et al. (1994) assessed stony coral cover and diversity at two leeward patch reefs between 1984 and 1992. At Telephone Pole Reef, rapidly growing colonies of *Porites porites* were replacing dead and broken branches of *Acropora cervicornis* which had been a spatial dominant until decimated by white-band disease (Aronson and Precht, 1997) during the early 1980s. During the same interval, the percent of live stony coral cover on nearby Snapshot Reef exhibited no significant change; however, there was an overall increase in the sizes of monitored coral heads and noticeably less macroalgae than seen at present. Similarly, Meyer et al. (1991) reported slight increases in the population densities of two species of the crinoid *Nemaster* residing in large colonies of the *Montastraea annularis* species complex at Snapshot Reef. More recently, stony coral cover, seaward of Snapshot Reef at the 10 m CARICOMP monitoring site, experienced a slight decline and macroalgae had a twofold increase between 1994 and 1998 (Woodley et al., 1997, 2000; Gerace et al., 1998).

In the Bahamas, the major commercial fishery is the spiny lobster, *Panulirus argus*, followed by snappers and groupers. Reports of overfishing are widespread (Woodley et al., 2000). The queen conch (*Strombus gigas*) provides another important, but smaller-scale (usually subsistence level), fishery although densities are declining (Stoner, 1996). Fishing regulations now prohibit the taking of nonlipped queen conch and the use of scuba gear for any kind of fishing. However, few data exist regarding the status of the marine fisheries off San Salvador Island.

Until recently San Salvador has experienced little pressure from human activities. Now the island is rapidly becoming a more popular tourist destination with impetus from the opening of a large Club Med in October 1992 and recent expansion of airport facilities, including extension of the runway to permit landing of intercontinental jet aircraft. Currently, several dive boats operate primarily off the western (leeward) coast bringing up to 200 snorkel and scuba divers per day to the reefs (Kevin Collin, personal communication). With San Salvador in a state of flux with respect to human impact, the Atlantic and Gulf Rapid Reef Assessment (AGRRA) surveys reported here provide an

important baseline for the current condition of San Salvador's coral reefs as well as the status of its finfish populations.

## METHODS

Our assessment of reef-building corals, algae, and fish populations was focused on three areas of San Salvador Island (Fig. 1). Three shallow (1-4 m) fore-reef sites on the exposed northern bank-barrier at Gaulin's Reef have low (<3 m) spur formations extending seaward from the reef crest to a carbonate pavement dominated by sea fans and *Millepora* spp. (*Acropora palmata* is the dominant coral of the reef crest.) Three sites at 2-8 m on the Gaulin's back reef are characterized by large (1-4 m tall) colonies of the *Montastraea annularis* complex along with *Millepora* spp. and *A. palmata*. Three leeward patch reefs in Fernandez Bay (Snapshot, Telephone Pole, Lindsay) are each in water depths of 3-7 m. Snapshot Reef (~200 m offshore) consists of an aggregation of individual coral colonies dominated by the *M. annularis* complex. Telephone Pole Reef (~250 m offshore) is dominated by large colonies of the *M. annularis* complex interspersed with *Porites porites* growing on dead *A. cervicornis*. Lindsay Reef extends out from a sandy beach and experiences a relatively high sediment load. Two windward patch reefs, at depths of approximately 3-5 m and shoreward of a well-developed reef crest in French Bay, are dominated by dead *A. palmata* and *Agaricia agaricites*. These 11 survey sites were selected to be representative of the majority of reef types and exposure conditions occurring off San Salvador. We also considered Telephone Pole and Snapshot Reefs to be "strategic sites" because of earlier survey data (Curran et al., 1994) and their popularity as tourist dive sites.

Stony coral and algal populations were assayed during June 1998 by five-six divers/survey. AGRRA Version 1 benthic protocols (see Appendix One, this volume) were used with the following modifications: stony corals  $\geq 10$  cm in diameter were included in the surveys; coral diameter and height were measured to the nearest cm for smaller corals (10-25 cm in diameter) and to the nearest 5 cm for larger (>25 cm) colonies. The *Montastraea annularis* complex was treated as a single species. Sediment deposits in the algal quadrats were removed by hand before estimating the abundance of crustose coralline algae. *Diadema antillarum*, being rare, was not counted. Training sessions were conducted with all divers censusing "practice" transects at the leeward patch reefs; species identifications, percent cover estimates, and coral disease and bleaching assessments were compared to ensure sampling consistency. We used Humann's (1993) reef coral guide for most coral species identifications.

A stationary visual census technique (Bohnsack and Bannerot, 1986) was employed by two divers to survey the fish populations. All sampling occurred between 10:00 a.m. and 3:00 p.m. At each sampling point, all species belonging to eight families (Acanthuridae, Chaetodontidae, Haemulidae, Labridae, Lutjanidae, Pomacentridae, Scaridae, Serranidae) observed in five minutes within a 7.5 m radius cylinder were recorded. Each census was begun three minutes after laying a measuring tape on the substratum by counting all individuals of all species observed in the pre-set radius within the initial field of view. New sectors of fields of view were then scanned by rotating in one direction. Abundances of species moving in schools were taken when first observed

in the sampling cylinder (it was important to count fishes moving in schools immediately because they were unlikely to remain in the sampling area). When very large schools were present it was sometimes necessary to estimate numbers in 10s or 50s. Fish lengths were estimated in cm using a T-shaped tool marked every 5 cm to help avoid underwater magnification problems (Bohnsack and Bannerot, 1986). We recorded the number of individuals, plus the minimum, maximum, and mean estimated lengths for each of the eight fish families. We used Humann's (1994) reef fish guide for species identifications. Littler and Littler (2000) was later consulted for macroalgae.

As time permitted on the leeward patch reefs, two divers measured herbivorous fish grazing rates following the AGRRA methodology given in Appendix One. All grazing rate measurements were made between 10:00 a.m. and 2:00 p.m. during the peak time for grazing activity (Lewis, 1986).

## RESULTS

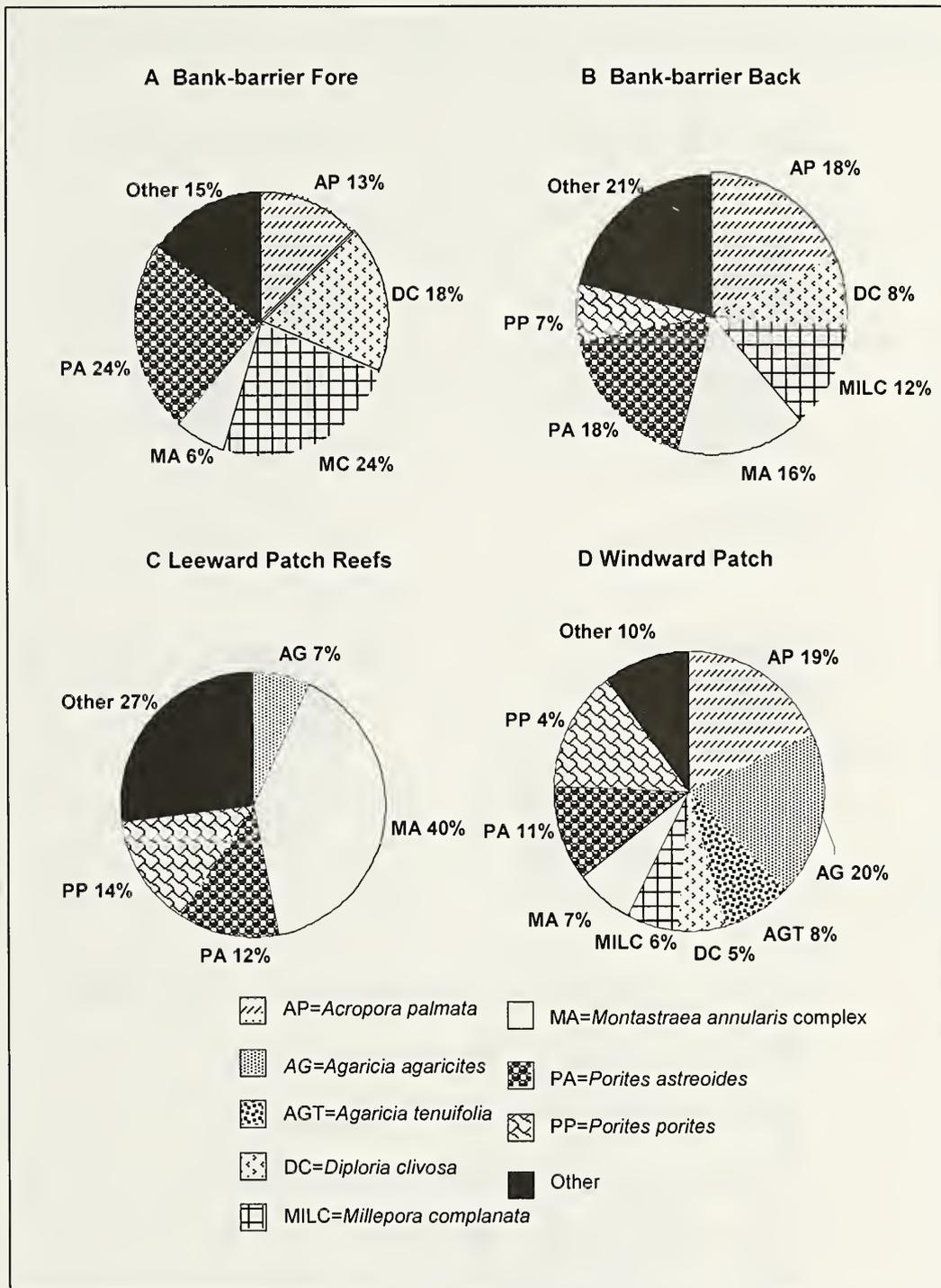
### Stony Corals

*Species composition.* At each site, we censused at least 10 transects with each transect sampling ~9-12 stony corals that were  $\geq 10$  cm in diameter (Table 1). In terms of numerical abundance, *Acropora palmata* was fairly important (means of 13-19%) along the Gaulin's bank-barrier reef and on the windward patch reefs at French Bay (Fig. 2A, B,D). While  $>50\%$  of the upper surfaces of the *A. palmata* colonies were long dead (means for the six Gaulin's sites ranged from 42-78%; mean of 65% at French Bay), very little ( $<2\%$ ) recent partial mortality of colony surfaces (hereafter recent mortality) was evident for this species.

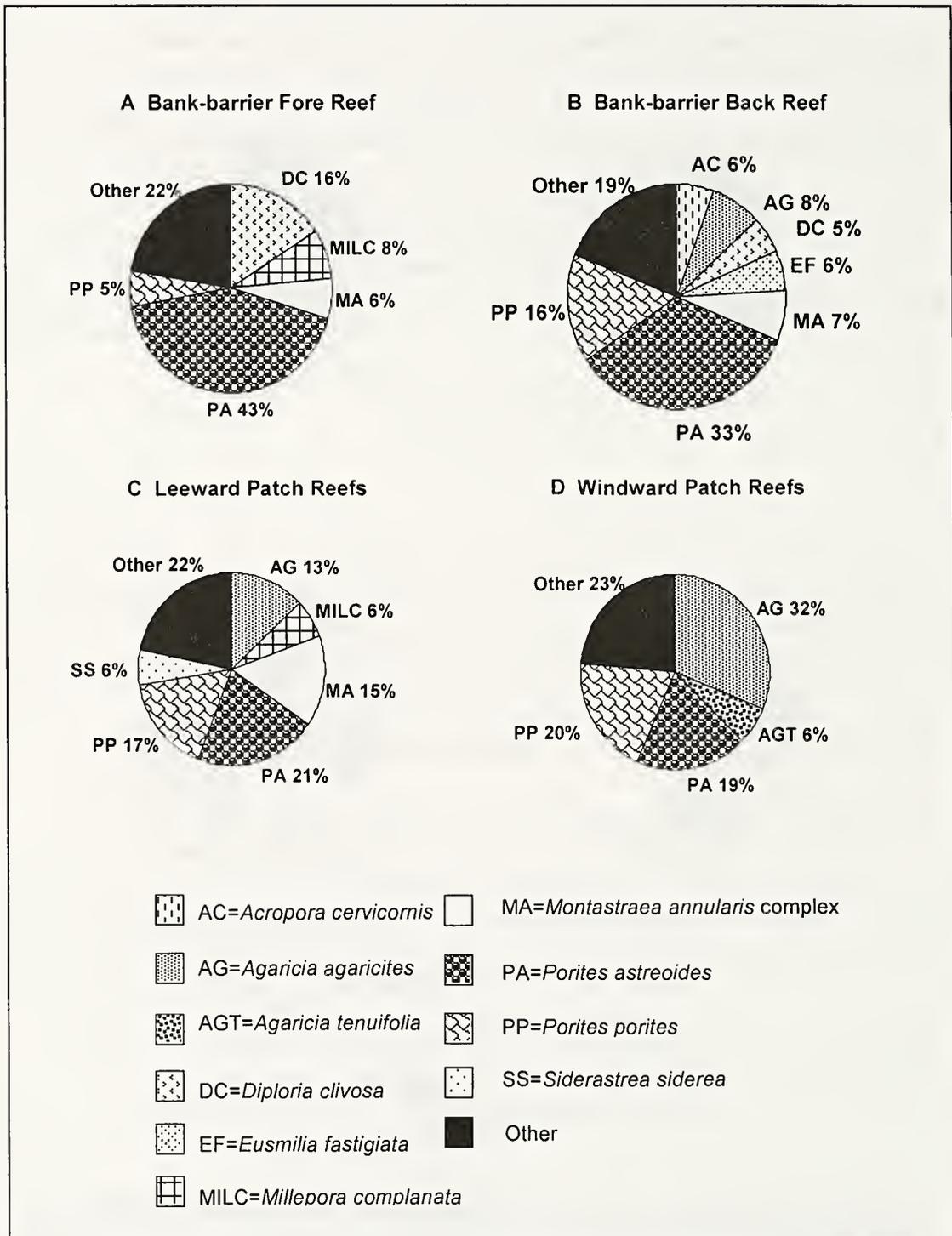
The *Montastraea annularis* complex dominated (mean 40% of all colonies) the leeward patch reefs in Fernandez Bay (Fig. 2C). Recent mortality here was fairly low ( $<4\%$  of their upper surfaces), while nearly 40% (range 30-48% for three leeward patch reefs) of the corresponding surfaces were long dead. The *Montastraea annularis* complex and *Diploria clivosa* together contributed about 25% of the colonies in Gaulin's back reef (Fig. 2B). Approximately 35% of the upper surfaces of colonies of *M. annularis* complex on the back reef were long dead.

*Millepora complanata*, which forms extensive thickets on the spurs, contributed nearly a quarter (24%) to the total abundance of stony corals at Gaulin's fore reef (Fig. 2A). Colonies of *Agaricia* spp. were important on the windward patch reefs and *Porites* spp. were numerically abundant at most sites (Fig. 2).

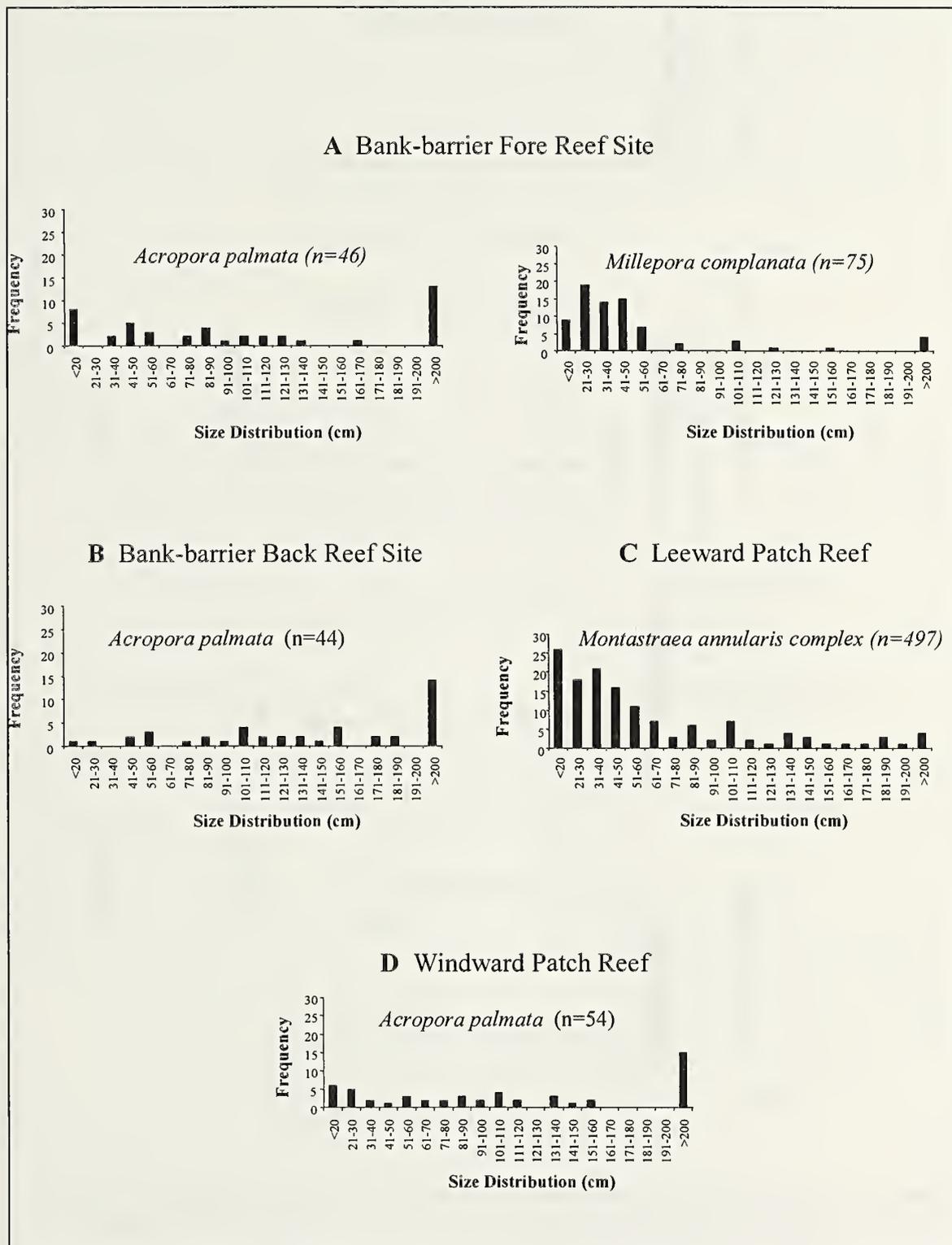
*Recruits.* The composition of recruits (Fig. 3) was largely dominated by *Porites astreoides* (especially on the bank-barrier reef), *P. porites* (all patch reefs, Gaulin's back reef) and *Agaricia agaricites* (especially on windward patch reefs). Collectively, poritids and agariciids contributed 45-70% of the recruit densities. Although 15% of the recruits were of the *Montastraea annularis* complex on the leeward patch reefs and accounted for  $<10\%$  of the recruits on the bank-barrier reef, none were found on the windward patch reefs. At all four habitat types, *Acropora palmata* represented  $<5\%$  of the recruits.



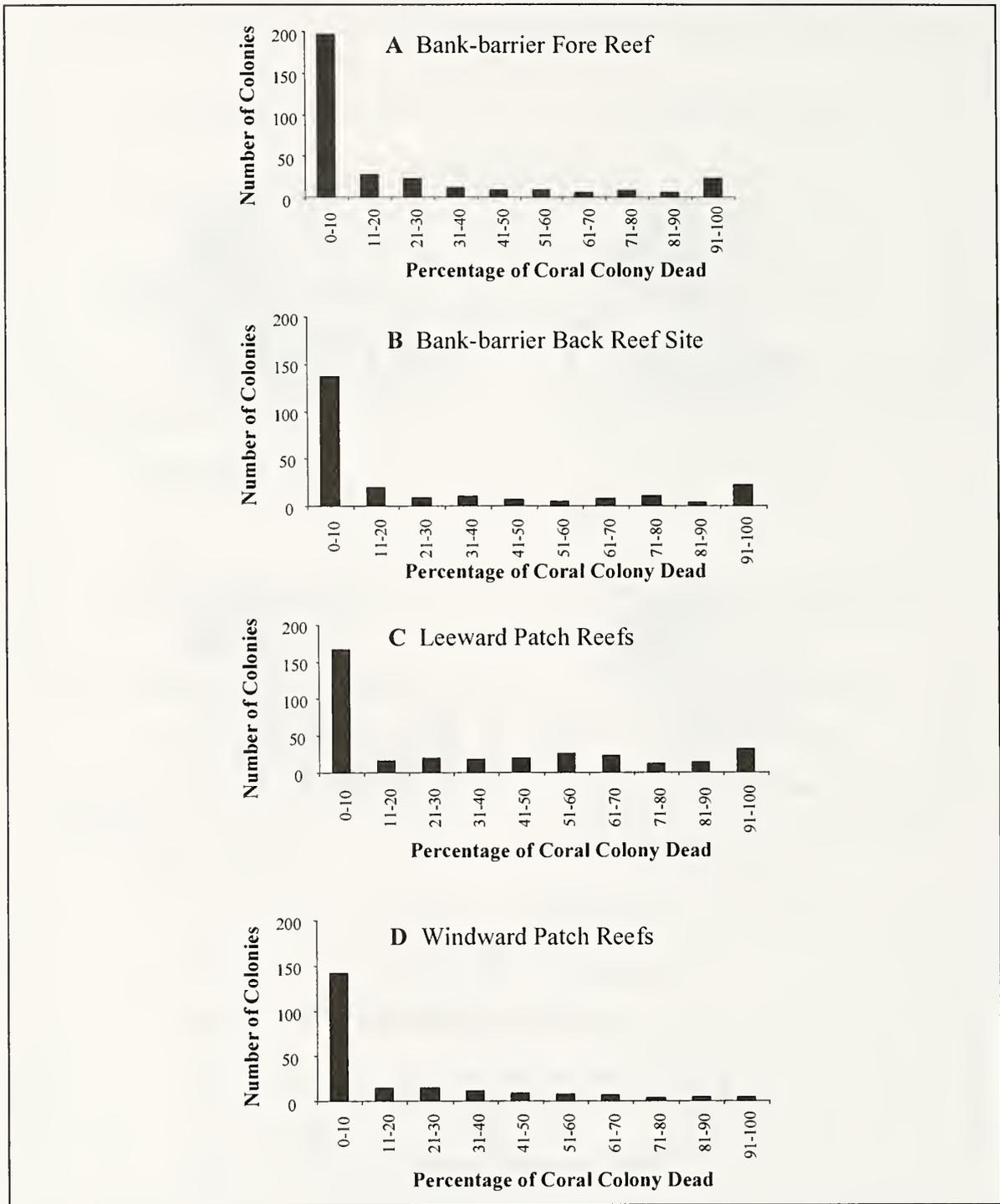
**Figure 2.** Species composition and mean relative abundance of the most abundant stony corals ( $\geq 10$  cm diameter) at (A) bank-barrier fore reef ( $n=319$ ), (B) bank-barrier back reef ( $n=232$ ), (C) leeward patch reefs ( $n=344$ ), (D) windward patch reefs ( $n=249$ ), off San Salvador Island, Bahamas. Other category = combined coral species, each with  $<5\%$  abundance of occurrence.



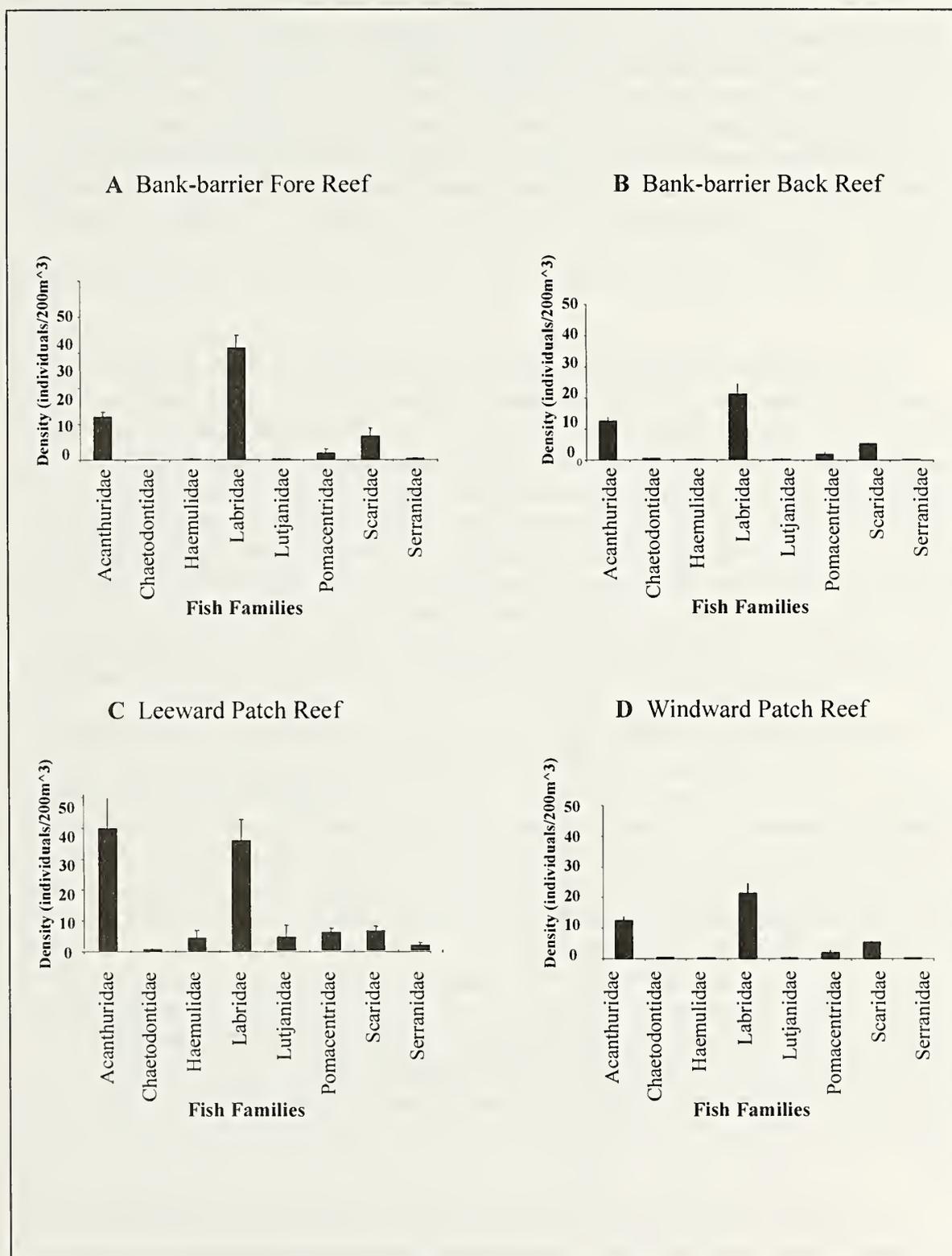
**Figure 3.** Species composition and mean relative abundance of all stony coral recruits ( $\leq 2$  cm diameter) at (A) bank-barrier fore reef (n=64), (B) bank-barrier back reef (n=52), (C) leeward patch reefs (n=96), (D) windward patch reefs (n=65), off San Salvador Island, Bahamas.



**Figure 4.** Size-frequency distribution as % of dominant stony corals ( $\geq 10$  cm diameter) at (A) bank-barrier fore reef, (B) bank-barrier back reef, (C) leeward patch reefs, (D) windward patch reefs, off San Salvador Island, Bahamas.



**Figure 5.** Frequency distribution of total (recent + old) partial colony mortality of all stony corals ( $\geq 10$  cm diameter) at (A) bank-barrier fore reef, (B) bank-barrier back reef, (C) leeward patch reefs, (D) windward patch reefs, off San Salvador Island, Bahamas.



**Figure 6.** Mean fish abundance (no. individuals/200 m<sup>3</sup>) at (A) bank-barrier fore reef, (B) bank-barrier back reef, (C) leeward patch reefs, (D) windward patch reefs, off San Salvador Island, Bahamas.

*Coral size.* The mean size of surveyed corals ( $\geq 10$  cm in diameter) ranged from 21 cm (at Lindsay Reef) to 85 cm (at Gaulin's A). However, mean colony sizes were remarkably similar among habitat types ranging from  $\sim 40$  cm on the leeward patch reefs to  $\sim 53$  cm in the bank-barrier back reef (Table 2). Size-frequency distributions of the major reef-building stony corals at each habitat type are shown in Figure 4. Wherever *Acorpora palmata* occurred a high proportion of colonies were very large ( $>200$  cm), emphasizing its importance as a major frame-builder. *Millepora complanata* also showed a relatively broad size range with colonies of  $<20$  cm to  $>200$  cm in diameter at the bank-barrier fore reef. While the *Montastraea annularis* complex only exceeded 100 cm in diameter on the leeward patch reefs, its massive growth forms provided substantial relief at most sites.

*Coral condition.* We observed evidence of coral disease in 2% or less of the censused colonies at most (8/11) sites (Table 2). Signs of disease were higher at one windward (3%) and two of the leeward patch reefs (4% at Telephone Pole; 8% at Snapshot). The coral diseases observed were black-band disease and yellow-blotch disease (both mainly on the *M. annularis* complex) and white-band disease (on *A. palmata*). The percentage of colonies that were bleached was low in all habitat types (from 1.3% in Fernandez Bay to 2.6% in Gaulin's fore and back reefs). Recent mortality was also very low at most sites (Table 2). The higher percentages of recent mortality at Snapshot (4%) and Telephone Pole (4.7%) were largely associated with the *Montastraea annularis* complex and *Porites porites*, respectively. The decline of *P. porites* was due in large part to overgrowth by the macroalga *Microdictyon marinum* and crustose coralline algae.

Although old partial-colony mortality (hereafter old mortality) exceeded 30% at three sites (Gaulin's back reef A, Snapshot, Telephone Pole), it largely was confined to the locally dominant coral species (Table 2). Total (recent + old) partial-colony mortality (hereafter total mortality) was lowest overall in the bank-barrier fore reef ( $\sim 20\%$ ) and highest at Snapshot ( $\sim 35\%$ , where total mortality of the *M. annularis* complex was nearly 55%), Gaulin's A ( $\sim 38\%$ , where *A. palmata* and the *M. annularis* complex together contributed 65% of the colonies measured) and Telephone Pole ( $\sim 42\%$ , with *P. porites* exhibiting about 44% total mortality).

Because recent mortality was never greater than 5% of the upper surface of coral colonies, we plotted frequency distributions of total mortality for the four reef habitat types (Fig. 5A-D). In all areas the vast majority of corals had  $<10\%$  total mortality. However, there was a clear signature ( $\sim 5\text{-}10\%$  of stony corals measured) of colonies showing  $>90\%$  total mortality at all sites except the windward patch reefs.

## Algae

Macroalgae dominated the benthic algae in three habitats (Gaulin's fore and back reefs, leeward patch reefs) and was the predominant component of the algal assemblages at all but three sites (Table 3). On leeward patch reefs, where their abundance was highest (about 50 to 70%), the fleshy green *Microdictyon marinum* was the dominant species. For example, at Telephone Pole Reef it commonly was completely overgrowing *Porites porites* resulting in total colony mortality. Elsewhere, brown algae, including *Dictyota*

*divaricata*, *D. bartayresii*, *Lobophora variegata*, *Padina sanctae-crucis*, *Turbinaria turbinata* and *Stypopodium zonale*, predominated. Green calcareous *Halimeda* spp. were extremely rare at all reef sites. Mean macroalgal height was generally lowest (~2 cm) at the windward sites (fore reef and patch reefs) but reached nearly 3 cm at Telephone Pole Reef. Mean macroalgal indices (absolute abundance of macroalgae x macroalgal height) varied from about 43 on the windward patch reefs to 150 on the leeward patches. Turf algae were more abundant at two sites (Gaulin's fore reef 1, French Bay 2) while crustose coralline algae were predominant at only a single site (French Bay 1).

## Fishes

Mean fish densities (Table 1) were highest at the leeward patch reefs, ranging from ~50 (Lindsay Reef) to 177 individuals/ 200 m<sup>3</sup> (Snapshot Reef). Fish abundances were relatively similar at the other sites varying between ~43 (windward patch reefs and bank-barrier back reef) and 53 individuals /200 m<sup>3</sup> (bank-barrier fore reef).

All reef habitats showed a dominance of wrasses (Labridae) and surgeonfishes (Acanthuridae) which collectively represented 76% of the total fish abundance on the leeward and windward patch reefs and over 90% on the bank-barrier reefs (Fig. 6A-D). Parrotfish (Scaridae) densities were low at all sites except Lindsay Reef where acanthurids were scarce (Table 4), yet the highest abundances of surgeonfishes occurred at the two other leeward patch reefs (Snapshot and Telephone Pole).

Groupers and other seabasses (Serranidae), snappers (Lutjanidae) and grunts (Haemulidae) were absent altogether on the Gaulin's back reef, present in extremely low densities on the Gaulin's fore reef and the windward patch reefs, and slightly more abundant on the leeward patch reefs (Fig. 6; Table 4). Considered together, these three families represented ~12% of total abundance in the eight censused fish families at Snapshot and Telephone Pole Reefs.

The size-frequency distributions for two major guilds (herbivores–parrotfishes, surgeonfishes, the yellowtail damselfish *Microspathodon chrysurus*; carnivores–grouper, snapper) at each of the four reef types are shown in Figure 7. Notably, no groupers or snappers were censused on the bank-barrier back-reef. Relatively low abundances of carnivores elsewhere make length comparisons among sites difficult. For all reef types, 40-50% of the herbivores fell within the 11-20 cm length category; generally, <10% were larger than 20 cm.

We found no relationship between herbivorous fish density and macroalgal index (Fig. 8). Grazing rates were nearly identical at Telephone Pole (mean=3.1 bites/minute, se =0.9, n=8) and Snapshot (3.5 bites/minute, se=1.1, n=12) Reefs, respectively. The grazing rate at Lindsay Reef was nearly three times those values (mean=11.1 bites/minute, se=0.4, n=5). Ninety percent of the grazing at Lindsay Reef was due to parrotfishes, whereas surgeonfishes were the dominant herbivores at the other two patch reefs.

## DISCUSSION

The stony corals at 11 evaluated sites of the reef system off San Salvador Island, Bahamas, are in reasonably good condition. Total (recent + old) partial-colony mortality

of upper surfaces (colonies  $\geq 10$  cm diameter) ranged from 15-42% and was particularly low ( $\sim 20\%$ ) on the bank-barrier fore reef. Interestingly, these estimates of old (and total) mortality were lowest ( $\sim 15\text{-}16.5\%$ ) at the sites on the fore reef and back reef that are nearest to land. These two sites experience higher wave surge and mixing than elsewhere on Gaulin's bank-barrier reef which perhaps provides a more favorable environment for coral survival. Similar to other Caribbean reefs, *Acropora palmata*, formerly a dominant in shallow windward zones, showed high ( $>50\%$ ) total colony mortality; nevertheless, we found evidence of its recruitment at several sites.

As found in other studies (e.g., Rogers et al., 1986), the species composition of the stony coral recruits did not reflect the major coral-reef builders at any site. Although 15% of the recruits at the leeward patch reefs were of the *Montastraea annularis* complex, which here represented 40% of the surveyed ( $\geq 10$  cm) corals, no recruits (and

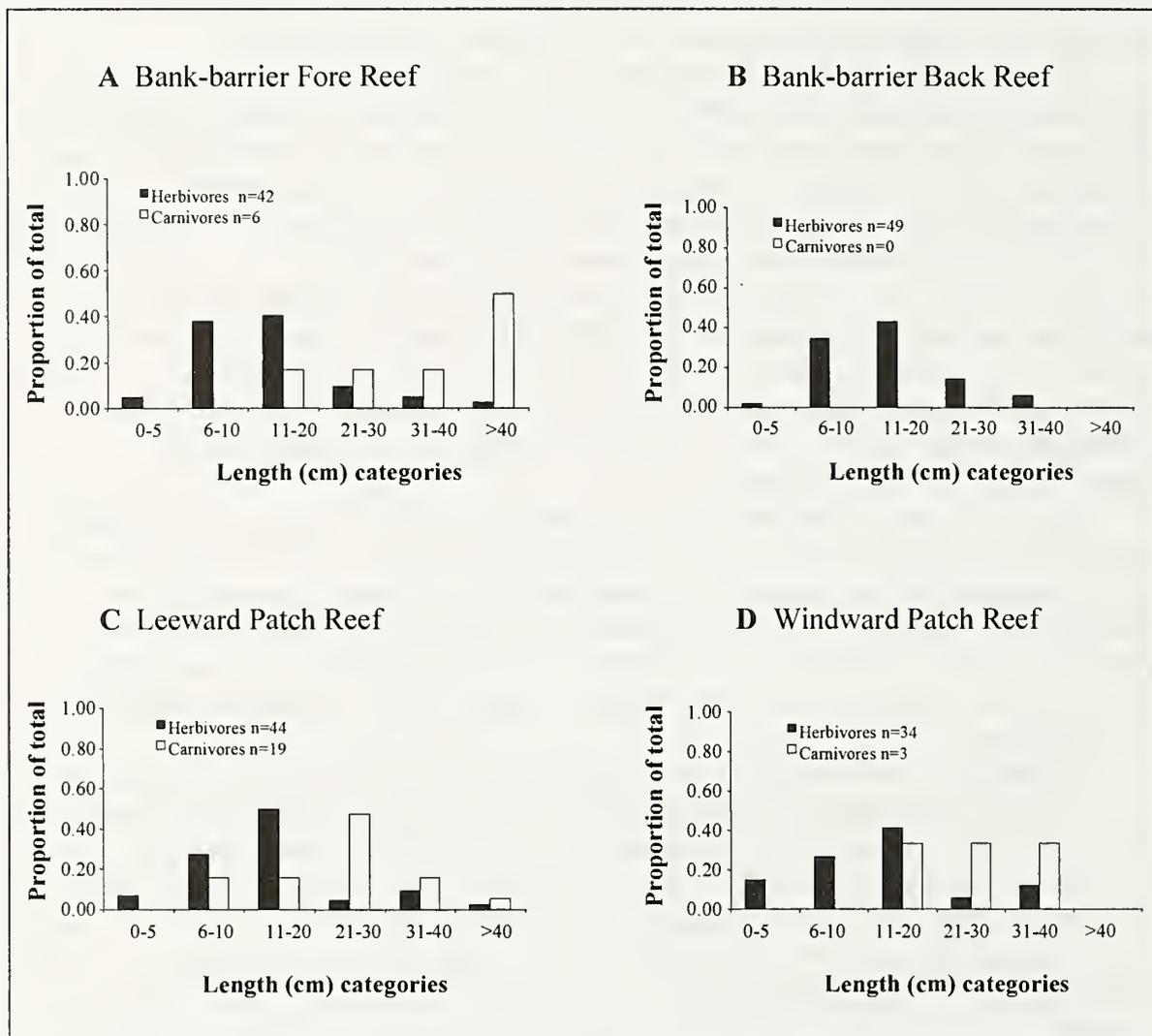
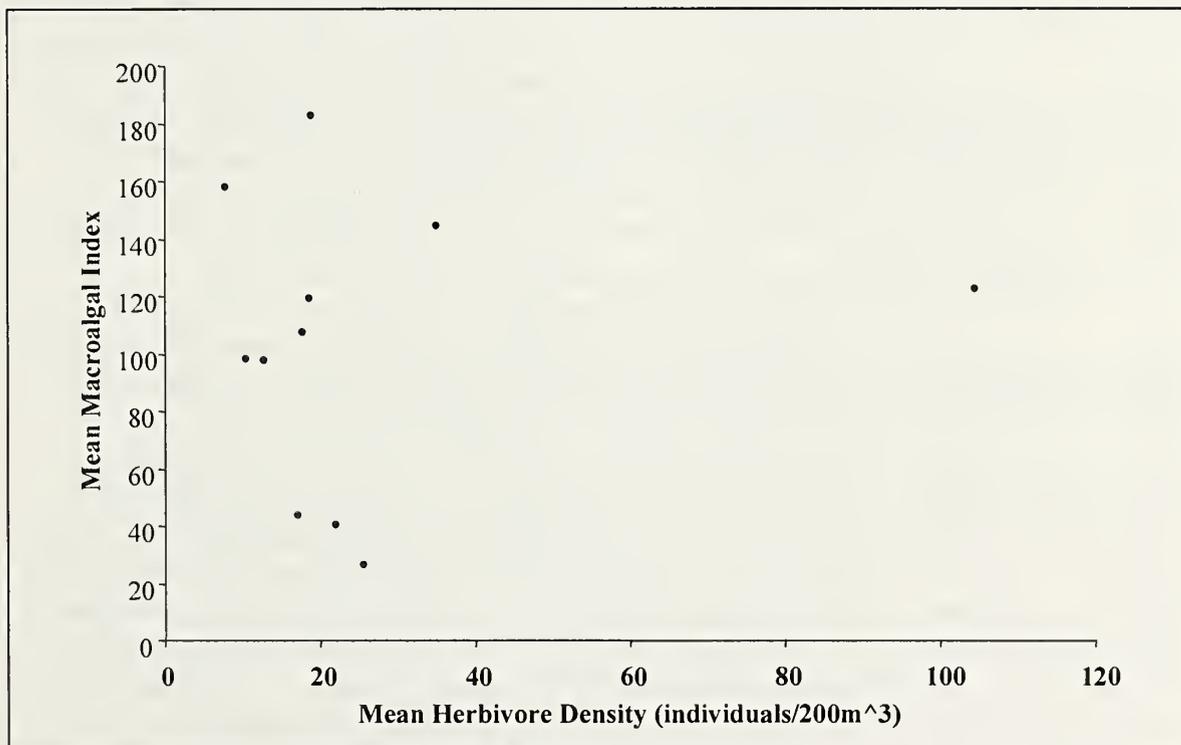


Figure 7. Size-frequency distribution of herbivores (all acanthurids and scarids, *Microspathodon chrysurus*) and carnivores (all lutjanids and serranids) at (A) bank-barrier fore reef, (B) bank-barrier back reef, (C) leeward patch reefs, (D) windward patch reefs, off San Salvador Island, Bahamas.

few  $\geq 10$  cm colonies) were found on the windward patch reefs. This species complex was a fairly important component (16% of  $\geq 10$  cm corals) in the bank-barrier back-reef region but it was represented by only 7% of the recruits.

Although receiving higher sediment loading from its proximity to the beach, recent and old mortality values were significantly lower at Lindsay Reef than on the other two leeward patch reefs. This apparent paradox may be an artifact of our sampling. Lindsay is a fairly “dead reef” overall; however, the live corals that remain are in good condition. The high percentages of partial mortality of the *Montastraea annularis* complex at Snapshot Reef, possibly from disease, and of *Porites porites* at Telephone Pole Reef from algal overgrowth, both occurring since Curran et al.’s (1994) assessment of these reefs, are causes for concern. The *M. annularis* complex has also suffered high rates of partial-colony mortality at other Caribbean sites. For example, two major warming events (1995, 1998) in Belize have resulted in massive coral bleaching with subsequent increased evidence of coral disease and mortality (McField, 1999; Peckol et al., this volume). Although the high incidence of diseased corals at Snapshot Reef (8% of censused colonies, 2.5% belonging to the *M. annularis* complex) in June 1998 predated the 1998 warming event that resulted in major coral bleaching worldwide (Wilkinson, 2000), San Salvador’s leeward patch reefs had previously bleached in 1995 (McGrath and Smith, 1999). However, these researchers noted that the major effect of the warming event was experienced by *Agaricia* spp. not *Montastraea*.

By June 1998, colonies of *Porites porites* were no longer expanding over the skeletons of *Acropora cervicornis* at Telephone Pole Reef. A more recent AGRRA survey during June 2000 showed that their condition had declined even further; partial-



**Figure 8.** Relationship between mean herbivore abundance (no. individuals/200 m<sup>3</sup>) and mean macroalgal index, by site in Sal Salvador.

colony mortality had risen from 44 % to >50% of the upper surfaces concomitant with an increase in macroalgal abundance from 57% to 88% (Peckol et al., unpublished). In Belize, Lewis (1986) demonstrated that macroalgae can directly overgrow and kill portions of *Porites astreoides* in herbivore exclusion treatments. Increases in macroalgae associated with declining coral cover have also been documented on other Caribbean reefs, including the San Blas Islands, Panamá (Ogden and Ogden 1994), Jamaica (Hughes 1994), and even areas remote from human impact (McClanahan et al, 1999).

High macroalgal abundances on San Salvador reefs may be related to the composition of the herbivorous fish guild where, in 1998, acanthurids (surgeonfishes) predominated at most sites. Lewis and Wainwright (1985) reported highest grazing rates for Belize in areas supporting higher parrotfish densities; similarly our highest grazing rates in Fernandez Bay were at Lindsay Reef where scarids were more common. Lewis (1985) also noted that parrotfish actively graze several genera of brown algae, including *Sargassum*, *Turbinaria*, and *Padina*, that are common at many of the San Salvador sites. Brown algae were not grazed at all by two surgeonfishes in Belize (Lewis 1985) and showed significant increases in percent cover and height (i.e., height is not a measure of abundance but rather of size/biomass) in response to experimental reduction in herbivory (Lewis, 1986). These findings may help to explain the high relative abundances of macroalgae off San Salvador reefs which are dominated by brown seaweeds at all but the leeward patch reefs. Lewis' (1985, 1986) results also may explain why there was no relationship between herbivore density (primarily acanthurids) and macroalgal index in the present study.

Absent or rare in most reef habitats, snappers, groupers and grunts were found in relatively high numbers only at Snapshot Reef in Fernandez Bay; this area may have a somewhat lower level of fishing activity than other sites. Hence, the San Salvador reefs are probably experiencing pressures from overfishing, but this conclusion cannot be stated with certainty because currently there are no comparable areas off San Salvador designated as "no-take" zones. In similar patch reefs off Belize, snapper and grouper densities and lengths are significantly greater within marine reserves and areas nearby (spillover effect) compared with sites not protected from fishing (Sedberry et al., 1992; Peckol et al., this volume).

Although we have documented partial mortalities of the major reef-building corals, *Acropora palmata* and the *Montastraea annularis* complex, that are relatively high in some habitats, the San Salvador reef system was fairly resistant to a large-scale disturbance from the passage of Hurricane Floyd. On September 13-14, 1999, this Category 4 storm passed within 20 to 30 nautical miles NE and N of the island with winds reaching 135 knots. Its greatest impact was felt on the leeward side of the island which experienced substantial coastal erosion and damage to buildings and infrastructure. However, the leeward patch reefs in Fernandez Bay showed little damage. We resurveyed the three leeward patch reefs in January 2000 and found insignificant change from June 1998 in the percent of recent or old partial mortality for the corals at these sites (compare Tables 2 and 5). As Smith and Buddemeier (1992) suggested for other reef systems, the San Salvador reefs displayed resilience in response to this large-scale natural disturbance.

Notwithstanding San Salvador's remote location, excellent marine water quality, and low human population density, the coral reefs surrounding the island are experiencing increased pressures from the combined effects of tourism and possibly

overfishing. Maintaining the integrity of San Salvador's coral reefs and adjacent marine ecosystems, and increasing the populations of fishes and invertebrates, will be important to the island's economic future. We strongly recommend the establishment of a marine reserve with an active management and regulations enforcement plan for all or a portion of San Salvador's reef system. Such designation should facilitate conservation of its critical fish and coral resources (Roberts, 1995; Nowlis and Roberts, 1997) and might contribute larval and adult fish to adjacent Bahamian insular and bank areas (Russ and Alcala, 1996; Stoner, 1996).

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Table 1. Site information for AGRRA stony coral, algae and fish surveys off San Salvador Island, Bahamas, June 1998.

Site name	Reef type	Latitude (°N)	Longitude (°W)	Survey date	Depth (m)	Benthic transects (#)	≥10 cm stony corals (#/10 m)	Fish cylinders (#)	Fish species (#) <sup>1</sup>	Fish density (#/200 m <sup>3</sup> )
Gaulins 1	bank barrier fore reef	24.16	74.46	June 03 98	1-4	10	10.5	6	21	55.5
Gaulins 2	bank barrier fore reef	24.17	74.48	June 04 98	1-4	13	10	8	19	45.5
Gaulins 3	bank barrier fore reef	24.14	74.45	June 07 98	1-4	10	10	6	15	57
Gaulins A	bank barrier back reef	24.15	74.47	June 04 98	2-8	10	8.5	7	24	46
Gaulins B	bank barrier back reef	24.16	74.48	June 04 98	2-8	10	11	7	26	57
Gaulins C	bank barrier back reef	24.14	74.46	June 07 98	2-8	10	9.5	6	18	33
Snapshot	leeward patch reef	24.04	74.54	June 01 98	4-7	11	9	6	34	177
Telephone Pole	leeward patch reef	24.03	74.54	June 02 98	6-7	10	12.5	6	28	74
Lindsay	leeward patch reef	24.01	74.53	June 10 98	3-6	10	10.5	6	20	49.5
French Bay 1	windward patch reef	23.95	74.54	June 06 98	3-5	13	9.5	8	24	47.5
French Bay 2	windward patch reef	23.95	74.55	June 09 98	3-5	10	10.5	6	21	35.5

<sup>1</sup>Fish species = all species of acanthurids, chaetodontids, haemulids, labrids, lutjanids, lomacentrids, scarids and serranids.

Table 2. Size and condition (mean ± standard error) of all stony corals (≥10 cm diameter), by site off San Salvador, Bahamas.

Site name	Stony corals		Partial-colony surface mortality (%)		Corals (%)			
	(#)	Diameter (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased
Gaulins 1	103	51.5 ± 7.0	2.5 ± 0.5	19.0 ± 2.8	21.5 ± 2.9	5.0 ± 3.1	6	0
Gaulins 2	131	55.5 ± 7.0	1.0 ± 0.6	24.0 ± 2.8	25.0 ± 2.8	6.5 ± 3.4	1	2
Gaulins 3	85	25.0 ± 5.0	1.5 ± 1.2	15.0 ± 3.0	16.5 ± 3.2	6.0 ± 2.5	1	1
Gaulins A	79	85.0 ± 14.0	1.5 ± 0.6	36.0 ± 4.7	37.5 ± 4.7	30.0 ± 8.4	4	1
Gaulins B	91	46.0 ± 6.0	0.5 ± 0.1	19.0 ± 3.0	20.0 ± 3.0	2.5 ± 1.7	4	1
Gaulins C	62	28.0 ± 4.0	<0.5 ± 0.1.5	15.0 ± 2.9	15.0 ± 3.0	0	0	2
Snapshot	93	61.0 ± 6.5	4.0 ± 1.3	31.0 ± 3.6	35.0 ± 3.6	5.0 ± 2.3	2	8
Telephone Pole	139	39.5 ± 4.0	4.5 ± 1.0	37.0 ± 3.1	41.5 ± 3.1	10.5 ± 3.2	4	4
Lindsay	112	21.0 ± 2.0	<0.5 ± 0.15	18.0 ± 2.6	18.0 ± 2.6	0	1	1
French Bay 1	150	63.0 ± 11.0	1.0 ± 0.5	28.5 ± 3.3	29.5 ± 3.3	28.5 ± 8.5	3	1
French Bay 2	99	36.5 ± 4.5	<0.5 ± 0.1	22.5 ± 2.8	22.5 ± 2.8	2.0 ± 1.3	0	3

Table 3. Algal characteristics and stony coral recruit abundance (mean  $\pm$  standard error) by site off San Salvador Island, Bahamas.

Site name	Quadrats (#)	Absolute abundance (%)			Crustose coralline algae	Macroalgal height (cm)	Macroalgal index <sup>1</sup>	Recruits (#/0625 m <sup>2</sup> )
		Macroalgae	Turf algae	Macroalgae				
Gaulins 1	70	30.5 $\pm$ 3.2	41.5 $\pm$ 2.4	28.0 $\pm$ 3.2	1.5 $\pm$ 0.2	26 $\pm$ 2.9	0.3 $\pm$ 0.1	
Gaulins 2	84	51.5 $\pm$ 3.3	30.5 $\pm$ 3.0	18.0 $\pm$ 2.5	2.0 $\pm$ 0.2	107 $\pm$ 9.6	0.4 $\pm$ 0.1	
Gaulins 3	54	51.0 $\pm$ 3.7	27.0 $\pm$ 3.2	21.5 $\pm$ 3.3	2.0 $\pm$ 0.1	119 $\pm$ 11.9	0.4 $\pm$ 0.1	
Gaulins A	67	37.0 $\pm$ 3.8	33.5 $\pm$ 2.8	29.0 $\pm$ 2.8	2.0 $\pm$ 0.2	98 $\pm$ 11.5	0.4 $\pm$ 0.1	
Gaulins B	45	49.0 $\pm$ 4.3	21.5 $\pm$ 2.8	30.0 $\pm$ 3.4	2.0 $\pm$ 0.1	98 $\pm$ 10.2	0.3 $\pm$ 0.0	
Gaulins C	49	57.5 $\pm$ 4.0	19.0 $\pm$ 3.3	23.5 $\pm$ 3.4	2.5 $\pm$ 0.1	158 $\pm$ 17.0	0.4 $\pm$ 0.1	
Snapshot	68	50.0 $\pm$ 4.0	27.0 $\pm$ 3.7	23.0 $\pm$ 3.1	2 $\pm$ 0.2	123 $\pm$ 15.0	0.2 $\pm$ 0.1	
Telephone Pole	48	56.5 $\pm$ 3.2	21.5 $\pm$ 2.0	22.0 $\pm$ 2.5	2.5 $\pm$ 0.1	144 $\pm$ 7.7	0.4 $\pm$ 0.1	
Lindsay	75	70.5 $\pm$ 4.1	24.5 $\pm$ 3.5	5.0 $\pm$ 2.7	2.5 $\pm$ 0.1	183 $\pm$ 14.3	0.7 $\pm$ 0.1	
French Bay 1	90	14.5 $\pm$ 2.2	35.5 $\pm$ 2.2	50.0 $\pm$ 2.7	2.5 $\pm$ 0.4	40 $\pm$ 7.2	0.5 $\pm$ 0.2	
French Bay 2	37	19.0 $\pm$ 3.8	53.5 $\pm$ 3.6	27.5 $\pm$ 3.8	2.0 $\pm$ 0.4	44 $\pm$ 10.8	0.4 $\pm$ 0.1	

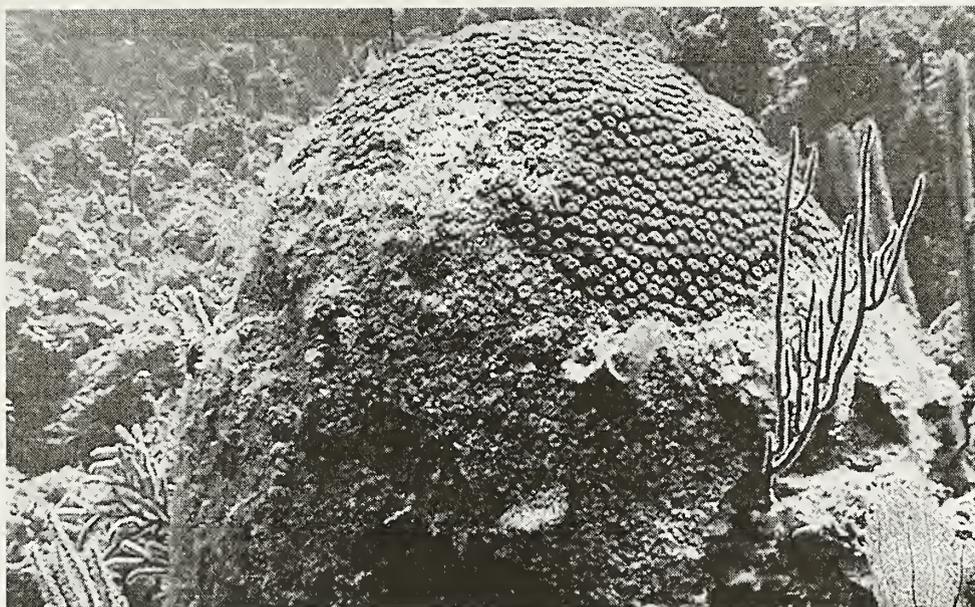
<sup>1</sup>Macroalgal index = absolute macroalgal abundance x macroalgal height.

Table 4. Fish abundance ( $\pm$  standard error) by site off San Salvador Island, Bahamas.

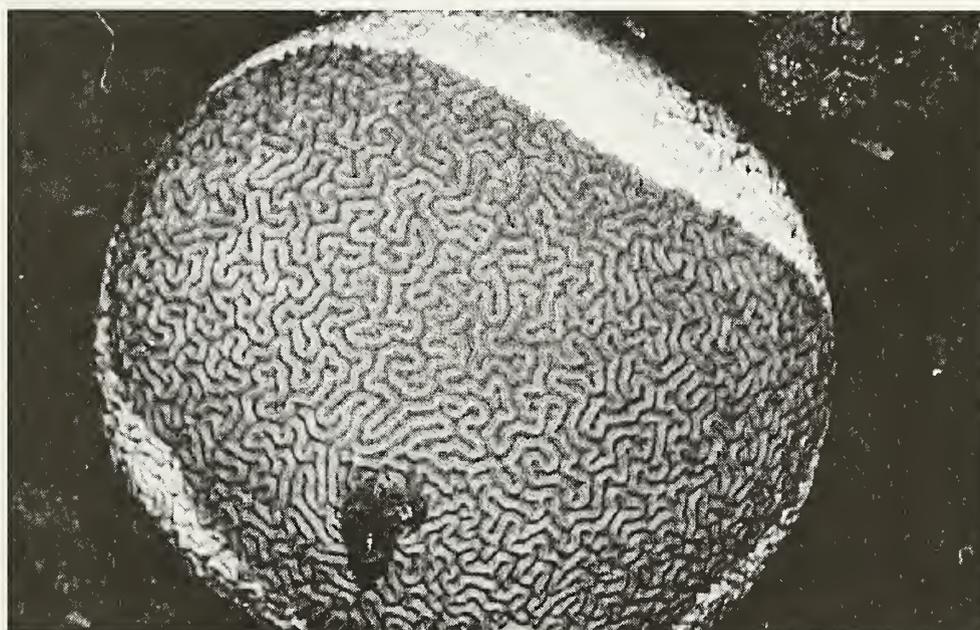
Site name	Reef type	Herbivores (#/200 m <sup>2</sup> )			Carnivores (#/200 m <sup>2</sup> )		
		Acanthuridae	Scaridae	Haemulidae	Lutjanidae	Serranidae	
Gaulins 1	bank-barrier fore reef	10.4 $\pm$ 2.2	10.8 $\pm$ 7.2	0.2 $\pm$ 0.2	0.2 $\pm$ 0.2	0.6 $\pm$ 0.4	
Gaulins 2	bank-barrier fore reef	11.0 $\pm$ 4.8	5.6 $\pm$ 1.6	0.3 $\pm$ 0.3	0.0 $\pm$ 0.0	0.4 $\pm$ 0.2	
Gaulins 3	bank-barrier fore reef	14.8 $\pm$ 6.7	3.5 $\pm$ 0.6	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	
Gaulins A	bank-barrier back reef	5.6 $\pm$ 1.1	4.1 $\pm$ 0.8	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	
Gaulins B	bank-barrier back reef	6.6 $\pm$ 1.3	2.4 $\pm$ 1.1	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	
Gaulins C	bank-barrier back reef	5.0 $\pm$ 0.4	1.5 $\pm$ 0.5	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	
Snapshot	leeward patch reef	91.8 $\pm$ 21.5	4.2 $\pm$ 1.2	8.8 $\pm$ 4.4	12.0 $\pm$ 6.0	1.2 $\pm$ 0.3	
Telephone Pole	leeward patch reef	23.3 $\pm$ 16.9	5.7 $\pm$ 1.3	3.5 $\pm$ 1.3	0.8 $\pm$ 0.7	3.3 $\pm$ 0.8	
Lindsay	leeward patch reef	5.5 $\pm$ 1.0	9.7 $\pm$ 1.3	0.3 $\pm$ 0.2	0.8 $\pm$ 0.7	0.7 $\pm$ 0.2	
French Bay 1	windward patch reef	13.8 $\pm$ 3.8	5.3 $\pm$ 1.9	0.25 $\pm$ 0.2	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	
French Bay 2	windward patch reef	11.2 $\pm$ 2.4	4.8 $\pm$ 1.3	0.0 $\pm$ 0.0	0.33 $\pm$ 0.0	0.3 $\pm$ 0.2	

Table 5. Partial-colony mortality (recent and old) of all stony corals ( $\geq 10$  cm diameter) by site at the leeward patch reefs, off San Salvador, Bahamas, in January 2000.

Site name	Stony corals (#)	Partial-colony surface mortality (% $\pm$ standard error)	
		Recent	Old
Snapshot Reef	160	0.8 $\pm$ 0.4	25.6 $\pm$ 2.6
Telephone Pole Reef	220	3.2 $\pm$ 0.8	35.1 $\pm$ 2.3
Lindsay Reef	200	1.3 $\pm$ 0.5	20.5 $\pm$ 2.1



**Plate 5A.** Partial mortality, as seen in a colony of *Montastraea cavernosa*, is a well-established indicator of reef condition. In the AGRRA benthos protocol, partial mortality is distinguished as either “recent” or “old.” The importance of this distinction is that “recent mortality” approximates tissue losses occurring within the previous days to months, whereas “old mortality” represents an integration of disturbances occurring over much longer time scales. (Photo Kenneth W. Marks)



**Plate 5B.** The percentages of “recent mortality” and of “old mortality” are assessed from above the colony in planar view and at an angle that is parallel to its axis of growth. (Photo Kenneth W. Marks)

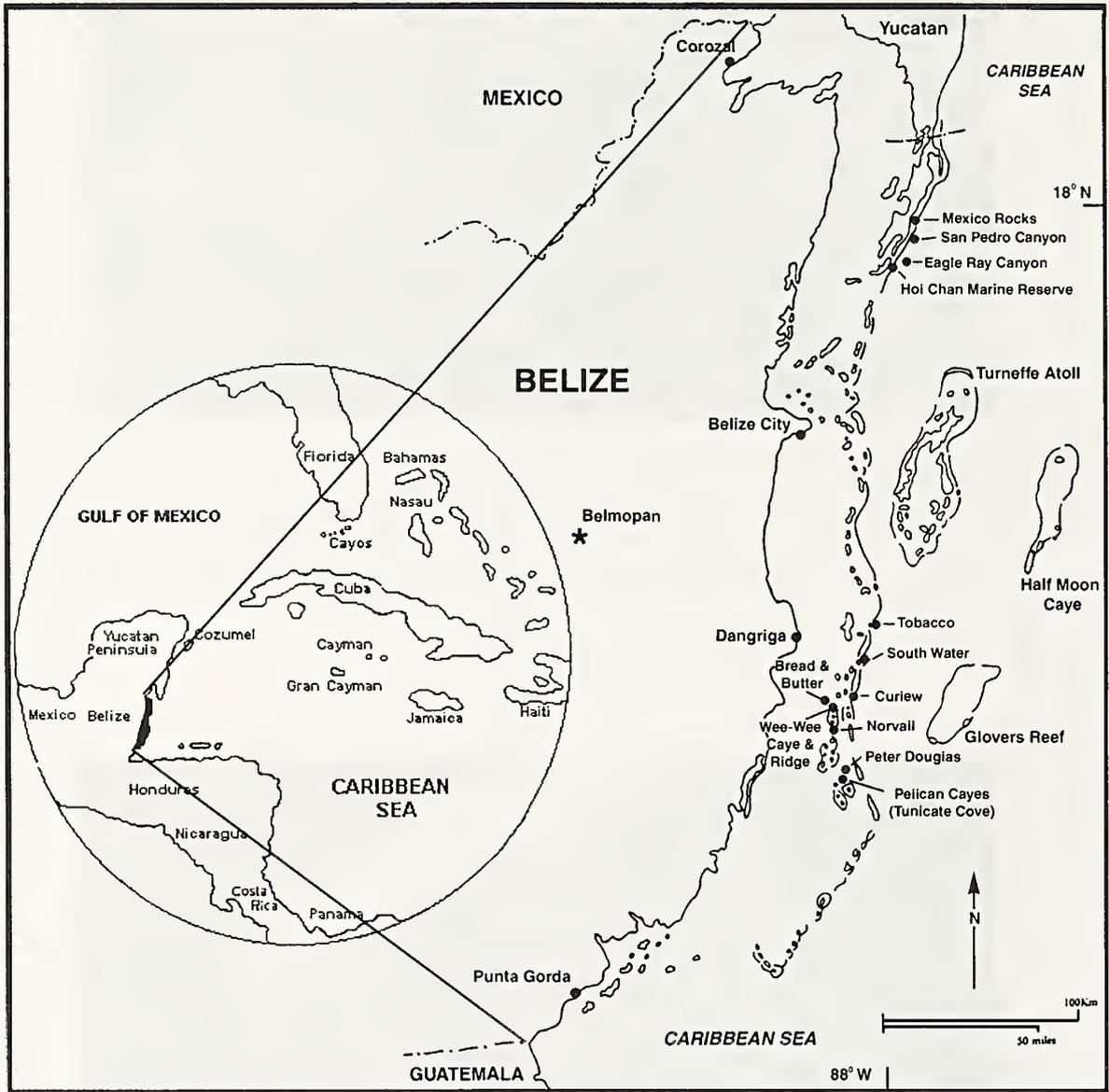


Figure 1. AGRRA survey sites along the Belize barrier reef system.

# ASSESSMENT OF SELECTED REEF SITES IN NORTHERN AND SOUTH-CENTRAL BELIZE, INCLUDING RECOVERY FROM BLEACHING AND HURRICANE DISTURBANCES (STONY CORALS, ALGAE AND FISH)

BY

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## ABSTRACT

The condition of coral, algal, and fish populations in fore reefs, patch reefs, and coral reef ridges was investigated at 13 sites along the northern and south-central Belize barrier reef during May 1999, documenting effects of the 1998 warming episode and Hurricane Mitch. We found high percentages of partial, or even complete, colony mortality of major reef-builders (*Acropora palmata*, the *Montastraea annularis* species complex and *Agaricia tenuifolia*) that were rarely censused as recruits. *A. tenuifolia*, formerly a space-dominant coral in reef ridges, had incurred nearly 100% mortality after bleaching. Nearly 45% of the *M. annularis* complex was still discolored (50% had been bleached in January 1999) on some south-central patch reefs where the total (recent + old) partial mortality exceeded 60% of colony surfaces. Although turf algae dominated patch reefs and coral reef ridges, macroalgae were quite prevalent representing >30% cover at six sites. Parrotfish densities exceeded surgeonfishes at most sites (11/13). Consistent patterns of lower partial-colony mortality of stony corals and greater fish densities and sizes near and within the Hol Chan Marine Reserve highlight the ecological benefits of protected areas for the maintenance of reef corals and attendant fish populations.

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## INTRODUCTION

Over the past two decades, there has been widespread deterioration of coral reefs worldwide (Wilkinson, 2000). Following the mass mortality in 1983 of a “keystone” herbivore, the sea urchin *Diadema antillarum*, Caribbean reefs emerged as an area of particular concern (see Ginsburg, 1994) when macroalgal abundances increased at many reefs (Lessios, 1988; Hughes et al., 1999; but see Lapointe, 1997, 1999). The Belize barrier reef system largely was spared from the negative effects of the disappearance of *D. antillarum* because, prior to the mass mortality event, its densities were extremely low compared with those on reefs experiencing concurrent pressures of overfishing (Hay, 1984). Until 1995, the Belize reefs had also escaped the mass coral bleaching events reported in many other areas of the Caribbean (Macintyre and Aronson, 1997; McField, 1999). In addition, Belize has a very low human population density (about 230,000 or about nine people/km<sup>2</sup>). The central and southern regions of the Belize barrier reef are distant from the effects of terrestrial development and, at least until recently, fishing has been primarily at the subsistence level. The Belize barrier reef ecosystem thus must have served as a large and important regional source of larvae and juvenile corals, other invertebrates, and fishes (Cortés, 1997). The coupling of low incidence of large-scale natural disturbances and minimal anthropogenic effects conserved this reef system in a nearly pristine state with a unique array of reef types and luxuriant coral communities (Perkins and Carr, 1985). However, the hurricanes in the ‘60s and 70’s that Stoddart (e.g., 1974) made famous and the effects of white-band disease, all occurring before the 1995 bleaching, demonstrate that natural perturbations pre-1995 are not without their effects in Belize.

The Belize barrier reef complex is the largest continuous reef system in the western North Atlantic-Caribbean Province extending a distance of 250 km south from the northern end of Ambergris Caye. It is one of the best-studied coral-reef areas in the western hemisphere (Cortés, 1997), particularly since the establishment of the Smithsonian Institution’s field station on Carrie Bow Caye in 1972 (Rützler and Macintyre, 1982; Macintyre and Aronson, 1997). A major NNE-trending fault system is clearly reflected in the alignment of the coastline, barrier reef, and three oceanic atolls. The reef is fringing at its northernmost end, but as it extends southward the system becomes a nearly continuous barrier with well-developed fore reefs, reef crests, and extensive back-reef (lagoonal) areas. In the shallower portions of the fore-reef zone, interdigitating coral buttresses and sandy troughs form a sometimes massive spur-and-groove system. The reef crest is a high-energy zone consisting of a shallow rampart built of coral rubble and dominated by the coral *Acropora palmata* (James and Ginsburg, 1979). The back-reef area has a large number of cayes (cays, or small islands), many with associated patch reefs. There is a complex network of steep-sided, rhomboidal-shaped shoals and reefs (herein referred to as coral reef ridges) in the south-central lagoon, which originated in fault patterns enhanced by subsequent Pleistocene karst solution and differential carbonate deposition (Macintyre and Aronson, 1997; Macintyre et al., 2000). Coral reef ridges previously supported spectacular coral development dominated by the staghorn *Acropora cervicornis* at depths of 3-8 m. Following mass mortality in the 1980s of *A. cervicornis* from white-band disease, the ridges were rapidly colonized by *Agaricia* spp., primarily *A. tenuifolia* (Aronson and Precht, 1997).

The biologically rich Belize barrier reef ecosystem contributes to a multimillion-dollar fishing industry and serves as a prime attraction for sport fishermen and scuba/snorkel divers. Tourism in Belize (see <http://www.belize tourism.org/arrival.html>) has been expanding rapidly from less than 100,000 visitors in 1985 to over 326,000 in 2000 and growing. Tourism generates over 26% of the Belizean gross national product (Higinio and Munt, 1993). Ambergris Caye has borne the brunt of the effects of coastal tourism including degradation of the reef, dwindling fish stocks, and surface runoff. In 1987, the Government of Belize established the Hol Chan Marine Reserve off Ambergris Caye in an effort to conserve a small but complete portion of this reef ecosystem including the coral reef, lagoon, and mangrove habitats (Carter et al., 1994). Modeled in large part after the Australian Great Barrier Reef management plan, this reserve is zoned with a multi-use management scheme that has achieved some success. Although the Hol Chan Marine Reserve does not encompass an expansive reef tract, it was designed to serve as an impetus for the creation of additional reserves in the future (Carter et al., 1994).

Within the past decade the Belize barrier reef has undergone a noticeable decline (Kramer et al., 2000), particularly as fishing pressures have reduced stocks (Carter et al. 1994). Several major disturbances, including warming events during 1995/96 (Burke et al., 1996; McField, 1999) and 1998 (Mumby, 1999; Aronson et al., 2000), and a near direct hit by Hurricane Mitch in autumn 1998, have accelerated its degradation. The 1998 bleaching event began in September following a month of calm weather and increasing water temperatures. The eye of Hurricane Mitch, a category 5 storm, passed ~200 km SE of Glovers Reef in late October 1998 (Mumby, 1999). The subsequent demise of *Agaricia tenuifolia* on the coral reef ridges in the south-central region, a phenomenon unprecedented in recent geologic history, has been particularly well described by Aronson et al. (2000).

Given the large size and the diversity of reef types present on the Belize barrier, there is a great need for larger-scale monitoring to document the short- and long-term effects of these widespread disturbances on this very important reef system. In addition, understanding the effects of marine reserves on the health and conservation of coral and fish populations is critical for long-term management decisions. Hence, we investigated the condition of stony coral, algae and fish populations at 13 sites along the northern and south-central regions of the Belize barrier reef, including fore reefs, patch reefs and coral reef ridges. The timing (May 1999) of our survey enabled us to document the effects of the major 1998 warming event and Hurricane Mitch. The Hol Chan Marine Reserve was included in our surveys to provide information useful for management decisions regarding this and other reserves planned for the Belize barrier reef.

## METHODS

The 13 sites that we surveyed during May 1999 were selected as being representative of the reef types and conditions in the northern and south-central regions of the Belize Barrier Reef (Fig. 1). The four northern sites near Ambergris Caye were also strategically chosen to compare areas within and outside the Hol Chan Marine Reserve.

Five fore-reefs were censused. San Pedro Canyon, located south of Mexico Rocks off Ambergris Caye at 12-15 m depth, is characterized by a low-relief (~2 m) spur-and-groove topography with small colonies of the *Montastraea annularis* species complex and numerous gorgonian corals. Eagle Ray Canyon (depth 12-18 m), which lies north of the Hol Chan Channel but is still within the reserve, has a well-developed spur-and-groove system. It supports a high diversity of corals in good condition dominated by large colonies of the *M. annularis* complex and *Agaricia tenuifolia*. Tobacco Reef, in south-central Belize, has a low-relief spur-and-groove formation at depths of about 12-13 m with large, healthy colonies of the *M. annularis* complex. *A. tenuifolia* was overgrowing dead skeletons of *Millepora complanata* on the fore-reef spurs. (In May 1999, the large coral-rubble ridge that runs a considerable distance from north of Tobacco Caye southward to near South Water Caye was covered with large fragments of what appeared to be recently dead coral colonies that were reported by locals to have been deposited by Hurricane Mitch.) The fore reef off South Water Caye (up to 14 m depth) has a high-relief (3-5 m) spur-and-groove development with much *Siderastrea siderea* and some large colonies of the *M. annularis* complex, *Millepora complanata* and *Agaricia* spp. were abundant on the spurs. Curlew Bank fore reef, located just south of Carrie Bow Caye, is characterized by a low-relief spur-and-groove formation with colonies of the *M. annularis* complex, *S. siderea*, *Diploria* spp., and *A. palmata* (both living and dead colonies).

Five surveys were in patch reefs. Mexico Rocks, seaward of Ambergris Caye, is an irregularly shaped complex of approximately 100 patch reefs at about 4 m depth. These patch reefs are dominated by large stands of the *M. annularis* complex (Burke et al., 1998), separated by sand and seagrass. Although this reef exhibited low coral diversity, live coral cover values were average for this reef type. Near the patch reef in the Hol Chan Marine Reserve, which is dominated by *Acropora palmata* (both living and dead colonies) and the *M. annularis* complex, a 10-m deep channel (Hol Chan Channel) runs through the reef crest and into the back-reef region. The three patch reefs in the south-central region were at depths of about 2-5 m on the windward sides of Wee-Wee, Bread and Butter, and Norvall Cayes, where the *M. annularis* complex is the dominant coral species. Most colonies of *A. palmata* were "long dead" and were covered with turf algae.

The three surveys in coral reef ridges were conducted at depths of 2-15 m off Wee-Wee Caye, Peter Douglas Caye, and an area locally known as "Tunicate Cove" within the Pelican Cayes. At these reefs most colonies of the dominant coral, *A. tenuifolia*, had died since the beginning of the 1998 warming event.

Coral and algal populations were assayed by six-seven divers on each dive. Atlantic and Gulf Rapid Reef Assessment (AGRRA) Version 1 benthos protocols (see Appendix One, this volume) were used with the following modifications: stony corals  $\geq 10$  cm in diameter were included in the surveys; coral diameter and height were measured to the nearest cm for smaller corals (10-25 cm in diameter) and to the nearest 5 cm for larger (>25 cm) colonies. The *Montastraea annularis* complex was treated as a single species. Sediment deposits in the algal quadrats were removed by hand before estimating the abundance of crustose coralline algae. As space occupied by turf algae growing on live crustose corallines was allocated to both functional groups, total values for absolute abundance at some sites are in excess of 100%. *Diadema antillarum*, being

rare, was not counted. Training sessions were conducted with all divers [most of whom were already well-trained from the 1998 San Salvador Island, Bahamas AGRRA assessment (Peckol et al. this volume)] censusing “practice” transects on patch reefs. Species identifications, percent cover estimates, and coral disease and bleaching assessments were compared to ensure sampling consistency. We used Humann’s (1993) reef coral guide for coral species identifications; Littler and Littler (2000) was later consulted for macroalgae.

A stationary visual census technique (Bohnsack and Bannerot 1986) was employed by two divers to survey the fish populations. All sampling occurred between 10:00 a.m. and 3:00 p.m. At each sampling point, all species belonging to eight families (Acanthuridae, Chaetodontidae, Haemulidae, Labridae, Lutjanidae, Pomacentridae, Scaridae, Serranidae) observed in five minutes within a 7.5 m radius cylinder were recorded. Each census was begun three minutes after laying a measuring tape on the substratum by counting all individuals of all species observed in the pre-set radius within the initial field of view. New sectors of fields of view were then scanned by rotating in one direction. Abundances of species moving in schools were taken when first observed in the sampling cylinder (it was important to count fishes moving in schools immediately because they were unlikely to remain in the sampling area). When very large schools were present it was sometimes necessary to estimate numbers in 10s or 50s. Fish lengths in cm were estimated using a T-shaped tool marked every 5 cm to help avoid underwater magnification problems (Bohnsack and Bannerot, 1986). We recorded the number of individuals, plus the minimum, maximum, and mean estimated lengths for each of the eight fish families. We used Humann’s (1994) reef fish guide for species identifications.

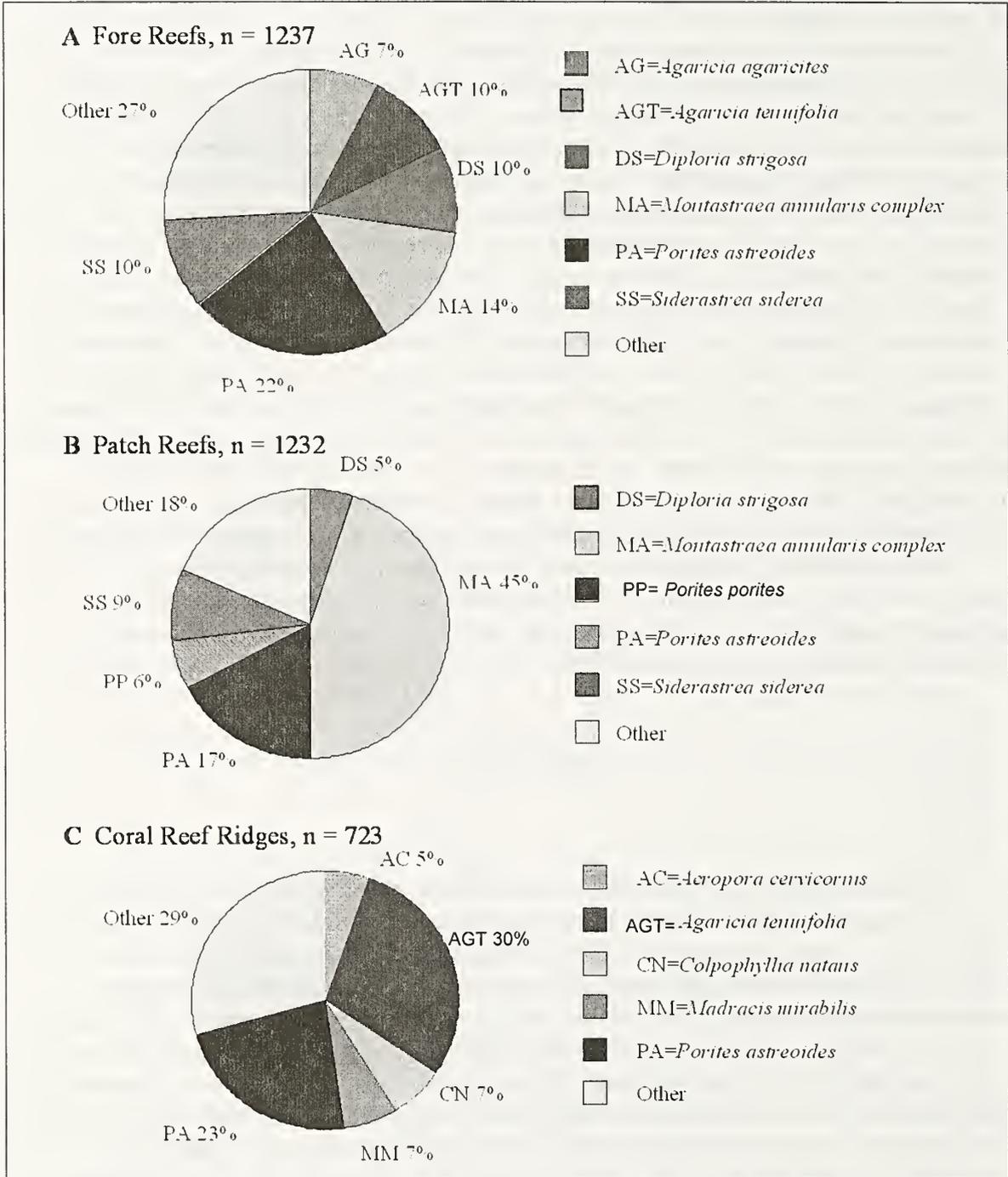
Herbivorous fish grazing rates were measured by two divers at 12 sites (all but the Eagle Ray fore reef), following the AGRRA methodology given in Appendix One. Grazing rate measurements were all made between 10:00 a.m. and 2:00 p.m. during the peak time for grazing activity (Lewis, 1986).

## RESULTS

### Stony Corals

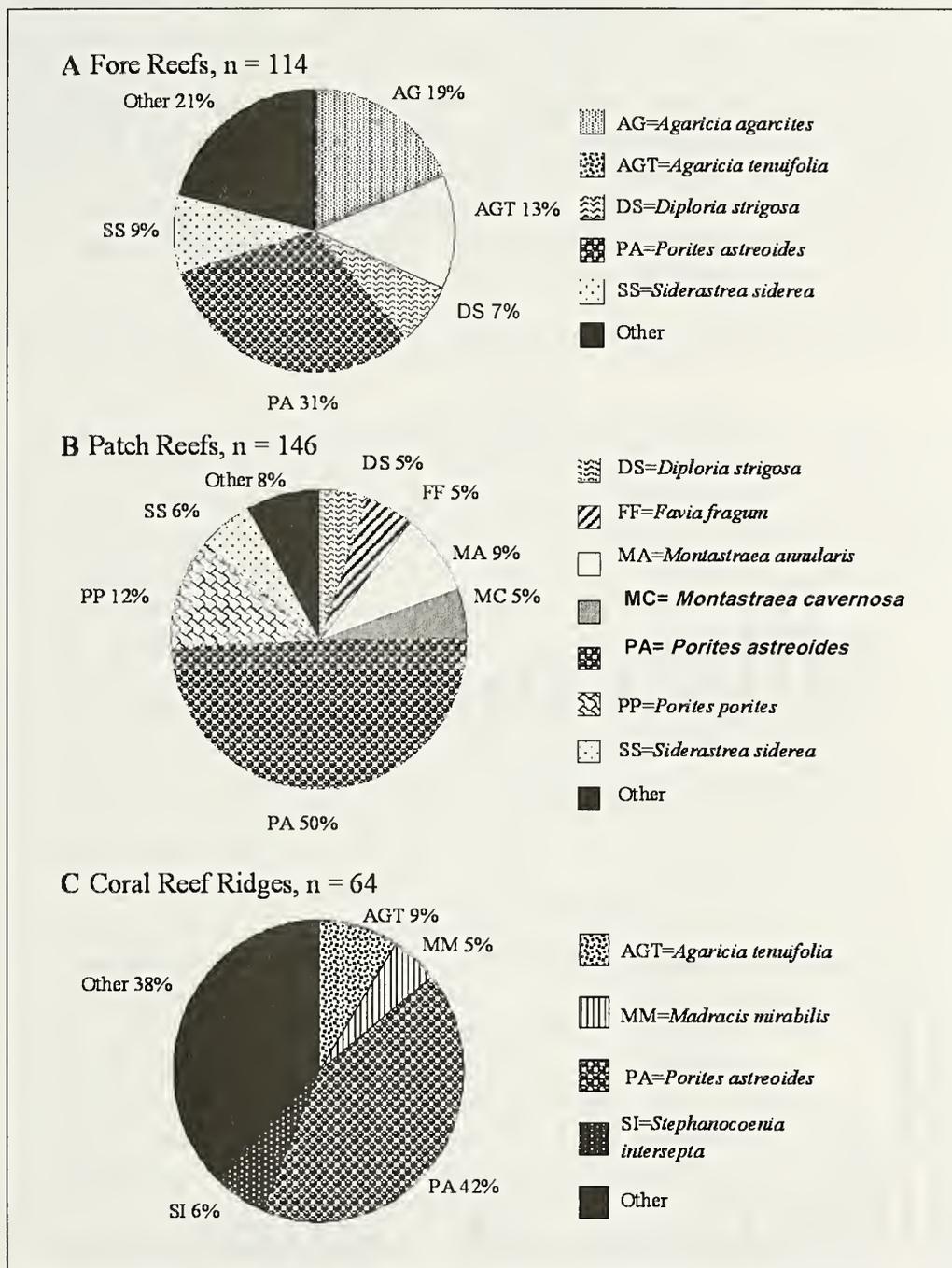
*Species composition.* At each site we censused between 12-18 10-m transects, each of which generally had about 15 stony corals that were at least 10 cm in diameter (Table 1). At most sites we measured >250 coral colonies. Coral species abundance patterns varied greatly among the three reef types (Fig. 2A-C). Numerically, the most abundant coral species censused in the fore-reef spurs were *Porites astreoides* (22% of total) and the *Montastraea annularis* species complex (14% of total). *Acropora palmata* occurred here in relatively low abundance (<5% of total) but showed high total (recent and old) partial mortality of colony surfaces ranging from 25% (South Water Caye) to 100% (Tobacco Caye) and averaging >55%. Due to high wave-energy conditions, we were unable to sample the reef crest where *A. palmata* dominated (some colonies were alive). The *M. annularis* complex was clearly dominant in patch reefs (45% of total). Collectively, the mound corals, *Siderastrea siderea*, *Diploria strigosa*, and the *M. annularis* complex represented ~60% of the corals sampled in patch reefs. Because their

colony sizes are generally larger, the numerically less abundant *D. strigosa* contributed more than *S. siderea* to reef habitat. Mostly dead *Agaricia tenuifolia* (30%) and *P. astreoides* (23%) dominated the south-central coral reef ridges. The *M. annularis* complex was a minor component, and species richness was highest in this unique reef type (30 scleractinian species sampled along transects compared with ~24 species on patch reefs and fore reefs).



**Figure 2.** Species composition and mean relative abundance of the most abundant stony corals ( $\geq 10$  cm diameter) at (A) fore reefs; (B) patch reefs; (C) coral ridge reefs along the Belize barrier reef system. Other category = combined coral species, each with  $< 5\%$  abundance of occurrence.

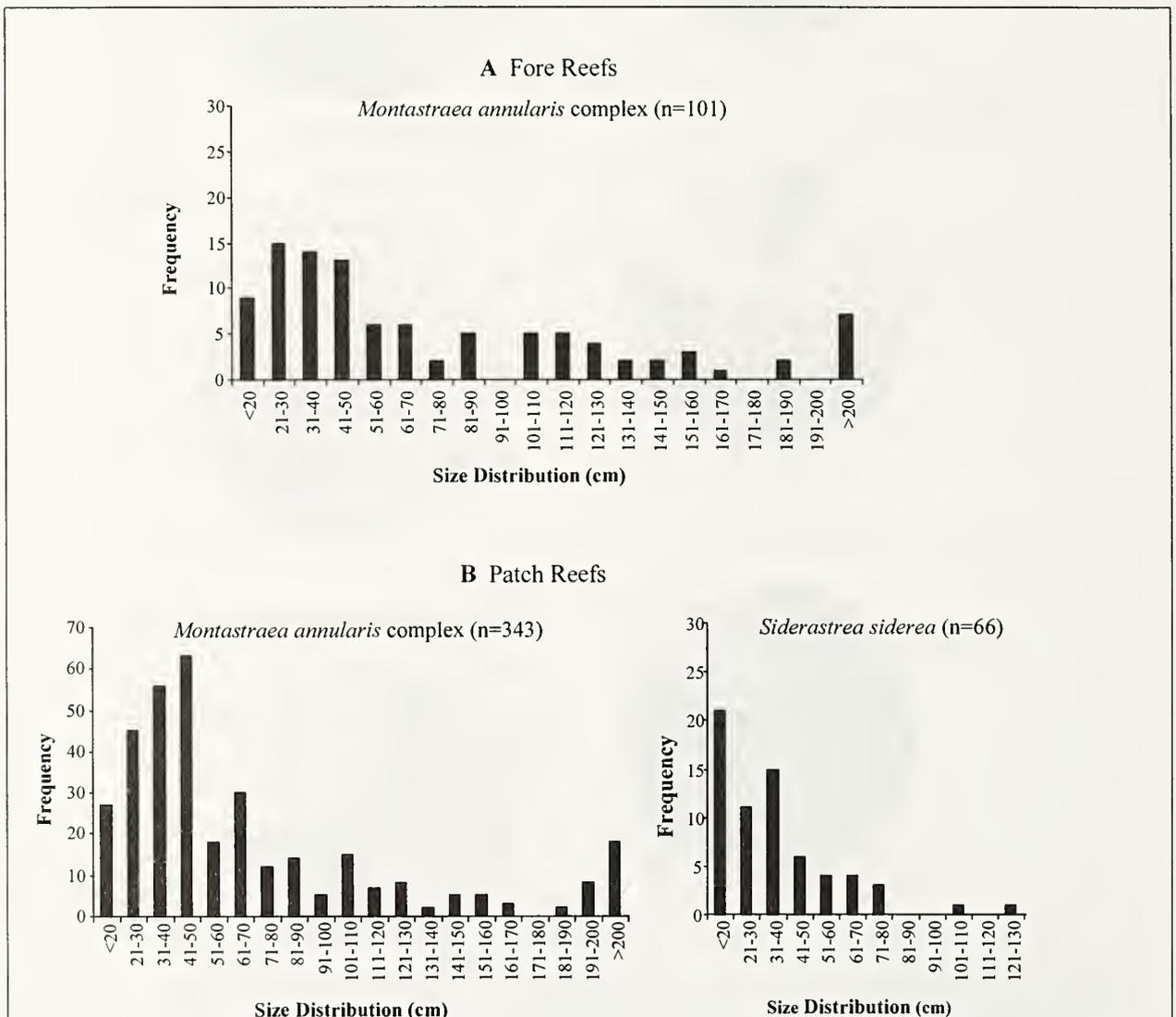
*Recruits.* The most abundant of the coral recruits was *Porites astreoides*, which ranged from 31% of the total in the fore reefs to 50% in the patch reefs (Fig. 3A-C). Recruits of the *M. annularis* complex were rare (<2%) in the fore reefs and only represented 9% of the total in the patch reefs. Recruits of *A. palmata* were only found in



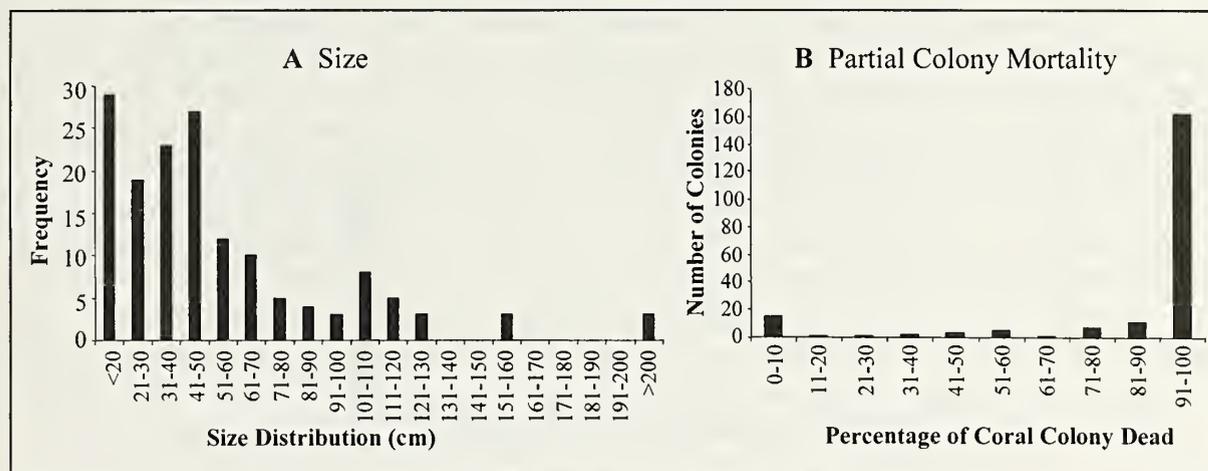
**Figure 3.** Species composition and mean relative abundance of all stony coral recruits ( $\leq 2$  cm diameter) at (A) fore reefs; (B) patch reefs; (C) coral ridge reefs along the Belize barrier reef. Other category = combined coral species, each with <5% abundance of occurrence.

the patch reefs but in extremely low abundance (<1% of total). *A. tenuifolia* was represented by 9% of the recruits in the ridges, but was more abundant (13% of total) in the fore reefs.

**Coral size.** The mean sizes of individually surveyed corals ( $\geq 10$  cm in diameter) varied nearly threefold (22-60 cm) and were more variable among reefs in the fore reefs and reef ridges than in the patch reefs (Table 2). While most colonies of the *M. annularis* complex in the fore reefs and patch reefs (and *Siderastrea siderea* in patch reefs) were <100 cm in diameter, we also measured a number of large, massive colonies (especially on the fore reefs), some being greater than 200 cm in diameter (Figs. 4A-B). *A. palmata*, although not abundant in the fore reefs, showed a large range in colony size from <20 cm to > 200 cm. *A. tenuifolia* in the reef ridges also showed a great range in size, but most colonies were <50 cm in diameter (Fig. 5A).



**Figure 4.** Size-frequency distribution as % of dominant stony corals ( $\geq 10$  cm diameter) at (A) fore reefs and (B) patch reefs along the Belize barrier reef.



**Figure 5.** (A) Size-frequency distribution as % and (B) frequency distribution of total (recent + old) partial-colony mortality of *Agaricia tenuifolia* ( $\geq 10$  cm diameter) at coral reef ridges in south-central Belize.

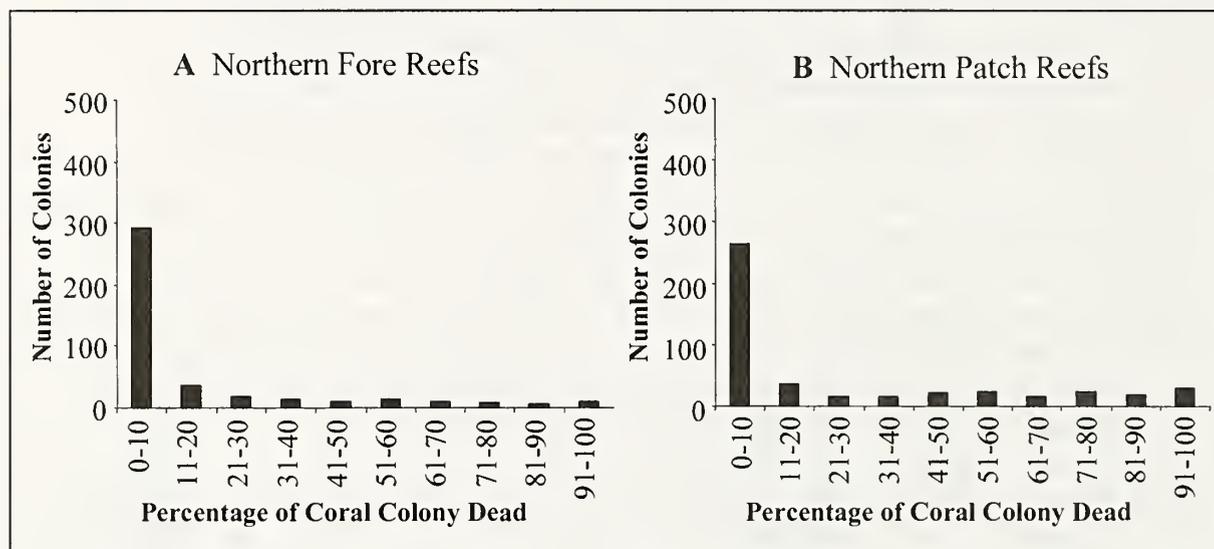
*Coral condition.* No diseased corals were found in the coral reef ridges and in the Hol Chan patch reef. Elsewhere, signs of disease were relatively low ( $\sim 2\%$ ) except in the South Water Caye fore reef where 5% of the censused corals were diseased (primarily black-band disease; Table 2). There was no clear difference in incidence of disease in fore reefs versus patch reefs but, considered together, the incidence of coral disease was twice as high in the south-central sites as in the northern region.

Long-term effects from the 1998 warming event were still evident in May 1999. Bleaching affected  $>44\%$  of the *M. annularis* complex in the south-central patch reefs, and 42% in both the northern and south-central fore reefs, while 26% of the colonies in the northern patch reefs were bleached. Fewer colonies ( $\sim 15\%$ ) of each of the other common mound corals, *Diploria* spp. and *Siderastrea siderea*, were affected by bleaching. The relatively high percentages of bleached corals in most (4/5) of the fore reefs and all five patch reefs, particularly Norvall Caye, were obviously significant (Table 2). The percentages of bleached corals were higher on the south-central patch reefs (17-40%) than in the corresponding northern reefs (15-16%). The fore reefs showed an opposite pattern with the northern reefs having proportionately more bleached corals than those in the south-central region ( $\sim 22\%$  versus  $\sim 13\%$ , respectively). Overall we found no clear spatial differences in extent of bleaching either between the fore reefs and patch reefs or between the northern and south-central regions (Table 2).

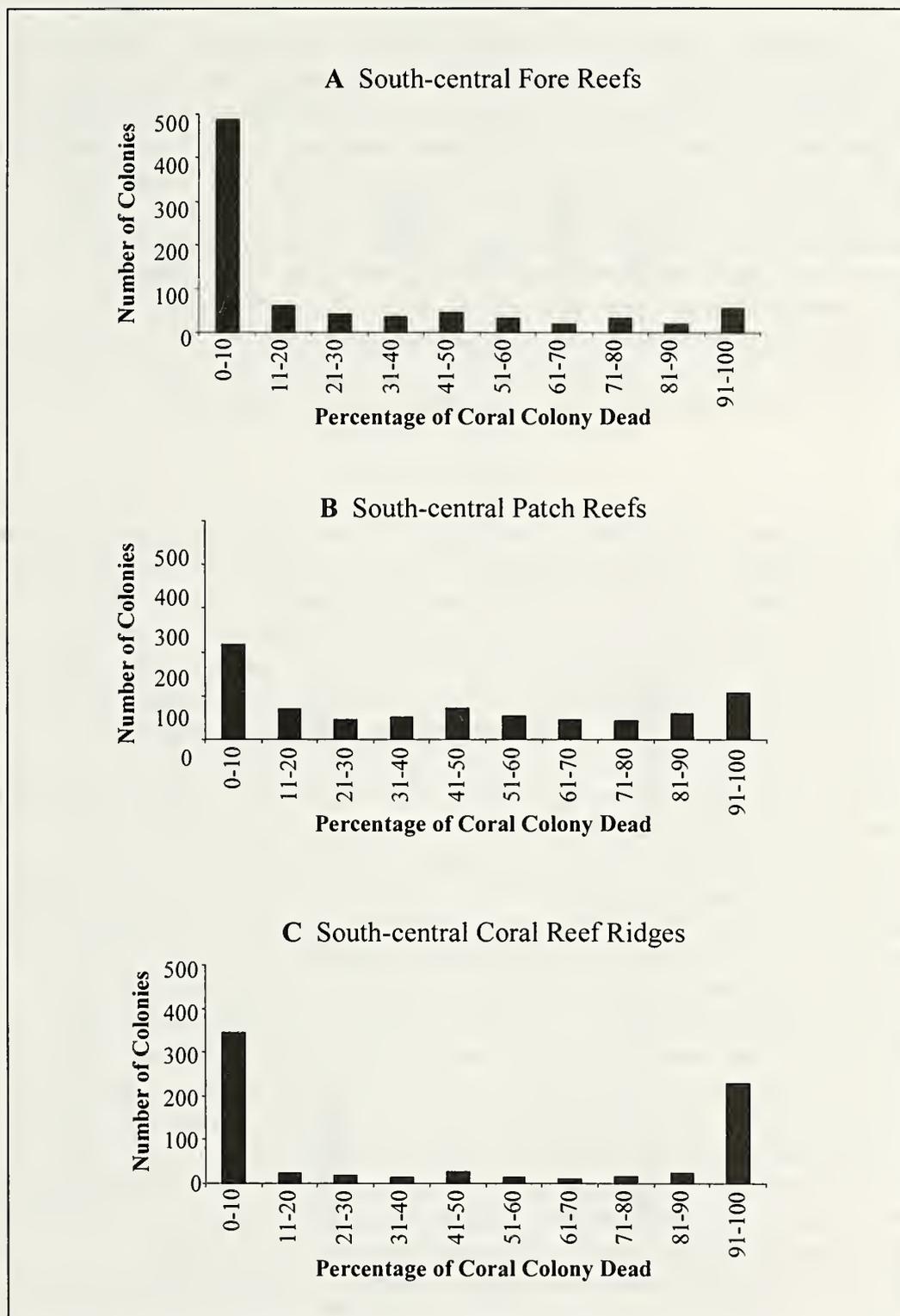
Mortality characteristics of the corals censused at the 13 sites are summarized in Table 2. Recent partial-colony mortality ranged from  $\sim 2\%$  of upper surfaces in the fore reefs to sometimes  $>13\%$  in the reef ridges. Comparable levels of old partial-colony mortality in the patch reefs and reef ridges were each about twice those in the fore reefs ( $\sim 35\%$  versus  $\sim 20\%$ ). Most colonies of *Agaricia tenuifolia* in the reef ridges were dead (Fig. 5B, with most of the *A. tenuifolia* in the 91-100% interval) and were covered by macroalgae and encrusting sponges (*Chondrilla* cf. *nucula* Schmidt; K. Rützler, pers. comm.). While the percent of colonies scored as "standing dead" (100% mortality of upper colony surfaces and still in growth position) was 4% and 8% in the fore reefs and patch reefs, respectively, standing dead ranged from 18-50% in the coral reef ridges.

Total (recent + old) partial-colony mortality (hereafter total mortality) was considerably higher in the south-central ridges and patch reefs (both ~45%) than in the northern patch reefs and south-central fore reefs (both ~25%) and relatively low in the northern fore reefs (~15%; Table 2). At the regional scale, the north Belizean fore reefs and patch reefs had lower total mortality than similar reefs in the south-central region (~20% versus ~35%, respectively). Total mortality for stony corals off Bread and Butter, Wee-Wee, and Norvall Cayes (mean = 45.5%, n = 3 reefs) was nearly two times greater than in the northern patch reefs. The greatest percentage was contributed by the *M. annularis* complex, for which total mortality values ranged between 34% (Bread and Butter) and >60% (Norvall, Wee-Wee) in the south-central patch reefs and from 35% (Hol Chan) to 50% (Mexico Rocks) in the northern region.

We plotted the frequency distributions of total mortality for sites located in northern (Fig. 6A,B) and south-central Belize (Fig. 7A-C) to illustrate these large-scale spatial differences. A majority of censused corals (>55-75% in patch reefs and fore reefs, respectively) in the northern region showed  $\leq 10\%$  total mortality of the colony surface. Although a substantial number of corals from the south-central region also had  $\leq 10\%$  total mortality, there was a greater proportion of colonies with higher percentages. For example, more than one-third of the censused corals at the coral reef ridges (n = 723) showed total mortality values of >90% of their surfaces (~75% of the colonies in this interval were *A. tenuifolia*). We also found a substantial number (>10%) of colonies with over 90% total mortality in the patch reefs (~28% of these colonies were the *M. annularis* complex).



**Figure 6.** Frequency distribution of total (recent + old) partial colony mortality of all stony corals ( $\geq 10$  cm diameter) at (A) fore reefs, (B) patch reefs in northern Belize.



**Figure 7.** Frequency distribution of total (recent + old) partial colony mortality of all stony corals ( $\geq 10$  cm diameter) at (A) fore reefs, (B) patch reefs, (C) coral reef ridges in south-central Belize.

The dominant component of the algal assemblage was turf algae in all patch reefs, all coral reef ridges, and one of the fore reefs. Macroalgae predominated in one fore reef, crustose corallines in a second, and the all three functional groups were approximately equally abundant in a third (Table 3). No difference was noted in macroalgal abundance between the northern and south-central fore reefs or patch reefs. Macroalgae in the fore reefs included the brown seaweeds, *Dictyota humifusa*, *Lobophora variegata*, and *Styopodium zonale*, and the calcified green, *Halimeda*. Dominant macroalgae in the patch reefs were *D. pulchella*, *Turbinaria turbinata*, and the red alga, *Galaxaura oblongata*. Overall, mean macroalgal height was low (< 2 cm). Macroalgal indices (mean absolute macroalgal abundance x mean macroalgal height) were: <25 in two of the reef ridges, the two most southerly patch reefs and the most southerly fore reef; <100 at six sites; and >100 in one each of the south-central fore reefs and patch reefs.

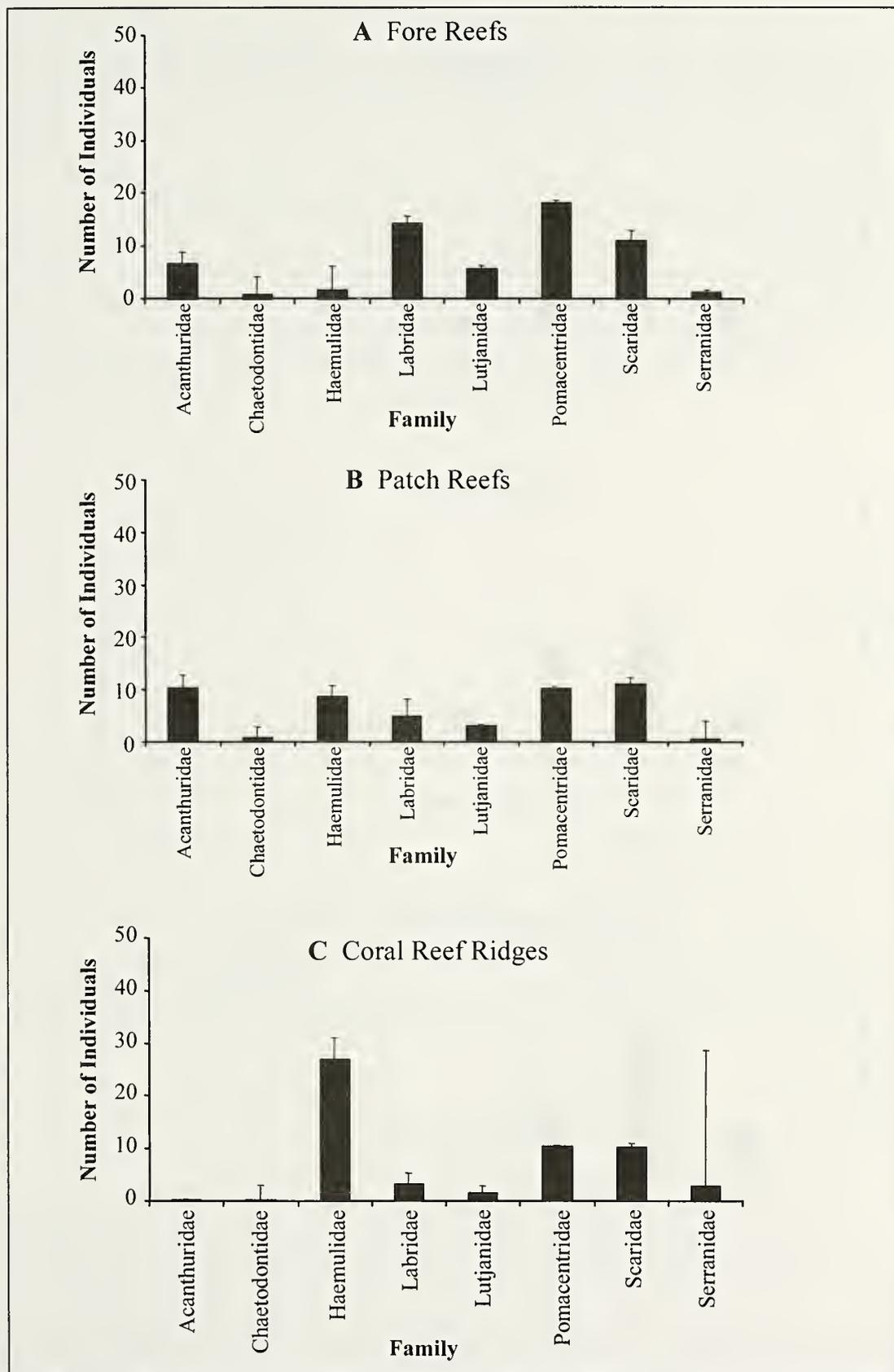
### Fishes

For the eight censused families, the highest fish abundances were in the Tunicate Cove ridge reef (mean = 105.1 individuals/200 m<sup>3</sup>, se = 26). Mean fish densities in the five fore reefs ranged from 38.2 (se = 8.7) at San Pedro Canyon to 86 (se = 20.7) individuals/200m<sup>3</sup> at Curlew but showed no regional trend (Table 1). The patch reefs showed relatively little spatial variation with fish densities ranging from 41.6 (se = 6.1) in Norvall to 60.3 (se = 12.2) individuals/200m<sup>3</sup> in Bread and Butter. The two other reef ridges had low fish abundances (overall averaging about 30 individuals/200 m<sup>3</sup>).

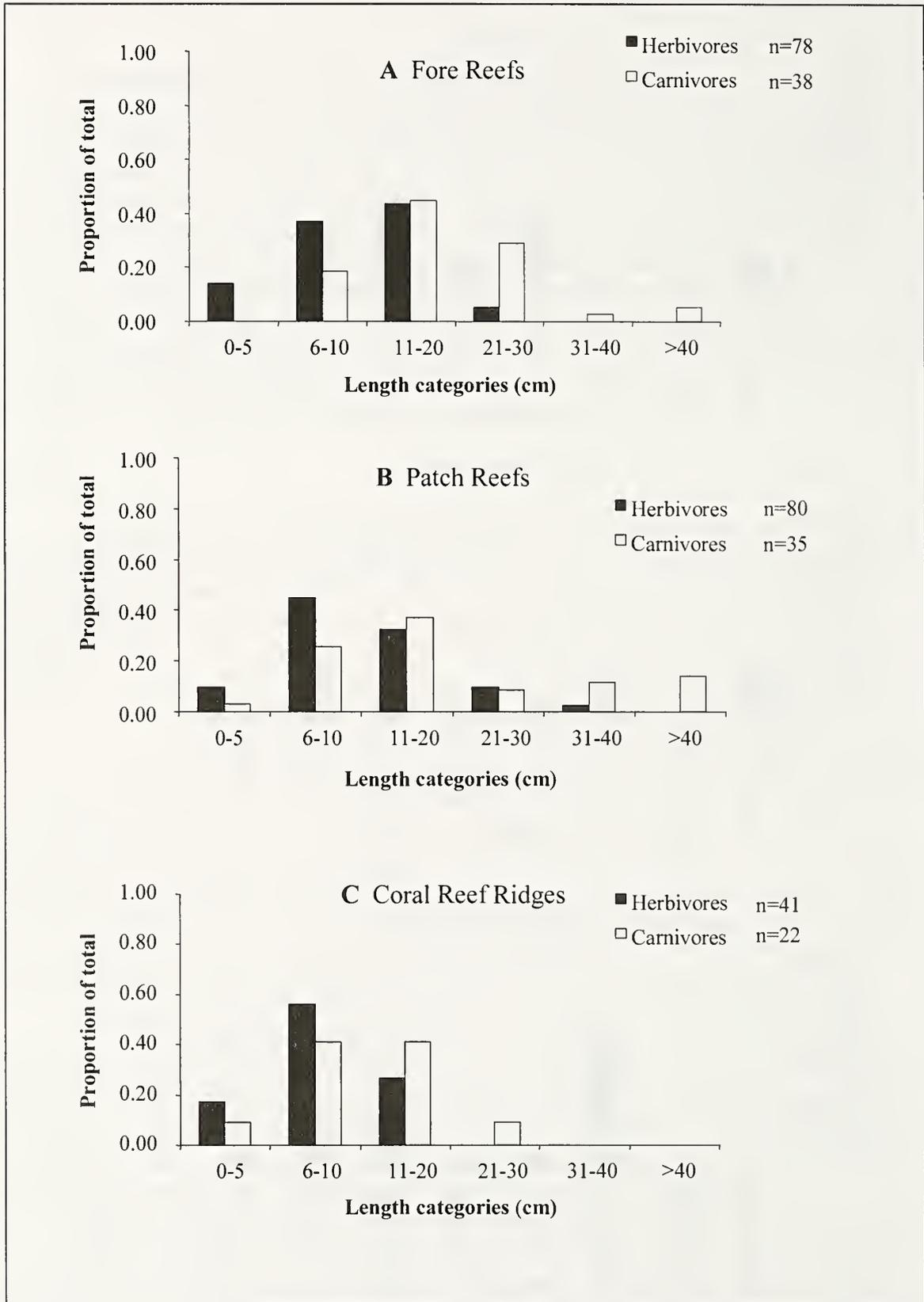
While damselfishes (Pomacentridae) were most abundant in the fore reefs (Fig. 8A), dominance was shared by four fish families in the patch reefs (Fig. 8B): surgeonfish (Acanthuridae), damselfish, parrotfish (Scaridae) and grunt (Haemulidae). Large (sometimes >100) schools of white grunt (*Haemulon plumieri*) associated with nearby seagrass habitat at Tunicate Cove ridge (Table 4) largely accounted for the high grunt densities in the reef ridges (Fig. 8C). Damselfish represented nearly a half (45%) of the total abundances in the other reef ridges. Parrotfish were more abundant than acanthurids in most (11/13) sites, and the latter were rare in the ridges. Only Mexico Rocks in northern Belize showed dominance by surgeonfishes over other herbivorous fishes.

Densities of seabasses (Serranidae), snappers (Lutjanidae) and grunts varied greatly among sites (Table 4), ranging from combined totals of ~2 individuals/200 m<sup>3</sup> (Wee-Wee reef ridge) to 86 individuals/200 m<sup>3</sup> (Tunicate ridge). The latter also had the highest abundances of serranids and grunts, whereas snappers were most common in the Tobacco Caye fore reef. Excluding Tunicate Cove, the Hol Chan Marine Reserve patch reef showed highest combined densities (~30 individuals/200 m<sup>3</sup>) of these three families with snappers and grunts contributing the greatest proportion to that total.

The size-frequency distributions for two major guilds of fishes, the herbivores (parrotfishes, surgeonfishes, the yellowtail damselfish *Microspathodon chrysurus*) and carnivores (groupers and snappers) in each of the three reef types are shown in Figure 9. While the 6-10 cm length category clearly predominated in patch reefs and reef ridges for the herbivores, the 11-20 cm category was slightly more common in the fore reefs. Few herbivorous fishes were greater than 20 cm in length in any site. Although the majority



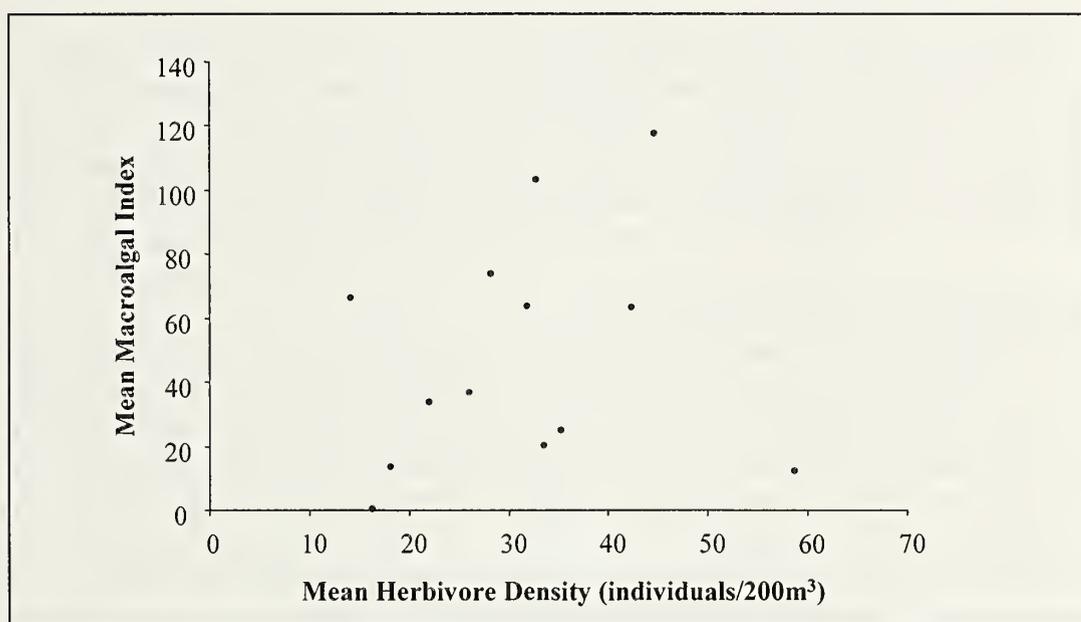
**Figure 8.** Mean fish density (no. individuals/200 m<sup>3</sup> ± se) for eight fish families at (A) fore reefs; (B) patch reefs; (C) coral ridge reefs along the Belize barrier reef.



**Figure 9.** Size-frequency distribution of herbivores (all acanthurids and scarids, *Microspathodon chrysurus*) and carnivores (all lutjanids and serranids) at (A) fore reefs; (B) patch reefs; (C) coral ridge reefs along the Belize barrier reef.

(80%) of the carnivores fell within the 6-10 cm or 11-20 cm length range, a greater range of sizes was present in the fore reefs and patch reefs than in the reef ridges. In particular, the patch reefs supported a substantial number (15% of the sample) of carnivores of the largest (>40 cm) size category, all of which were found within the Hol Chan Marine Reserve.

At the 12 sites for which we have measurements, herbivorous fish grazing rates ranged from 1.4 bites/min. (Peter Douglas reef ridge) to 13.4 bites/min. (Norvall Caye patch reef; Table 5). Overall, grazing rates were slightly higher in the patch reefs (mean = 6.4 bites/min.,  $n = 5$  reefs) than in the fore reefs (mean = 3.2 bites/min.,  $n = 5$  reefs). Generally, parrotfishes and surgeonfishes were the predominant grazers in patch reefs and fore-reefs, respectively. No relationship was found between herbivorous fish density and macroalgal index (Fig. 10) nor between grazing rate and macroalgal index (regression analysis,  $F < 1$ ).



**Figure 10.** Regression plot between mean herbivore density (no. individuals/200 m<sup>3</sup>) and mean macroalgal index, by site along the Belize barrier reef.

## DISCUSSION

Our May, 1999 AGRRA surveys along the northern and south-central regions of the Belize barrier reef were conducted approximately seven-eight months after the occurrence of two large-scale disturbances, a major thermal anomaly and the passage of Hurricane Mitch. Many corals had not yet recovered from the September 1998 warming event. Although the fore reefs and patch reefs appeared to have lower recovery rates than the coral reef ridges, this is only because *Agaricia tenuifolia*, the species most affected by the thermal anomaly, had sustained high rates of total colony mortality (see also Aronson et al., 2000). In the south-central patch reefs, 50% of the colonies of another major reef

builder, the *Montastraea annularis* complex had been bleached the previous January (Peckol et al., 2001). Over 44% of this population was still affected in May, 1999 when comparable percentages in the northern patch reef averaged 26%. However, earlier bleaching data are lacking from these northern sites, and we do not know if this regional difference represents more rapid recovery or is related to a lower initial stress during the thermal anomaly. Other common mound corals (*Diploria* spp. and *Siderastrea siderea*) showed higher incidences of recovery; whereas ~34% had been bleached in January, only 15% of all species remained discolored in May in south-central sites.

In May 1999, 42% of the *M. annularis* complex in our fore reefs had discolored tissues. The following month Kramer and Kramer (2000) found significant remnant bleaching (to >40% of the colonies) at depths of 8-20m in 36 fore reefs off Belize. In contrast, Mumby (1999) reported that though 70-90% of coral colonies in the fore reef (at 8-10 m) of Glovers Atoll (Belize) appeared to be either fully or partially bleached in response to the 1998 warming event, these colonies rapidly (by late December 1998) regained their usual coloration .

A number of studies have reported that recovery from bleaching proceeds more slowly in corals occurring at greater depths (see Kramer and Kramer, 2000). The long-term impacts and potential for recovery of the Belize barrier reef from such a large-scale thermal anomaly are uncertain. Generally, our survey of the Belize barrier reef suggests a degradation of this once largely pristine reef environment (Perkins and Carr, 1985; Cortés, 1997). Such deterioration may stem from a combination of direct and indirect effects from global climate change (Aronson et al., 2000) and regional human activities (Higinio and Munt, 1993; Carter et al., 1994).

Within-region partial mortality rates of stony corals were higher in lagoonal reefs than in corresponding fore reefs, suggestive of greater effects overall from the 1998 coral bleaching event than from Hurricane Mitch. The greatest impacts from this major storm were found in fore reefs and outer atoll reefs off Belize (Mumby, 1999). Kramer and Kramer (2000) reported major damage at several sites off Belize where damage from Hurricane Mitch resulted in a notable loss of reef structure; they noted storm damage down to their deepest survey site at 20 m in the fore reef. During a previous AGRRA survey in January 1999 in a fore reef near South Water Caye (Peckol et al., unpublished data), we documented breakage of branching corals, including *Millepora alcicornis* (33%), *M. complanata* (10%), and *Porites porites* (30%). We also recorded toppling of about 3% of the colonies of the *M. annularis* complex. Substantial amounts of rubble and entire mound corals had been transported onto and behind the reef crest. By May 1999, however, our AGRRA survey found limited evidence of the hurricane effects on the attached corals occurring in fore reefs.

Because major frame-builders, such as the *Montastraea annularis* complex and *Acropora palmata*, were rare as recruits in our sites, reef recovery may be slow following major disturbances like hurricanes and widespread bleaching. For example, although the *M. annularis* complex represented nearly 50% of the colonies  $\geq 10$  cm diameter in the patch reefs, it showed relatively low (9%) recruit abundance. Moreover, increasing macroalgal and sponge cover on the dead colonies of *Agaricia tenuifolia* in the coral reef ridges following the thermal anomaly (Peckol et al., unpublished data) may limit coral recruitment onto this substrate. Aronson and Precht (2001) reported that *Echinometra viridis* had suppressed macroalgal growth after the mass mortality of *A. cervicornis* from

white-band disease in the 1980s, allowing recruitment of *A. tenuifolia*. Although *Porites astreoides* was the dominant recruit (nearly 50% numerically) in this habitat, we did find some recruits (9% of total) of the formerly dominant *A. tenuifolia*.

Turf algae dominated in patch and coral reef ridges, yet macroalgae were quite prevalent representing >30% absolute abundance in the quadrats at 6/13 sites and exceeding 40% in two of the fore reefs (Table 3). Many Caribbean reefs have experienced exponential increases in macroalgal cover (Hughes, 1994; Shulman and Robertson, 1996; Peckol et al., this volume) often attributed to reductions in the sea urchin, *Diadema antillarum*, and herbivorous fishes (Hughes, 1994; Hughes et al., 1999). However, densities of *D. antillarum* were relatively low in the Belize barrier reef before the mass mortality event (Hay, 1984) and, until recently, much of this reef system has experienced low fishing pressures. Rapidly increasing abundances of brown macroalgae have been reported in a number of fore reefs in several studies of the Belize barrier reef (Littler et al., 1987; Aronson et al., 1994), including areas considered relatively remote from human activities (McClanahan et al., 1999). The local cause(s) of such change for the Belize barrier reef remains unclear although high rates of coral mortality, the two warming events of the 1990s, nutrient loading, and reduced herbivory from fishes have been suggested (Aronson and Precht 1997; McClanahan et al., 1999; Aronson et al., 2000). Because many areas experiencing increased macroalgal cover are remote from human activities, nutrient loading is unlikely to be the direct cause of such changes (McClanahan et al., 1999).

The grazing rates measured in the present study were approximately twice as high in patch reefs as in fore reefs. In addition, mean parrotfish densities were somewhat greater in south-central patch reefs compared with other sites in Belize and grazing by parrotfish represented >50% of the observed activities in these patch reefs where turf algae still dominate. Parrotfish densities overall (~5.2/200 m<sup>3</sup> and in leeward patch reefs (~6.5/200m<sup>3</sup>) documented in a 1998 AGRRA survey off San Salvador Island, Bahamas (Peckol et al., this volume) were about half those measured in the Belize barrier reef (overall 1~10.9/200 m<sup>3</sup>; patch reefs ~11.2/200 m<sup>3</sup>). Absolute macroalgal abundances were approximately two (44% overall) to three (59% on patch reefs) times greater off San Salvador Island than in Belize (~25%). Similar to Belize, the San Salvador reefs are also experiencing increases in cover by brown seaweeds. Lewis (1985) reported that parrotfishes actively grazed on several brown algal genera not consumed by surgeonfishes. Perhaps changing abundance patterns of parrotfishes, particularly in fore reefs, in part has contributed to increases in brown macroalgae in the Belize barrier reef. Further study should allow us to document such a relationship.

The Hol Chan patch reef had the highest combined densities of carnivorous fishes (seabasses, snappers, grunts) of all the surveyed patch reefs and the largest carnivores were found within the reserve. Partial-coral mortality was lower in the northern region than at the south-central sites. These spatial differences in reef condition (coral colony mortality, fish population abundance, and size) in part may be related to the establishment of the Hol Chan Marine Reserve in 1987. Prior to its establishment, the area was subject to uncontrolled fishing pressures that had removed most of the large and mobile fish from the reef and to burgeoning numbers of tourists who often damaged its corals. With the establishment of the reserve, these issues were largely addressed (e.g., installation of mooring buoys, education of tourists by local guides about behavior while

diving on reefs), and the community gained a critical conservation ethic and pride in their unique marine resource (Carter et al., 1994). Recent studies, which included the Hol Chan Marine Reserve (Sedberry et al., 1992; Roberts, 1995), have demonstrated that snapper and grouper populations are more abundant and fishes are larger within reserves than outside their boundaries. The added benefit of marine reserves is that they can result in increased fish abundances in adjacent reefs as a "spill-over effect" (Russ and Alcala, 1996). Clearly, this small but successful reserve admirably serves as a model for future reef conservation for the Belize barrier reef system as well as elsewhere.

### ACKNOWLEDGMENTS

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Table 1. Site information for AGRRA stony coral, algal and fish surveys off Belize, May 1999.

Site name	Reef type	Latitude (°N)	Longitude (°W)	Survey date	Depth (m)	Benthic transects (#)	≥10 cm stony corals (#/10 m)	Fish cylinders (#)	Fish species <sup>1</sup> (#)	Fish density (#/200 m <sup>3</sup> )
Eagle Ray Canyon	forereef	17.92	87.95	May 21 99	12-17	13	15	6	23	46.0
San Pedro Canyon	forereef	17.92	87.94	May 21 99	12-15	13	14.5	8	16	38.3
Tobacco	forereef	16.87	88.07	May 21 99	11-14	14	16.5	8	31	82.3
South Water	forereef	16.81	88.06	May 25 99	10-14	17	16.5	8	31	50.1
Curlwe	forereef	16.78	88.08	May 24 99	10-13	16	15	7	29	86.0
Hol Chan	patch reef	17.89	87.96	May 20 99	2-5	18	13.5	10	32	55.1
Mexico Rocks	patch reef	17.94	87.94	May 20 99	2-4	12	13	7	16	52.0
Wee Wee Patch	patch reef	16.76	88.14	May 25 99	2-5	15	17.5	8	26	42.3
Bread & Butter	patch reef	16.77	88.16	May 28 99	1-5	13	20.5	8	22	60.3
Norvall	patch reef	16.71	88.17	May 27 99	1-4	12	18	8	25	41.6
Wee Wee Ridge	coral reef ridge	16.76	88.14	May 26 99	5-13	13	12	8	17	39.9
Peter Douglas	coral reef ridge	16.71	88.17	May 27 99	7-13	12	17.5	6	14	21.7
Tunicate	coral reef ridge	16.66	88.19	May 27 99	5-14	12	16	7	21	105.1

<sup>1</sup>Fish species = all species of acanthurids, chaetodontids, haemulids, labrids, lutjanids, pomacentrids, scarids and serranids.

Table 2. Size and condition (mean  $\pm$  standard error) of all stony corals ( $\geq 10$  cm diameter) by site off Belize.

Site name	Stony corals		Partial-colony surface mortality (%)			Stony corals (%)		
	(#)	Diameter (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased
<b>Fore reefs</b>								
Eagle Ray	205	29.0 $\pm$ 2.2	1.5 $\pm$ 0.6	16.0 $\pm$ 1.8	17.0 $\pm$ 2.0	1.5 $\pm$ 1.0	26	2
San Pedro	209	22.0 $\pm$ 1.5	2.5 $\pm$ 0.7	10.5 $\pm$ 1.5	13.0 $\pm$ 1.5	1.5 $\pm$ 1.5	18	1
Tobacco	256	50.5 $\pm$ 5.3	1.5 $\pm$ 0.5	22.5 $\pm$ 2.0	24.0 $\pm$ 2.0	8.5 $\pm$ 2.5	13	1
South Water	297	49.5 $\pm$ 2.6	3.0 $\pm$ 0.7	21.0 $\pm$ 1.7	24.0 $\pm$ 2.0	3.5 $\pm$ 1.5	18	5
Curlew	270	51.0 $\pm$ 6.4	2.0 $\pm$ 0.5	21.5 $\pm$ 1.9	23.5 $\pm$ 2.0	3.0 $\pm$ 1.0	7	2
<b>Patch Reefs</b>								
Mexico Rocks	178	46.0 $\pm$ 6.3	2.0 $\pm$ 0.6	20.5 $\pm$ 2.4	22.5 $\pm$ 2.5	3.0 $\pm$ 1.5	16	2
Hol Chan	287	46.0 $\pm$ 3.7	2.1 $\pm$ 0.6	26.5 $\pm$ 1.9	28.5 $\pm$ 2.0	5.0 $\pm$ 2.0	15	0
Bread & Butter	278	52.0 $\pm$ 3.1	5.5 $\pm$ 0.3	34.0 $\pm$ 2.0	39.0 $\pm$ 2.3	10.5 $\pm$ 3.0	17	2
Wee-Wee	278	48.5 $\pm$ 3.3	4.5 $\pm$ 1.0	46.5 $\pm$ 2.7	50.5 $\pm$ 2.8	14.5 $\pm$ 4.5	23	2
Norvall	211	60.0 $\pm$ 4.8	5.5 $\pm$ 1.0	41.5 $\pm$ 2.3	47.0 $\pm$ 2.2	6.5 $\pm$ 2.0	40	2
<b>Coral Reef Ridges</b>								
Wee-Wee	222	29.0 $\pm$ 4.9	3.5 $\pm$ 0.9	32.0 $\pm$ 2.7	35.5 $\pm$ 2.8	18.0 $\pm$ 4.0	2	0
Peter Douglas	266	29.0 $\pm$ 1.7	7.0 $\pm$ 1.3	28.5 $\pm$ 2.4	35.5 $\pm$ 2.6	26.5 $\pm$ 4.5	7	0
Tunicate	235	56.5 $\pm$ 7.4	13.5 $\pm$ 1.9	45.5 $\pm$ 3.0	58.5 $\pm$ 3.0	50.0 $\pm$ 6.5	6	0

Table 3. Algal characteristics and stony coral recruit abundance (mean  $\pm$  standard error) by site off Belize.

Site name	Quadrats (#)	Absolute abundance (%)		Macroalgal		Recruits (#/.0625m <sup>2</sup> )	
		Macroalgae	Turf algae	Crustose coralline algae	Height (cm)		Index <sup>1</sup>
<b>Fore reefs</b>							
Eagle Ray	35	40.0 $\pm$ 5.5	19.0 $\pm$ 4.0	35.5 $\pm$ 4.5	0.7 $\pm$ 0.1	37 $\pm$ 7	0.4 $\pm$ 0.1
San Pedro	37	33.0 $\pm$ 4.0	51.0 $\pm$ 4.0	15.5 $\pm$ 3.5	1.0 $\pm$ 0.1	34 $\pm$ 5	0.4 $\pm$ 0.1
Tobacco	60	33.5 $\pm$ 4.5	33.5 $\pm$ 4.0	35.3 $\pm$ 4.5	1.4 $\pm$ 0.1	63 $\pm$ 12	0.5 $\pm$ 0.1
South Water	62	43.5 $\pm$ 1.5	49.0 $\pm$ 2.0	40.5 $\pm$ 2.0	1.6 $\pm$ 0.1	103 $\pm$ 7	0.4 $\pm$ 0.1
Curllew	39	18.5 $\pm$ 4.5	27.5 $\pm$ 5.0	43.0 $\pm$ 5.5	0.2 $\pm$ 0.1	12 $\pm$ 6	0.5 $\pm$ 0.1
<b>Patch Reefs</b>							
Mexico Rocks	38	26.5 $\pm$ 5.0	60.0 $\pm$ 5.5	10.0 $\pm$ 4.0	1.4 $\pm$ 0.2	64 $\pm$ 17	0.5 $\pm$ 0.1
Hol Chan	61	26.5 $\pm$ 4.0	54.0 $\pm$ 4.5	17.5 $\pm$ 3.5	1.5 $\pm$ 0.2	66 $\pm$ 13	0.6 $\pm$ 0.1
Bread & Butter	63	37.5 $\pm$ 5.0	48.5 $\pm$ 4.5	9.0 $\pm$ 3.0	1.7 $\pm$ 0.2	117 $\pm$ 23	0.3 $\pm$ 0.1
Wee-Wee	72	21.0 $\pm$ 3.5	61.0 $\pm$ 4.5	5.0 $\pm$ 2.0	0.7 $\pm$ 0.1	25 $\pm$ 6	0.4 $\pm$ 0.1
Norvall	40	12.0 $\pm$ 3.0	68.5 $\pm$ 4.5	2.5 $\pm$ 2.0	0.9 $\pm$ 0.1	20 $\pm$ 6	0.3 $\pm$ 0.1
<b>Coral Reef Ridges</b>							
Wee-Wee	40	30.5 $\pm$ 4.5	35.5 $\pm$ 6.5	5.0 $\pm$ 2.5	1.5 $\pm$ 0.3	74 $\pm$ 14	0.7 $\pm$ 0.1
Peter Douglas	11	2.0 $\pm$ 2.5	54.5 $\pm$ 11.0	4.5 $\pm$ 3.0	0.1 $\pm$ 0.0	<1 $\pm$ <1	0.7 $\pm$ 0.1
Tunicate	31	10.5 $\pm$ 2.5	32.5 $\pm$ 5.5	9.5 $\pm$ 3.0	0.7 $\pm$ 0.2	13 $\pm$ 4	0.6 $\pm$ 0.1

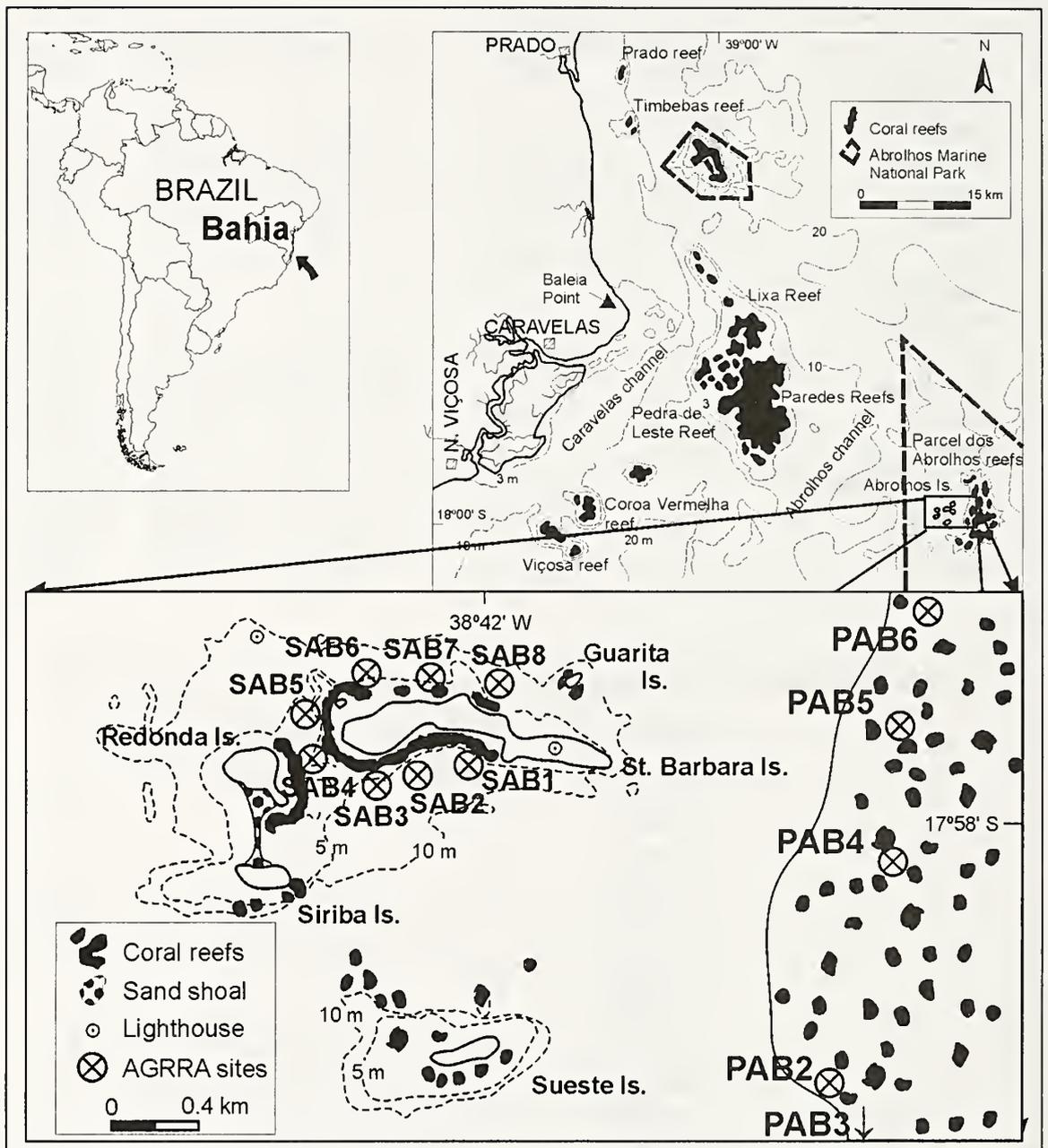
<sup>1</sup>Macroalgal index = absolute macroalgal abundance x macroalgal height.

Table 4. Density (mean  $\pm$  standard error) of selected fish families by site off Belize.

Site name	Herbivores (#/200 m <sup>3</sup> )		Carniivores (#/200 m <sup>3</sup> )		
	Acanthuridae	Scaridae	Haemulidae	Lutjanidae	Serranidae
<b>Fore reefs</b>					
Eagle Ray	4.2 $\pm$ 0.8	11.2 $\pm$ 2.1	0.6 $\pm$ 0.3	1.4 $\pm$ 0.4	3.2 $\pm$ 0.8
San Pedro	2.7 $\pm$ 1.2	7.0 $\pm$ 2.8	1.6 $\pm$ 0.7	6.9 $\pm$ 3.9	1.3 $\pm$ 0.5
Tobacco	7.4 $\pm$ 0.8	14.1 $\pm$ 2.4	3.5 $\pm$ 1.9	12.3 $\pm$ 4.1	1.3 $\pm$ 0.6
South Water	6.9 $\pm$ 0.9	12.1 $\pm$ 1.8	1.9 $\pm$ 0.4	4.3 $\pm$ 1.4	0.3 $\pm$ 0.3
Curlew	13.4 $\pm$ 7.2	11.3 $\pm$ 2.8	1.4 $\pm$ 0.4	4.1 $\pm$ 1.9	0.4 $\pm$ 0.3
<b>Patch Reefs</b>					
Mexico Rocks	18.3 $\pm$ 12.0	5.1 $\pm$ 1.3	6.4 $\pm$ 1.9	3.3 $\pm$ 0.9	0.1 $\pm$ 0.1
Hol Chan	3.6 $\pm$ 2.2	4.9 $\pm$ 1.1	20.6 $\pm$ 11.1	7.5 $\pm$ 1.9	0.7 $\pm$ 0.3
Bread & Butter	12.5 $\pm$ 9.8	22.5 $\pm$ 5.9	11.4 $\pm$ 8.8	1.6 $\pm$ 0.6	0.8 $\pm$ 0.4
Wee-Wee	8.4 $\pm$ 5.3	9.1 $\pm$ 2.4	3.5 $\pm$ 1.8	1.8 $\pm$ 0.7	0.0 $\pm$ 0.0
Norvall	9.0 $\pm$ 6.2	14.3 $\pm$ 1.8	1.9 $\pm$ 0.3	1.4 $\pm$ 0.3	1.3 $\pm$ 0.1
<b>Coral Reef Ridges</b>					
Wee-Wee	0.1 $\pm$ 0.1	10.4 $\pm$ 2.6	0.2 $\pm$ 0.2	0.6 $\pm$ 0.3	1.6 $\pm$ 0.4
Peter Douglas	0.0 $\pm$ 0.0	6.7 $\pm$ 0.8	2.8 $\pm$ 1.2	0.8 $\pm$ 0.3	1.3 $\pm$ 0.2
Tunicate	0.4 $\pm$ 0.4	13.6 $\pm$ 2.5	77.6 $\pm$ 24.5	3.0 $\pm$ 1.5	5.4 $\pm$ 2.7

Table 5. Herbivorous fish grazing rates (mean  $\pm$  standard error) for sites in Belize.

Site name	Observations (#)	Grazing rate (#bites/min.)
<b>Fore reefs</b>		
Eagle Ray		not done
San Pedro	5	2.6 $\pm$ 1.2
Tobacco	4	5.4 $\pm$ 1.9
South Water	4	4.7 $\pm$ 2.2
Curlew	2	2.1 $\pm$ 0.5
<b>Patch Reefs</b>		
Mexico Rocks	5	10.1 $\pm$ 3.8
Hol Chan	10	3.5 $\pm$ 1.1
Bread & Butter	4	3.2 $\pm$ 1.2
Wee-Wee	4	2.8 $\pm$ 0.8
Norvall	4	13.4 $\pm$ 6.2
<b>Coral Reef Ridges</b>		
Wee-Wee	4	2.8 $\pm$ 0.9
Peter Douglas	4	1.4 $\pm$ 0.8
Tunicate	4	7.8 $\pm$ 1.9



**Figure 1.** AGRRRA survey sites off Santa Bárbara Island (SAB) and in the Parcel dos Abrolhos chapeirões (PAB) in the Abrolhos, Brazil. See Table 1 for site codes.

# RAPID ASSESSMENT OF THE ABROLHOS REEFS, EASTERN BRAZIL (PART 1: STONY CORALS AND ALGAE)

BY

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## ABSTRACT

In March-April 2000, a survey applying the Atlantic and Gulf Rapid Reef Assessment benthos protocol was accomplished in the Abrolhos National Marine Park. The Santa Bárbara Island fringing reef and offshore “chapeirões” (isolated columnar reefs) were assessed to evaluate their present status and provide standards for future monitoring programs. The chapeirões were rated as well preserved in terms of the density and health of large ( $\geq 25$  cm in diameter) stony corals, the density of stony coral recruits, and the scarcity of benthic macroalgae. However, concerns were raised about the fringing reef, particularly off the island’s southern coast. Although the causes for its poor condition here are not well understood, the intrinsic oceanographic setting, in particular exposure to storm waves in winter, and the presence of numerous tourist divers and snorkelers during the summer must be investigated.

## INTRODUCTION

The southernmost coral reefs in the western Atlantic Ocean are found in Brazil (Fig. 1). They are spread over a distance of approximately 2,000 km between 0°50' and 19° S latitude. Although some information about the Brazilian reefs has existed for over a century, many reef areas are still poorly known and there are few quantitative assessments of their condition. Along the coast of the state of Bahia (between 12°30' and 19° S latitude), where most coral reefs occur, few previous reef assessments have been performed (Leão et al., 1997; Castro and Pires, 1999; Leão et al., 1999). In the southern part of Bahia state (at 17°-19° S latitude), the continental shelf widens to 200 km forming the Abrolhos Bank, where the largest and richest of Brazil’s coral reefs are scattered over

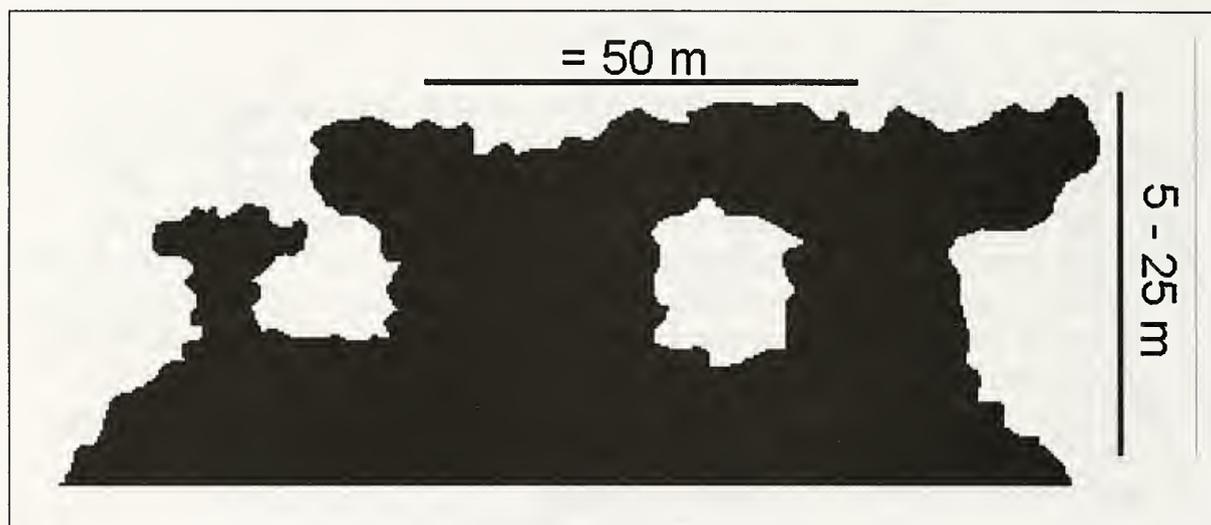
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an area of approximately 6,000 km<sup>2</sup> (Fig. 1). Isolated bank reefs of varied shapes (elongated, circular, etc.), with dimensions that range from less than 1 km to about 20 km in length, occur nearshore in depths of 10-20 m. Shallow offshore (<10 m deep) fringing reefs border the islands of the Abrolhos Archipelago, and enormous (maximally to 25 m high) isolated mushroom-shaped reef columns, or “chapeirões” (Fig. 2), are found in deeper water. The morphology of these major reef types seems to be related to the underlying substratum (older reefs, Precambrian bedrock, beachrock, etc.), the prevailing hydraulic regime, and their position relative to present sea level (Leão, 1996; Kikuchi and Leão, 1998; Leão and Kikuchi, 1999; Leão et al., 2001).



**Figure 2.** Cross-sectional sketch of the mushroom-like growth form of the Brazilian chapeirões.

The Abrolhos reefs are built by a stony coral fauna that is characterized by its very low diversity compared with those of the North Atlantic or Indo-Pacific Oceans, by the endemic character of its major reef-builders, and by the complete absence of species with branching morphologies (Belém et al., 1986; Castro, 1994). Of the 18 species identified so far in Brazilian reefs, 17 are reported from the Abrolhos region. Most common are the six endemic scleractinians (*Mussismilia braziliensis*, *M. hispida*, *M. hartii*, *Siderastrea stellata*, *Favia gravida* and *F. leptophylla*), some of which have affinities with modern Caribbean species, while others are related to a Tertiary coral fauna that may have been isolated from the Caribbean Sea by the Amazon River flow, after its reversal due to the elevation of the Andes during the Tertiary. The hydrocoral *Millepora alcicornis* occupies reef-edge habitats, but all Brazilian reefs lack the acroporids which are major components of many reef-crest and fore-reef zones in North Atlantic reefs.

Stony corals inhabiting the nearshore bank reefs are naturally exposed to influxes of terrigenous sediment (Martin et al., 1985). They are now threatened by accelerated urban expansion which has caused an increase in coastal runoff and untreated sewage discharges. The offshore reefs, in contrast, are located away from the mainland and are partially protected by law (see Kikuchi et al., this volume). Previous assessments of potential human impacts in the Abrolhos National Marine Park have referred only to the

anchorage of tourist boats in seagrass beds around the islands (Creed and Amado Filho, 1999).

The northern coasts of the Abrolhos islands (Fig. 1) are exposed to relatively high wave energies between September and February, their southern sides to somewhat larger waves from April to August, while the western (leeward) coasts are protected from major wind trends. Their fringing reef flats extend about 30 m from shore, are poorly developed and, during low tides, are subaerially exposed. Seaward, the reefs gradually slope to their edges at depths of 4-5 m where they drop off to a sandy bottom at 8-10 m. The shallow fringing reefs around some of the islands are intensively used for recreational scuba diving and snorkeling. In particular, tourists dive all around Santa Bárbara Island (SAB), most frequently on its southern side.

The windward Parcel dos Abrolhos (PAB) chapeirões, which are located about 2 km east of the Abrolhos islands, form huge mushroom-shaped structures, some of which attain heights of over 20 m. Due to the limited penetration of light beneath the overhangs around their sides, the maximum development of reef corals is at depths of 5-8 m on the tops of the chapeirões. The PAB chapeirões, which are seldom visited by tourists due to difficulty of navigation and rougher sea conditions, are known to be in good condition (personal observations).

Our initial purpose in using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocols in the Abrolhos National Marine Park was to collect information on the condition of its offshore reef communities in order to establish standards for a long-term monitoring program in the park. The fish community assessment results can be found in Kikuchi et al. (this volume).

## METHODS

Benthic surveys were conducted in eight fringing reef sites around SAB that are considered representative of its northern (three sites), southern (three sites) and western (two sites) coasts. Each site was located in shallow water (3.5-5.5 m) as close as possible to the seaward edge of the reef. The three sites off the southern coast were also strategic choices because they are preferred recreational dive sites. The crests of five representative chapeirões, located at depths of 6-6.5m, were surveyed. All of these sites are routinely exposed to large waves, except for the leeward (western) fringing reef, which experiences strong tidal currents.

Two divers utilized the AGRRA Version 2.0 benthos protocol (see Appendix One, this volume) on each dive. Damselfish algal gardens were not assessed; rather the fish assessment team recorded the sizes and densities of all damselfishes. The average maximum height of all the "large" (>25cm in diameter) corals surveyed in each transect was used to approximate the rugosity of the reef surface. Two days were spent in consistency training before the beginning of the surveys.

Reef environments (the windward fringing reef off southern and northern SAB, the leeward fringing reef off western SAB, and the chapeirões) were compared on the basis of average live stony coral cover, number of large corals per transect, average maximum height of large corals, and number of large corals affected by old partial-colony mortality. Although tests of normality showed that the data did not depart from

normal distributions, most of the variances are rather unequal. Thus we chose to perform a Kruskal-Wallis non-parametric ANOVA for multisample hypothesis testing (Zar, 1999). When the null hypothesis was rejected by the Kruskal-Wallis test, we applied a non-parametric Tukey-type method to locate the differences among the data, using the test proposed by Dunn (see Zar, 1999).

## RESULTS

### Stony Corals

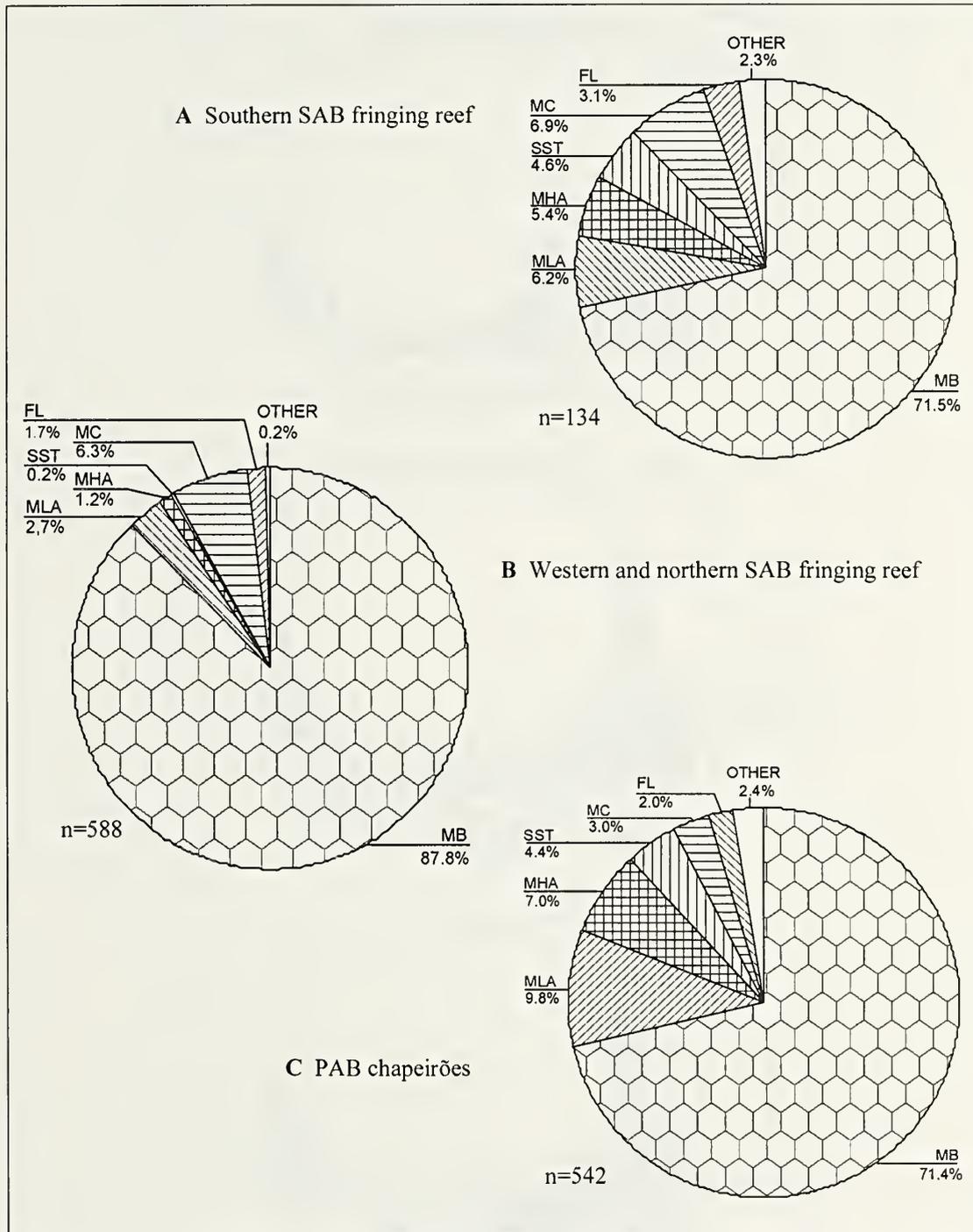
Live stony coral cover averaged about 20% in the chapeirões (Table 1) and was significantly higher than in the SAB fringing reef (Kruskal-Wallis,  $p < 0.001$ ), where corresponding cover estimates ranged from 3.5-12%. The density of large (>25 cm in diameter) stony corals varied between ~0.4/m (southern SAB) and ~1.1/m (western SAB) and was significantly smaller in the southern SAB sites (Kruskal-Wallis,  $p < 0.001$ ). Less variation was seen in the maximum height of the large corals with the tallest, averaging ~30 cm, occurring in the chapeirões (Table 2). However, stony corals in the northern SAB sites and the chapeirões are significantly taller than those in the southern and western SAB sites (Kruskal-Wallis,  $p < 0.001$ ). The mean diameter of the large corals was quite similar in the four surveyed environments varying from about 37 to 48 cm (Table 2).

Six species accounted for more than 95% of the large corals. In all sites, *Mussismilia braziliensis* was by far the dominant species (comprising >66% of all colonies) especially in the western and northern SAB sites (Table 3, Fig 3). *Millepora alcicornis* and *Montastraea cavernosa* were present in all sites but the former was more common in the chapeirões and the latter in the fringing reef. *Mussismilia hartti* was slightly more abundant in the chapeirões than in the fringing reef. *Favia leptophylla* was found in all sites, while *Siderastrea stellata* was absent from the western side of the island.

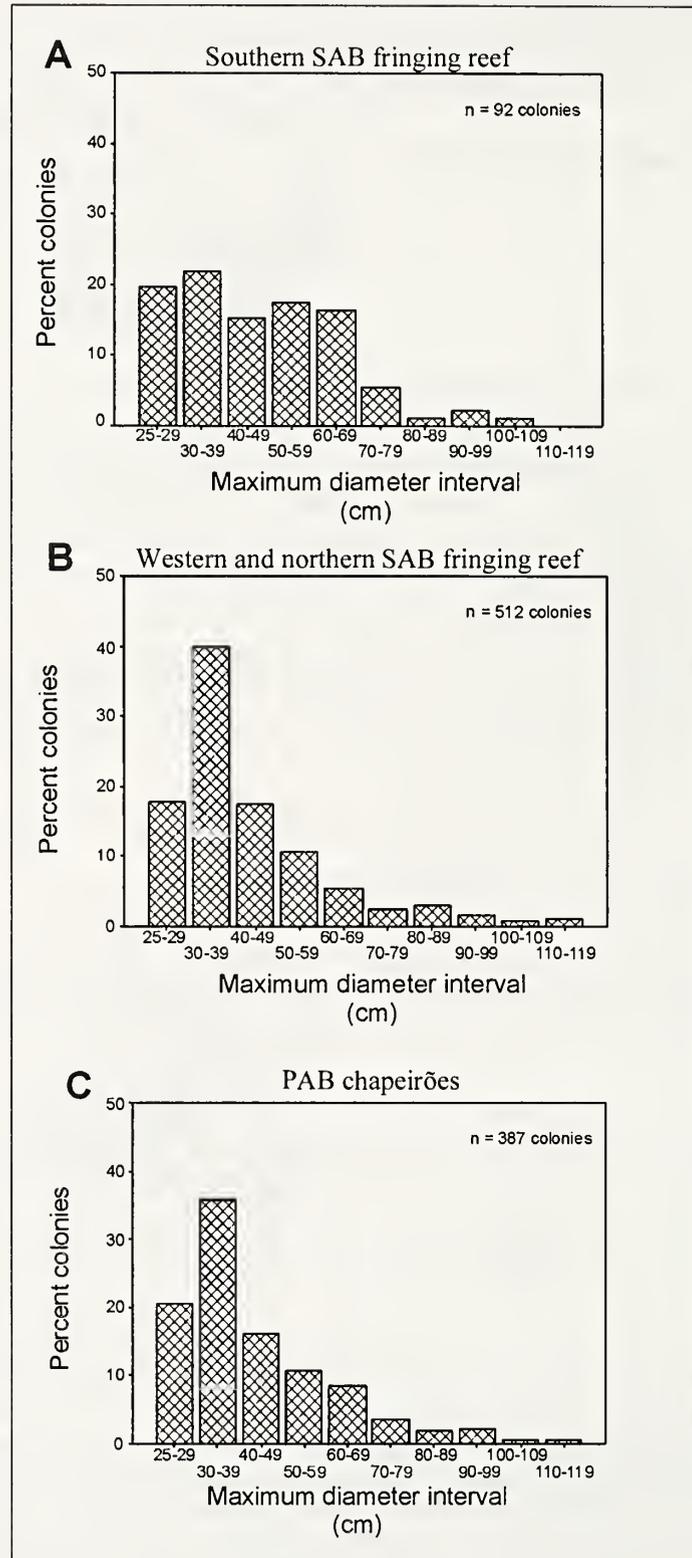
Size distributions of *Mussismilia braziliensis*, shown in Figure 4, were unimodal in the western and northern SAB fringing reef and the chapeirões, with the modes in the 30-40 cm interval. However, colonies were more evenly distributed across size classes in the southern SAB sites. Corals greater than 70 cm in diameter were more frequent in the fringing reef (where they formed more than 10% of the population) than in the chapeirões (Fig. 4).

Recent partial-colony mortality (hereafter recent mortality) of large stony corals ranged from 0 to 10% with the highest overall averages (~5-6.5%) in the chapeirões (Table 2, Fig. 5). Estimates of recent mortality for *Mussismilia braziliensis* resembled those of the entire assemblage of large corals except for one southern SAB site and one of the chapeirões where no colonies of this dominant species had incurred any recent tissue loss (Table 3).

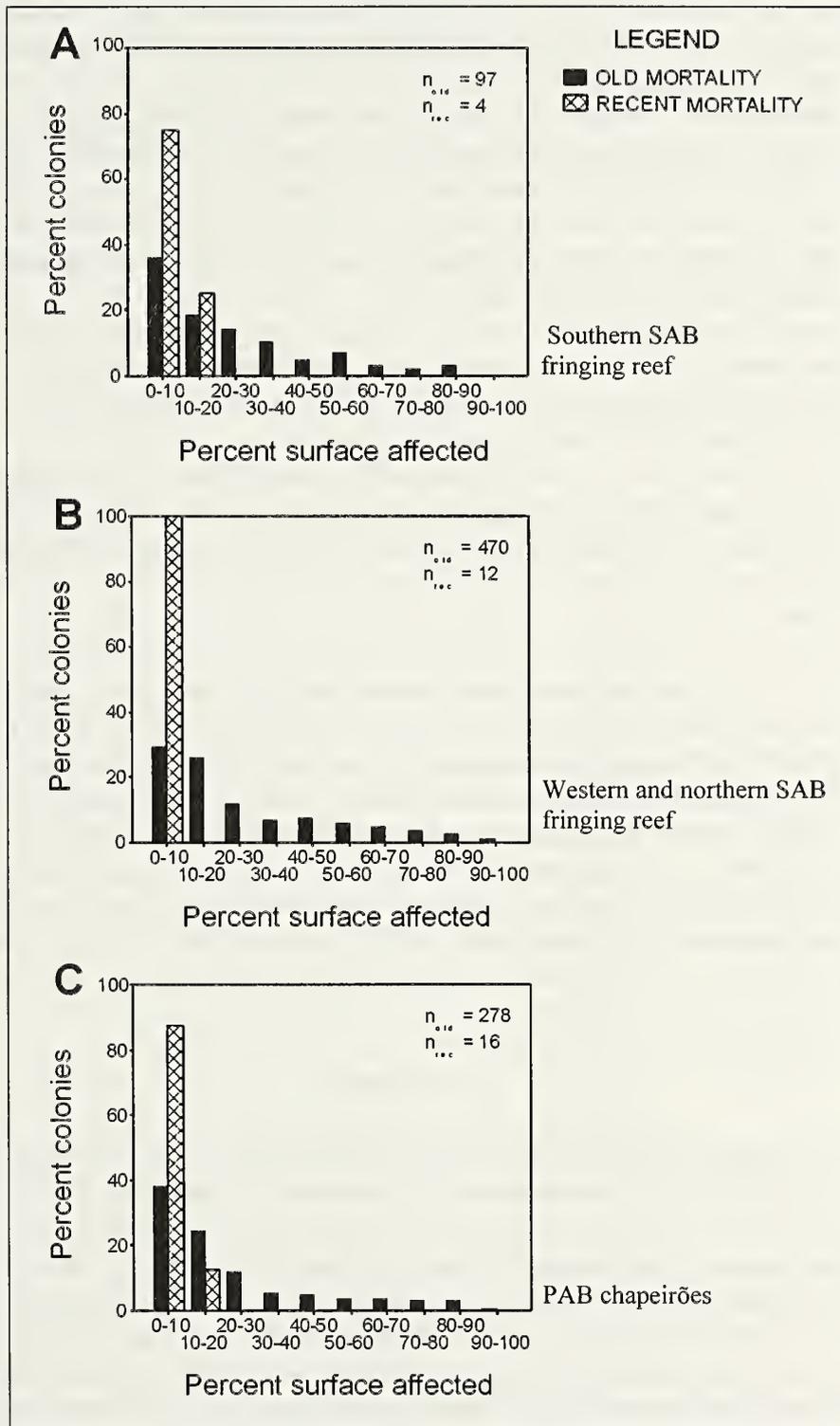
Mean estimates of old partial-colony mortality (hereafter old mortality) for large corals overlapped in the four surveyed environments (individual site averages varied twofold, between ~17 and 32%) as shown in Figure 5, although proportionately fewer



**Figure 3.** Species composition and mean relative abundance of all stony corals ( $\geq 25$  cm diameter) in the (A) southern and (B) western and northern SAB fringing reef, (C) PAB chapeirões, in the Abrolhos, Brazil. FL = *Favia leptophyla*, MC = *Montastraea cavernosa*, MLA = *Millepora alcicornis*, MB = *Mussismilia braziliensis*, MHA = *M. hartti*, SST = *Siderastrea stellata*.



**Figure 4.** Size-frequency distribution of colonies ( $\geq 25$  cm diameter) of *Mussismilia braziliensis* in the (A) southern and (B) western and northern SAB fringing reef, and (C) PAB chapeirões in the Abrolhos, Brazil.



**Figure 5.** Frequency distribution of recent partial colony mortality and old partial colony mortality of all stony corals ( $\geq 25$  cm diameter) in the (A) southern, (B) western and northern SAB fringing reef, and (C) PAB chapeirões in the Abrolhos, Brazil.

colonies were affected in the chapeirões than in the fringing reef (~35-62% versus ~64-86%, respectively) (Table 2). The number of colonies with old mortality was significantly greater in the western SAB sites than in the southern SAB sites and the chapeirões (Kruskal-Wallis,  $p < 0.001$ ).

Mean estimates of total (recent + old) partial-colony mortality (hereafter total mortality) exhibited similar patterns to those described for old mortality (Table 2) by overlapping in the four surveyed environments with proportionately fewer colonies being affected in the chapeirões than in the fringing reef (~38-64% compared to 64-84%, respectively). Similar patterns for old mortality and total mortality characteristics were seen in *Mussismilia braziliensis* (Table 3).

No signs of disease were seen in any of the large stony corals in the survey sites (Table 2). No bleached colonies were observed in the northern SAB sites and they were rare (0-3.5% of corals) off the southern and western sides of the island. Levels of bleaching were slightly higher in the chapeirões (1.5-7%).

The density of coral recruits was higher in the chapeirões than in the fringing reef (Table 4). The most common species found as recruits in both habitats were *Siderastrea stellata*, *Agaricia agaricites*, *Favia gravida* and, very rarely, *Mussismilia braziliensis*.

#### Algae and *Diadema antillarum*

Turf algae clearly predominated everywhere except in one of the northern SAB sites (SAB8), and were particularly abundant (>80% relative abundance) in the other two northern sites (Table 4). The relative abundance of macroalgae was higher overall in the fringing reef (<0.5-58%, n=8 sites), where they formed the dominant algal group in one northern site (SAB8), than in the chapeirões (<0.5-7%, n=5 sites). The average height of the macroalgae and the macroalgal index (relative macroalgal abundance x macroalgal height) were also greater in all but one of the SAB sites than in the chapeirões. Crustose coralline algae were relatively more abundant in the chapeirões than in the fringing reef. No individuals of *Diadema antillarum* were found during the surveys.

## DISCUSSION

Although the crests of the chapeirões were slightly deeper (6-6.5 m) than the fringing reef (3.5-5.5 m), most sites were similar in terms of the diameter and old partial-mortality of large ( $\geq 25$  cm) stony corals and in the predominance of turf algae on the substratum. Nevertheless, considering their higher values for live stony coral cover, maximum coral height and coral recruit density, as well as their low macroalgal indices (a proxy for macroalgal biomass), the chapeirões appeared to be in better condition overall than the SAB fringing reef. The worst indicators were found on the southern fringing reef where the density of large corals and live stony coral cover are both particularly low. It should be remembered that the southern SAB reef is most heavily visited by tourists (divers and snorkelers) during the summer.

Species richness among large stony corals may be higher in the more exposed southern fringing reef and the chapeirões than in the western and northern SAB sites. Storms may prevent *Mussismilia braziliensis* (which, due to its characteristic mushroom

shape, breaks and tumbles easily) from dominating other species. Although large (>40 cm) colonies of *M. braziliensis* are relatively more common in the southern SAB sites where coral density is low, we found no colonies greater than 110 cm in diameter in this environment. Intense fragmentation and subsequent mortality of the fragments may reduce the abundance both of the small and of the very large (>110 cm) size classes; alternatively recruitment and/or survival of young colonies may be less successful in this habitat.

So far neither outbreaks of disease nor any mass mortality events have been witnessed among the Brazilian stony coral populations. The somewhat higher values of recent partial-colony mortality and the slightly higher percentage of bleached corals that were present in the chapeirões presently lack an explanation. On the other hand, the percentage of colonies with old partial-mortality was substantially higher in the fringing reef than in the chapeirões (Table 2). That the southern SAB sites were not significantly different from the chapeirões can be tentatively explained by the scarcity of large corals (i.e., small sample sizes) off this side of the island.

The density of coral recruits was inversely related to the relative abundance of macroalgae, hence higher in the chapeirões than in the SAB fringing reef. The dominance that we found by recruits of *Siderastrea* and *Agaricia* is expected since the life history strategies of many species in these genera are to concentrate energy in reproduction rather than long-term survival (Bak and Engel, 1979). Clearly, this is not the case for *M. braziliensis*; nonetheless, a few of its recruits were present in this survey.

The higher relative abundance of macroalgae and the much higher macroalgal index values found in the SAB fringing reef compared to the more isolated chapeirões are also expected, given their proximity to the island. In particular, sewage seep from the houses (near SAB8) and inputs from the nesting birds and/or the decomposed remains of dead birds in the “bird’s cemetery” on the northwestern coast (near SAB5), would augment nutrients in natural runoff during storms (see Kikucki et al., this volume). Grazing pressures could also be higher in the chapeirões: *Echinometra lucunter*, which is known to be a significant herbivore, is more abundant than *Diadema antillarum* along the eastern coast of Brazil, but its impact in the offshore Abrolhos reefs is not known. However, *E. lucunter* is more common in the fringing reef than the chapeirões (personal observations), as are key herbivorous surgeonfishes and parrotfishes although the latter are smaller off SAB than in the PAB (Kikuchi et al., this volume).

An overall impression given by the AGRRA benthos indicators is that reef condition is fairly good in the chapeirões and in the northern SAB fringing reef, although the cover of live stony corals is low in the latter. Interestingly, the two reef areas that appeared to be in poor condition (i.e., the southern and, to a lesser extent, the western SAB coasts) include the sites preferred by tourists for diving and snorkeling. Thus, closer monitoring and evaluation of the relative impacts of summertime tourism and winter storm damage are warranted.

To have a more complete picture of the entire Abrolhos region, the offshore islands that are inaccessible to visitors will be assessed for comparison with the recreational dive sites at SAB. We will investigate the condition of some chapeirões located near the archipelago which are popular with tourist divers and snorkelers. We also plan to apply the AGRRA protocols in the coastal reefs that have been most severely impacted by anthropogenic effects.

## ACKNOWLEDGMENTS

The AGRRA Organizing Committee facilitated financial support as described in the Forward to this volume. The authors acknowledge the logistical support offered by the Abrolhos National Marine Park during the field work and the financial support of the Geology Graduate course of Federal University of Bahia headed by Dr. Geraldo S. Vilas Boas. The AGRRA editorial committee offered important suggestions to the final version of this manuscript.

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Table 1. Site information for AGRRA stony coral and algal surveys in the Abrolhos, Brazil.

Site name	Site code	Relative exposure/ Reef type	Latitude (° S)	Longitude (° W)	Survey date(s)	Depth (m)	Benthic transects (#)	≥25 cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)
<b><i>Santa Bárbara Island</i></b>									
South 1	SAB1	exposed/fringing	17 58.89	38 41.71	Mar 28 00	4	10	3	3.5 ± 1.3
South 2	SAB2	exposed/fringing	17 57.82	38 42.06	Mar 29 00	5.5	13	4	6.5 ± 4.3
South 3	SAB3	exposed/fringing	17 57.92	38 42.22	Mar 30 00	5.5	10	5.5	8.0 ± 6.3
Leeward 1	SAB4	sheltered/fringing	17 57.75	38 42.40	Mar 31-Apr 1 00	3.5	18	10.5	7.5 ± 5.0
Leeward 2	SAB5	sheltered/fringing	17 57.85	38 42.20	Apr 1-2 00	4.5	12	12	12.0 ± 4.9
North 3	SAB6	exposed/fringing	17 57.60	38 42.11	Apr 4-5 00	3.5	10	6	7.0 ± 5.2
North 2	SAB7	exposed/fringing	17 57.68	38 41.98	Apr 5-6 00	4.5	10	12.5	11.5 ± 6.1
North 1	SAB8	exposed/fringing	17 58.04	38 41.54	Apr 9-10 00	4	10	7	7.0 ± 4.3
<b><i>Parcel dos Abrolhos</i></b>									
Chapeirões 2	PAB2	exposed/chapeirões	17 59.00	38 40.04	Mar 21 00	6.5	9	11	20.0 ± 10.5
Chapeirões 3	PAB3	exposed/chapeirões	17 59.52	38 40.17	Mar 22 00	6.5	10	11	23.0 ± 8.1
Chapeirões 4	PAB4	exposed/chapeirões	17 58.08	38 39.34	Mar 23 00	6.5	10	12	22.0 ± 7.5
Chapeirões 5	PAB5	exposed/chapeirões	17 57.31	38 39.20	Mar 24 00	6.0	11	9.5	22.5 ± 6.4
Chapeirões 6	PAB6	exposed/chapeirões	17 56.40	38 39.36	Mar 25 00	6.0	11	9.5	14.5 ± 5.7

Table 2. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 25$  cm diameter) by site in the Abrolhos, Brazil.

Site name/ Site code	Stony corals			Partial-colony mortality (%)			Stony corals (%)					
	#	Diameter (cm)	Height (cm)	Recent	Old	Total	Recent mortality	Old mortality	Total mortality	Standing dead + Diseased	Bleached	
<b><i>Santa Bárbara Island</i></b>												
South 1/SAB1	30	42.0 $\pm$ 17.3	14.0 $\pm$ 11.7	10.0 $\pm$ 7.1	24.5 $\pm$ 26.2	25.5 $\pm$ 27.2	6.5	73.5	73.5	0	3.5	
South 2/SAB2	50	41.0 $\pm$ 16.5	23.5 $\pm$ 16.3	0	17.0 $\pm$ 15.7	17.0 $\pm$ 15.7	0	64	64	0	2	
South 3/SAB3	54	47.5 $\pm$ 21.7	20.0 $\pm$ 14.3	3.0 $\pm$ 2.8	23.5 $\pm$ 20.6	24.0 $\pm$ 20.6	3.5	79.5	79.5	0	0	
Leeward 1/SAB4	188	46.5 $\pm$ 20.0	20.5 $\pm$ 13.7	2.5 $\pm$ 2.1	31.5 $\pm$ 24.9	32.0 $\pm$ 24.8	1.5	85.5	84.5	0	2.5	
Leeward 2/SAB5	144	37.0 $\pm$ 11.4	17.0 $\pm$ 18.0	3.0 $\pm$ 2.0	17.0 $\pm$ 18.1	17.0 $\pm$ 18.0	3.5	80	82.5	0	2	
North 3/SAB6	58	45.0 $\pm$ 23.9	26.0 $\pm$ 14.6	5	21.0 $\pm$ 20.9	21.0 $\pm$ 21.3	1.5	74	74	0	0	
North 2/SAB7	126	43.0 $\pm$ 19.7	25.0 $\pm$ 14.6	5	20.5 $\pm$ 19.3	21.0 $\pm$ 19.3	1.5	75.5	75.5	0	0	
North 1/SAB8	72	36.0 $\pm$ 13.1	23.0 $\pm$ 9.6	3	21.0 $\pm$ 22.9	20.5 $\pm$ 22.8	1.5	78	79	0	0	
<b><i>Parcel dos Abrolhos</i></b>												
Chapeirões 2/PAB2	101	38.5 $\pm$ 16.2	26.5 $\pm$ 15.4	6.5 $\pm$ 2.5	22.5 $\pm$ 23.8	22.5 $\pm$ 23.5	4	62.5	64.5	0	7	
Chapeirões 3/PAB3	110	39.0 $\pm$ 17.8	30.5 $\pm$ 21.7	6.5 $\pm$ 2.9	19.0 $\pm$ 19.0	18.5 $\pm$ 18.8	3	56.5	58	0	2	
Chapeirões 4/BAP4	121	48.0 $\pm$ 36.9	30.0 $\pm$ 19.9	5	18.5 $\pm$ 20.3	18.5 $\pm$ 20.3	1	58.5	58.5	0	1.5	
Chapeirões 5/PAB5	104	39.0 $\pm$ 17.6	24.0 $\pm$ 20.5	5.0 $\pm$ 0.0	23.0 $\pm$ 23.4	23.0 $\pm$ 23.4	3	43.5	43.5	0	4	
Chapeirões 6/PAB6	106	47.5 $\pm$ 24.3	37.0 $\pm$ 27.1	5.0 $\pm$ 0.0	19.5 $\pm$ 20.0	18.5 $\pm$ 19.5	4.5	35	38.5	0	3	

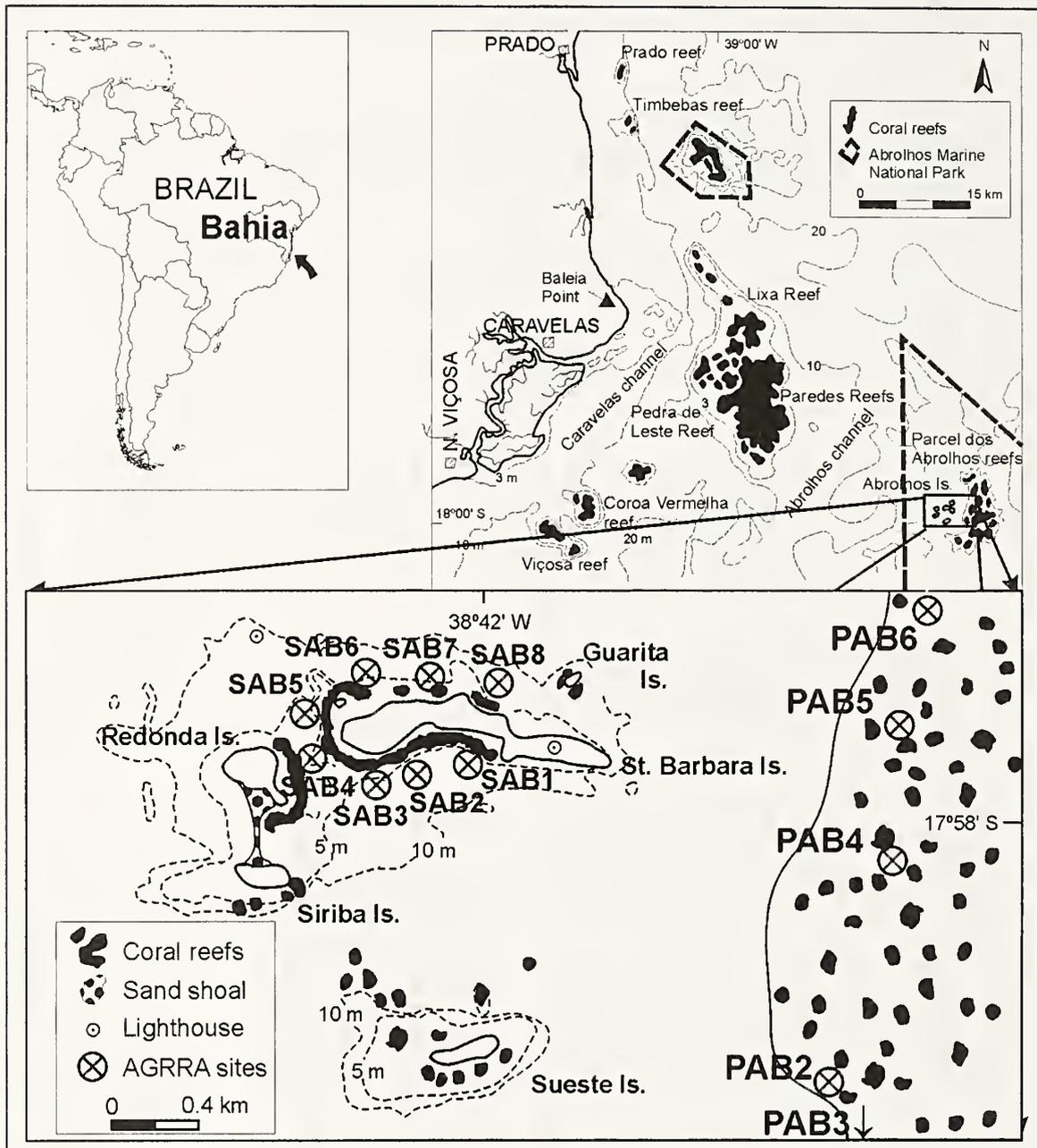
Table 3. Size and condition (mean  $\pm$  standard deviation) of all *Mussismilia braziliensis* ( $\geq 25$  cm diameter) by site in the Abrolhos, Brazil.

Site name/ site code	<i>M. braziliensis</i>		Partial-colony mortality (%)		<i>M. braziliensis</i> (%)				
	#	% of total	Diameter (cm)	Recent	Old	Recent mortality	Old mortality	Total mortality	Bleached
<b><i>Santa Bárbara Island</i></b>									
South 1/SAB1	20	66.5	45.5 $\pm$ 18.4	0	25.0 $\pm$ 31.3	0	70	70	0
South 2/SAB2	33	66	45.0 $\pm$ 16.3	0	14.5 $\pm$ 16.0	0	72.5	72.5	0
South 3/SAB3	40	74	50.5 $\pm$ 23.9	3.0 $\pm$ 2.8	20.5 $\pm$ 20.8	5	77.5	77.5	0
Leeward 1/SAB4	172	91.5	47.0 $\pm$ 19.4	2.5 $\pm$ 2.1	31.5 $\pm$ 25.8	1.5	85	85	1.5
Leeward 2/SAB5	118	82	37.5 $\pm$ 11.6	5.0	16.0 $\pm$ 17.2	1.5	83	84	1.5
North 3/SAB6	57	98.5	45.5 $\pm$ 24.1	5.0	21.5 $\pm$ 21.1	2	73.5	73.5	0
North 2.SAB7	114	90.5	43.5 $\pm$ 20.5	5.0	21.5 $\pm$ 19.9	2	74.5	74.5	0
North 1/SAB8	55	76.5	36.5 $\pm$ 14.4	3.0	20.0 $\pm$ 23.0	2	76.5	78	0
<b><i>Parcel dos Abrolhos</i></b>									
Chapeirões 2/PAB2	67	66.5	37.5 $\pm$ 15.9	5.0 $\pm$ 0	25.5 $\pm$ 26.2	4.5	71.5	73	7.5
Chapeirões 3/PAB3	81	73.5	38.0 $\pm$ 15.0	6.5 $\pm$ 2.9	17.0 $\pm$ 18.4	3.5	59.5	61.5	2.5
Chapeirões 4/BAP4	82	68	41.5 $\pm$ 15.4	0	19.0 $\pm$ 21.0	0	56	56	0
Chapeirões 5/PAB5	77	74	38.0 $\pm$ 13.3	5.0 $\pm$ 0	21.5 $\pm$ 21.2	4	45.5	45.5	1.5
Chapeirões 6/PAB6	80	75.5	47.0 $\pm$ 21.4	5.0 $\pm$ 0	18.5 $\pm$ 18.2	5	35	39	2.5

Table 4. Algal characteristics (mean  $\pm$  standard deviation), density of stony coral recruits and *Diadema antillarum* by site in the Abrolhos, Brazil.

Site name/ site code	Quadrats #	Relative abundance (%)			Macroalgal		Recruits (#/0.625 m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )
		Macroalgae	Turf algae	Crustose coralline algae	Height (cm)	Index <sup>1</sup>		
<b><i>Santa Bárbara Island</i></b>								
South 1/SAB1	50	13.0 $\pm$ 17.9	76.5 $\pm$ 17.7	10.5 $\pm$ 11.1	2.5 $\pm$ 0.9	36	0.48	0
South 2/SAB2	62	13.5 $\pm$ 18.0	78.0 $\pm$ 21.0	8.2 $\pm$ 7.5	3.5 $\pm$ 0	49	0.65	0
South 3/SAB3	50	17.5 $\pm$ 15.6	72.5 $\pm$ 15.0	10.0 $\pm$ 8.2	4.0 $\pm$ 0	69	0.82	0
Leeward 1/SAB4	90	13.5 $\pm$ 17.7	78.5 $\pm$ 21.5	8.0 $\pm$ 14.5	2.5 $\pm$ 0	34	0.62	0
Leeward 12/SAB5	60	36.0 $\pm$ 31.6	56.0 $\pm$ 30.2	8.0 $\pm$ 9.3	2.5 $\pm$ 0	94	0.95	0
North 3/SAB6	50	<0.5 $\pm$ 0.7	93.5 $\pm$ 8.3	6.5 $\pm$ 8.4	2.0 $\pm$ 0	<1	0.88	0
North 2/SAB7	50	7.0 $\pm$ 12.8	82.5 $\pm$ 20.4	10.5 $\pm$ 17.1	2.5 $\pm$ 0	16	1.84	0
North 1/SAB8	50	58.0 $\pm$ 28.9	37.0 $\pm$ 27.4	5.5 $\pm$ 7.9	2.5 $\pm$ 14.0	133	0.4	0
<b><i>Parcel dos Abrolhos</i></b>								
Chapeirões 2/PAB2	50	7.0 $\pm$ 21.5	70.0 $\pm$ 29.6	22.5 $\pm$ 23.9	1.0 $\pm$ 0.8	7	2.9	0
Chapeirões 3/PAB3	50	4.5 $\pm$ 11.2	59.5 $\pm$ 26.3	36.0 $\pm$ 23.8	1.0 $\pm$ 0.5	6	2	0
Chapeirões 4/PAB4	50	3.0 $\pm$ 6.8	77.5 $\pm$ 21.7	19.5 $\pm$ 19.7	1.5 $\pm$ 0.4	5	2.1	0
Chapeirões 5/PAB5	50	4.5 $\pm$ 11.7	65.5 $\pm$ 32.5	30.0 $\pm$ 30.0	2.0 $\pm$ 0.9	9	2.2	0
Chapeirões 6/PAB6	50	<0.5 $\pm$ 0.5	75.0 $\pm$ 26.8	25.0 $\pm$ 26.5	1.0 $\pm$ 0	<1	2.7	0

<sup>1</sup>Macroalgal index = % relative abundance of macroalgae x macroalgal height



**Figure 1.** AGRRA survey sites off Santa Bárbara Island (SAB) and in the Parcel dos Abrolhos chapeirões (PAB), Abrolhos, Brazil. See Table 1 for site codes.

## **RAPID ASSESSMENT OF THE ABROLHOS REEFS, EASTERN BRAZIL (PART 2: FISH COMMUNITIES)**

BY

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### **ABSTRACT**

The first rapid assessment of fish communities in the Abrolhos Marine National Park was undertaken in three offshore chapeirões and at eight sites in the fringing reef off Santa Bárbara Island. Fish communities in the two habitat types differed at both the family and species levels. Total species numbers were higher in the chapeirões (38-40/site) than in the fringing reef (13-37/site). Total fish density was higher in the fringing reef than in the chapeirões, as were the densities of most surveyed families, except for pomacentrids, lutjanids and scarids. The most common size class for key herbivores (scarids  $\geq 5$  cm, acanthurids) and carnivores (lutjanids, large-sized serranids) was 11-20 cm. Herbivores grazing dead skeletal areas may accidentally damage stony corals, particularly in the chapeirões. Limited impacts from sewage discharge may affect the northern coast of Santa Bárbara Island; tourists or artisanal fishers may have reduced the density of carnivores in this fringing reef.

### **INTRODUCTION**

Brazilian coral reefs, which are the southernmost reefs in the western Atlantic Ocean, extend approximately 2,000 km between 0°50' S and 19° S latitude. Although they have been studied for over a century, many reef areas are still poorly known and there are few quantitative assessments of their condition (see Leão et al., 1999). The Abrolhos reef complex in the southern part of the eastern state of Bahia (between 17°S and 19°S) contains the largest and most thoroughly investigated coral reefs in the western South Atlantic (Fig. 1). The reefs are scattered over an area of approximately 6,000 km<sup>2</sup> on the inner and middle continental shelf of the Abrolhos Bank. There are three reef

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types: isolated nearshore bank reefs of varied shapes (e.g., elongated, circular) and sizes (<1 km to ~20 km) in depths of 10-20 m, offshore shallow (<10 m deep) fringing reefs that parallel the shorelines of the five volcanic islands of the Abrolhos Archipelago; and “chapeirões”— huge (15 to >25 m high), isolated mushroom-shaped reefs found in water depths greater than 20 m.

The northern coasts of the Abrolhos islands are exposed to relatively high wave energies between September and February, their southern sides to somewhat larger waves from April to August, while the western (leeward) coasts are protected from major wind trends. The fringing reef flats extend about 30 m from shore, are poorly developed, and, during low tides, are subaerially exposed. To seaward, the reefs gradually slope to their edges at depths of 4-5 m where they drop off to a sandy bottom at 8-10 m. The windward Parcel dos Abrolhos (PAB) chapeirões are located about 2 km east of the Abrolhos islands (Fig. 1). Due to the limited penetration of light beneath the overhangs around their sides, the maximum development of stony corals is at depths of 5-8 m on the crests of the chapeirões.

A marine protected area was established in the Abrolhos reef complex in 1983 in recognition of its unique reef morphology, the presence of endemic scleractinians, and because most Brazilian reefs occur in this region. As the focus of the Abrolhos Marine National Park (AMNP) was to preserve the isolated chapeirões, ninety percent is located approximately 60 km from the mainland in the offshore reefs, with only 10% of the Park in the coastal zone. Meanwhile, as urban expansion and deforestation have accelerated, the coastal reefs are being exposed to increasing amounts of river runoff and untreated sewage discharges. However, the offshore reefs are located beyond the reach of the turbid coastal waters.

It is important to note that endemism of fish species in the Brazilian province is about 18% to 20% (Floeter and Gasparini, 2000, 2001) and the Abrolhos area is the southernmost geographic limit of some endemic tropical Brazilian species (e.g., *Dasyatis marianne*, Gomes et al., 2000; Moura et al., 2001). Almost 20 years have passed since the AMNP was established, yet there have been only a few assessments of its fish communities (Telles, 1998; Ferreira and Gonçalves, 1999). While there has been, to date, no evidence of excessive harvesting in the offshore reefs, which are designated as “no-take” zones, the coastal reefs appear to have been overfished (Ferreira and Gonçalves, 1999).

Some of the fringing reefs that border the offshore Abrolhos islands have been intensively used by recreational divers and snorkelers since 1989 (Leão et al., 1994). Although tourists dive all around Santa Bárbara Island (SAB) in particular, its southern side is most heavily visited. The PAB chapeirões are less popular due to difficulty of navigation and rougher seas but its fish and coral populations are considered to be in relatively good condition (personal observations). Previous assessments of potential human disturbances in the AMNP have only referred to the anchorage of tourist boats in algal beds around the islands (Creed and Amado Filho, 1999).

Our initial purpose in using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocols in the AMNP was to collect information on the condition of the offshore chapeirões and the SAB fringing reef in order to establish standards for a long-term monitoring program in the park. The results of the benthos assessment can be found in Kikuchi et al. (this volume).

## METHODS

Fish surveys were conducted in eight fringing reef sites around SAB which are considered representative of its northern (three sites), southern (three sites) and western (two sites) coasts. The three southern sites were also strategically chosen because they are preferred recreational dive sites. Each site was located in shallow (3.5-5.5 m) water as close as possible to the seaward edge of the reef. The 6-6.5 m crests of three representative chapeirões were surveyed. Except for the leeward fringing reef, most of these sites are exposed to large waves, and the leeward sites experience strong tidal currents.

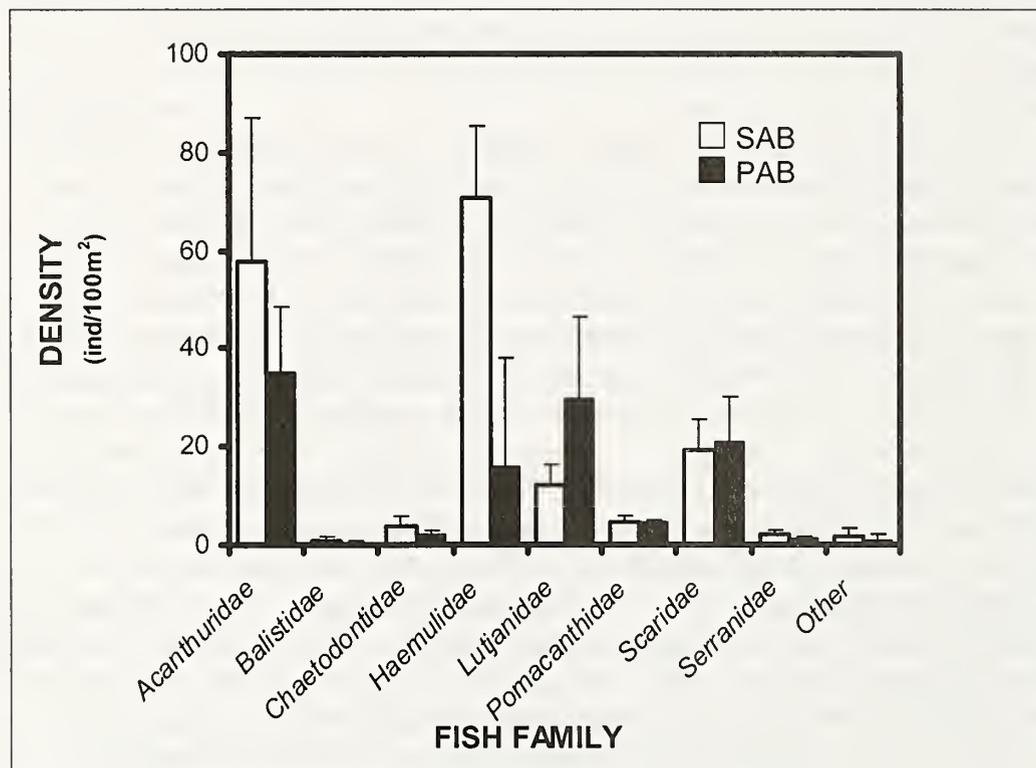
On each dive, two divers utilized the AGRRA Version 2.0 fish protocol (see Appendix One, this volume), which was modified as follows. As many of the chapeirões are less than 30 m in width, we made 30 transects/site, each 10 m long by 2 m wide. Except for the absence of a T-bar for estimating fish lengths, the transects were swum in the AGRRA manner. Juvenile grunts (haemulids) and parrotfishes (scarids) less than 5 cm in total length were tallied, as were all species of common reef fishes. Any juvenile that could not be identified to species was only assigned a genus name. The name *Scarus trispinosus* Valenciennes, 1840 was substituted for *Scarus coelestinus* which is no longer thought to occur in Brazil (Moura et al. 2001). Other identifications were based on the descriptions of Humann (1994), Smith (1997), Carvalho-Filho (1999) and Rocha and Rosa (2001).

Roving Diver Technique surveys were not made. Therefore, the number of fish species at each site is taken from the transect results. To ensure consistency, all analyses are based on the data collected by Sampaio. Juvenile grunts and parrotfishes <5 cm in length were excluded from calculations of AGRRA fish densities. Scatterplots and regression plots were used to investigate possible relationships between fish characteristics [total densities and densities of key carnivores, key herbivores, surgeonfishes (acanthurids), and parrotfishes] and some ecological parameters [live stony coral cover, height or partial-colony mortality of "large" ( $\geq 25$  cm diameter) stony corals, macroalgal index = macroalgal relative abundance x macroalgal height] that are given in Kikuchi et al. (this volume). Comparisons were made with both reef types (SAB fringing reef and the PAB chapeirões) grouped together and separately. Regression coefficients of determination and significance probability based on ANOVAs are presented for the best relationships. Densities of surgeonfishes, parrotfishes  $\geq 5$  cm, and both herbivores combined showed a rather good fit to a normal distribution model; hence fish densities and numbers of species were used without transformation. Cluster dendrogram and MDS diagrams were plotted based on Bray-Curtis similarity coefficients between the mean total density of fishes grouped by family at each site after the mean densities were square-root transformed. The complete linkage option was chosen in hierarchical agglomerative clustering (Clarke and Warwick, 1994).

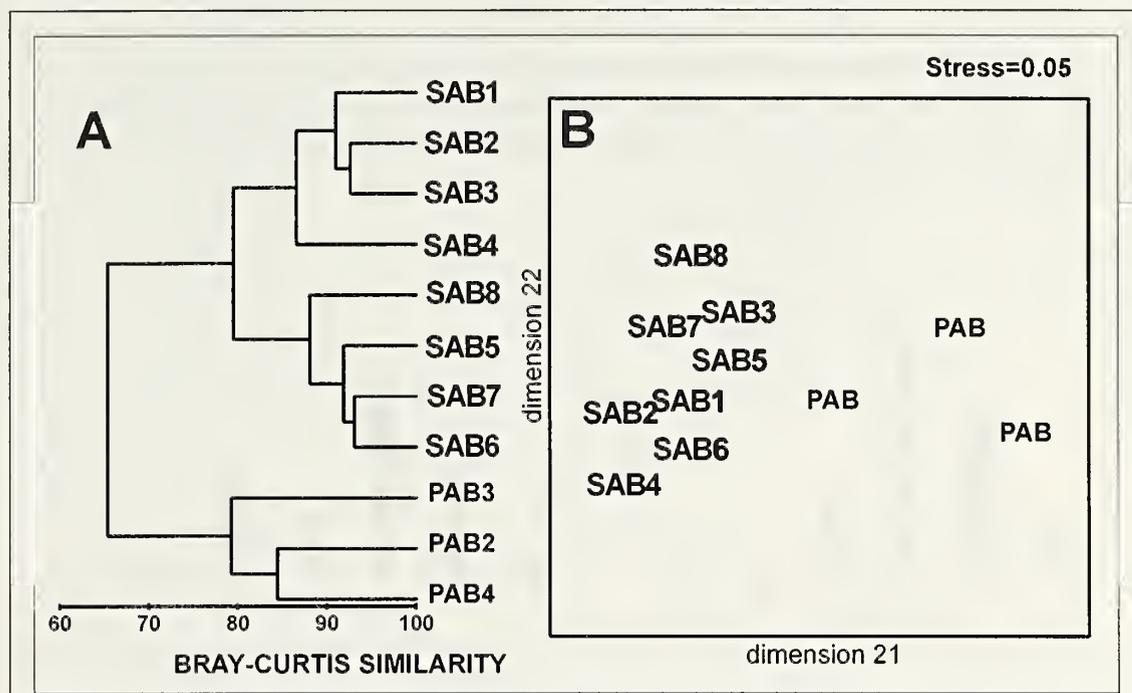
## RESULTS

The total fish community at many (7/11) sites had over 30 species of common reef fishes, and three of the remaining sites had more than 20 species (Table 1). Total species numbers were consistently higher at 6.5 m in the chapeirões (39-40/site,  $n=3$ ) than at 3.5-5.5 m in the fringing reef (13-37/site,  $n=8$ ). In contrast, 6/8 of the SAB sites had 17-19 of the species on the AGRRA list, whereas corresponding numbers in the PAB were somewhat lower (14-16/chapeirão).

Fish densities (AGRRA species only) were higher off SAB than in the chapeirões (Fig. 2, Table 2) as were densities of the numerically dominant grunts (all sizes) and surgeonfishes. When all sites are considered together, the small omnivorous grunt, *Haemulon aurolineatum*, had the highest density, followed by another omnivore (and non-AGRRA-listed fish) the sergeant major, *Abudefduf saxatilis*, damselfishes (*Stegastes* spp.), two surgeonfishes (*Acanthurus chirurgus* and *A. coeruleus*) and, in sixth position, the carnivorous snapper (lutjanid), *Ocyurus chrysurus* (Table 3). Densities of parrotfishes ( $\geq 5$  cm) were about 20 individuals/100 m<sup>2</sup> in both habitat types (Fig. 2). Acanthurids were more plentiful off SAB (~60 individuals/100 m<sup>2</sup>) than in the chapeirões (35 individuals/100 m<sup>2</sup>). Snapper densities in the PAB were double those in the fringing reef (30 versus 15 individuals/100 m<sup>2</sup>, respectively). Groupers were fewer than about 5 individuals/100 m<sup>2</sup> everywhere.



**Figure 2.** Mean fish density (no. individuals/100 m<sup>2</sup> ± sd) for all species (and all sizes of fish) in 11 fish families in the SAB fringing reef and PAB chapeirões, Abrolhos, Brazil. Other = *Caranx ruber*, *Caranx crissus*, *Sphyrna barracuda*.



**Figure 3.** (A) Hierarchical cluster analysis and (B) MDS ordination plot of all common reef fishes (based on belt transect data), by site in the Abrolhos, Brazil.

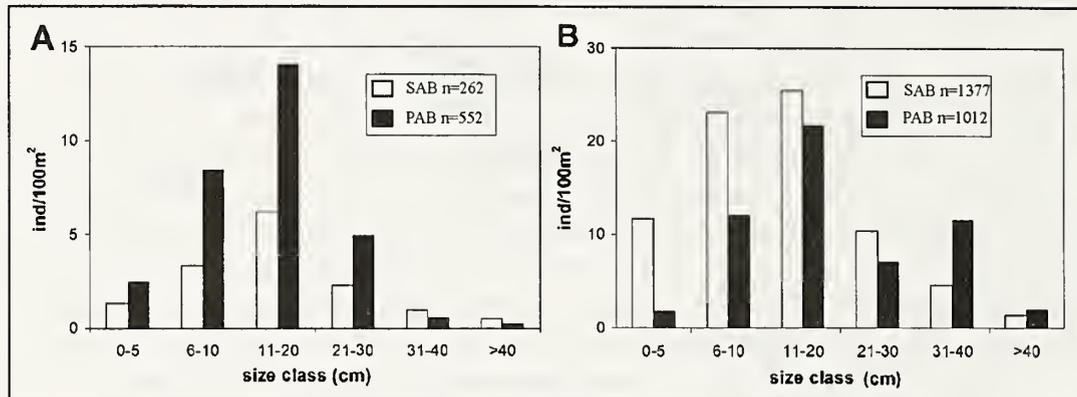
Site comparisons at the family level clearly separated the fringing reef from the chapeirões (Fig. 3) on the basis of both the Bray-Curtis similarity matrix and the MDS analyses.

The size-frequency distributions of the key carnivores (snappers and large groupers) were similar in the two habitats (Fig. 4A), the most common length class being 11-20 cm. The mean size of the key herbivores, parrotfishes  $\geq 5$  cm, and surgeonfishes (the damselfish *Microspathodon chrysurus* was not seen despite its known presence in Brazilian waters) was a little larger in the chapeirões than in the fringing reef, even though the most common length class at both was 11-20 cm (Fig. 4B). Carnivore sizes in both environments (fringing reef and chapeirões) were normally distributed, as were herbivores in the chapeirões, however, the size distribution of the SAB herbivores was slightly skewed towards larger fishes.

When all sites are included in the analysis, regardless of whether or not juvenile scarids ( $< 5$  cm in length) are retained, herbivore density showed no relationship to macroalgal index (Fig. 5A). However, when the two island sites with the highest macroalgal indices (SAB5, SAB8) and the chapeirões reefs are removed, herbivore (scarids  $\geq 5$  cm long, acanthurids) density was inversely related ( $R^2=0.800$ ,  $p=0.016$ ) to the macroalgal index of the remaining six SAB sites (Fig. 5B). A separate test of surgeonfish density versus macroalgal index in these six sites also produced good fit ( $R^2=0.910$ ,  $p=0.003$ ), but there was no relationship between parrotfish ( $\geq 5$  cm) density and macroalgal index in these same six sites. Despite the moderately good fit between herbivore density and macroalgal index in the chapeirões, the significance test indicates a high probability that the curve is horizontal ( $R^2=0.619$ ,  $p=0.423$ ).

The total number of fish species/site showed no relationship with the density of large ( $\geq 25$  cm in diameter) stony corals when all sites are taken into consideration

(Fig.6), and the positive regression for the chapeirões sites is not significant ( $R^2=0.881$ ,  $p=0.224$ ). No relationship emerged from the comparison of live stony coral cover and fish density. Neither was fish density related to the density of large stony corals nor to their average maximum height (which was used to approximate the rugosity of the reef surface, Kikuchi et al., this volume).



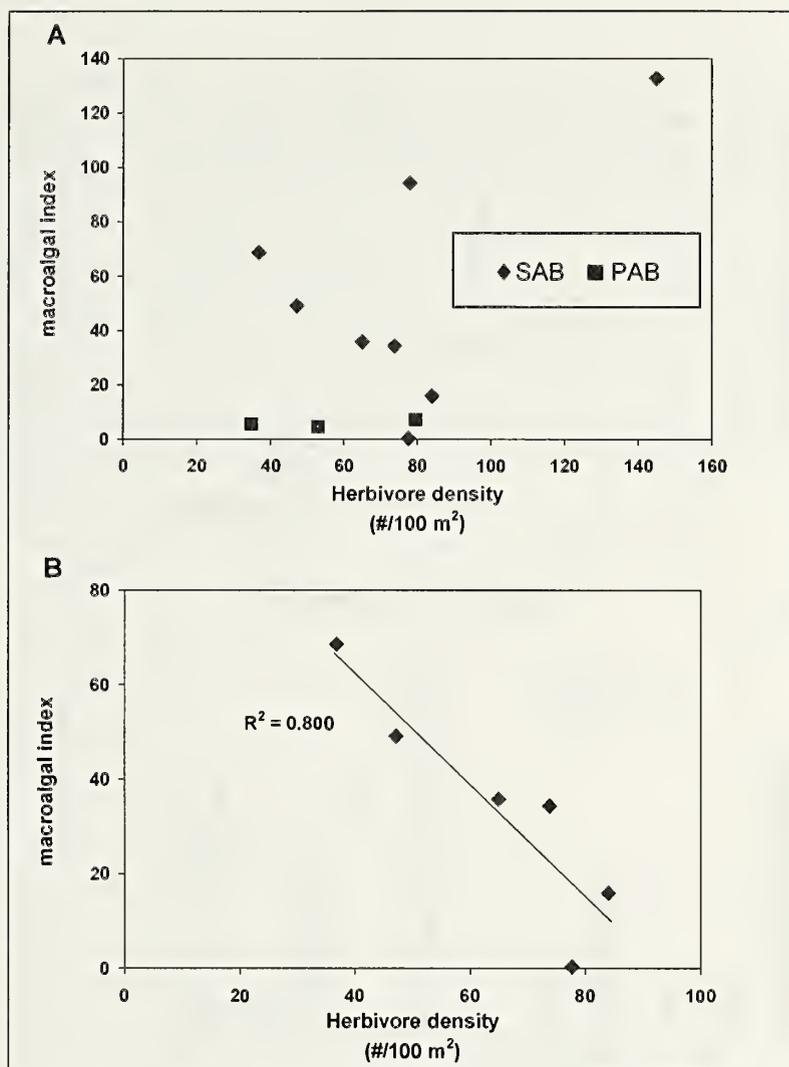
**Figure 4.** Size frequency distribution of (A) carnivores (all lutjanids, select serranids) and (B) herbivores (all acanthurids, scarids  $\geq 5$  cm) in the SAB fringing reef and PAB chapeirões, Abrolhos, Brazil.

When all sites are considered, no relationship was found between the total density of the AGRRA fishes and values for old partial-colony mortality of the large stony corals. Amongst the herbivores, only in the chapeirões (Fig. 7A, B, C) was old (but not recent) partial mortality positively related to the density of parrotfishes ( $R^2=0.978$ ,  $p=0.031$ ) and surgeonfishes ( $R^2=0.997$ ,  $p=0.037$ ).

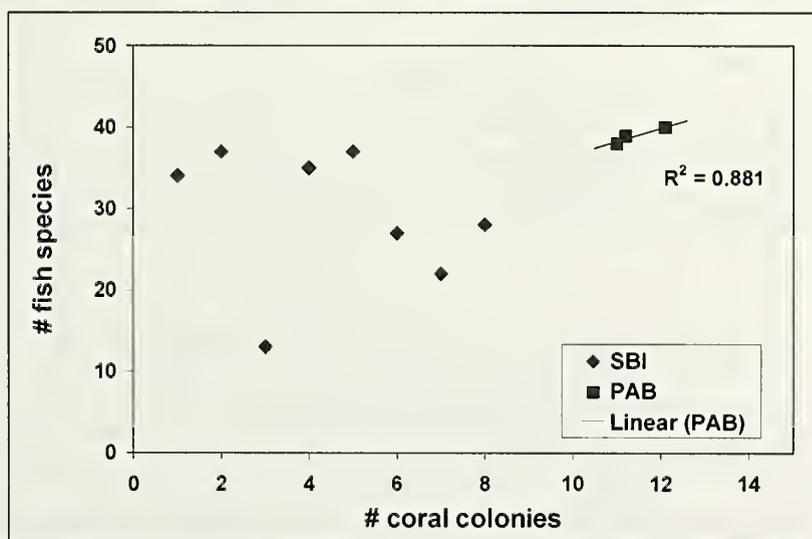
## DISCUSSION

The structure of the SAB reef fish community seems representative of the offshore insular fringing reefs in the Abrolhos Archipelago as a whole. Our results are very similar at the family level to those of Telles (1998) who, in 1995, quantified 20 common fish taxa in sites that were evenly distributed among the fringing reefs of all five islands. (Four of these sites were located in SAB, three were in Sueste, two each were in Redonda and Siriba and one was in Guarita.) Telles' (1998) belt transects, which were 50 m x 4 m, each being surveyed three times (i.e., there were three replicates of each 200 m<sup>2</sup> transect), were swum at depths of 2-6 m.

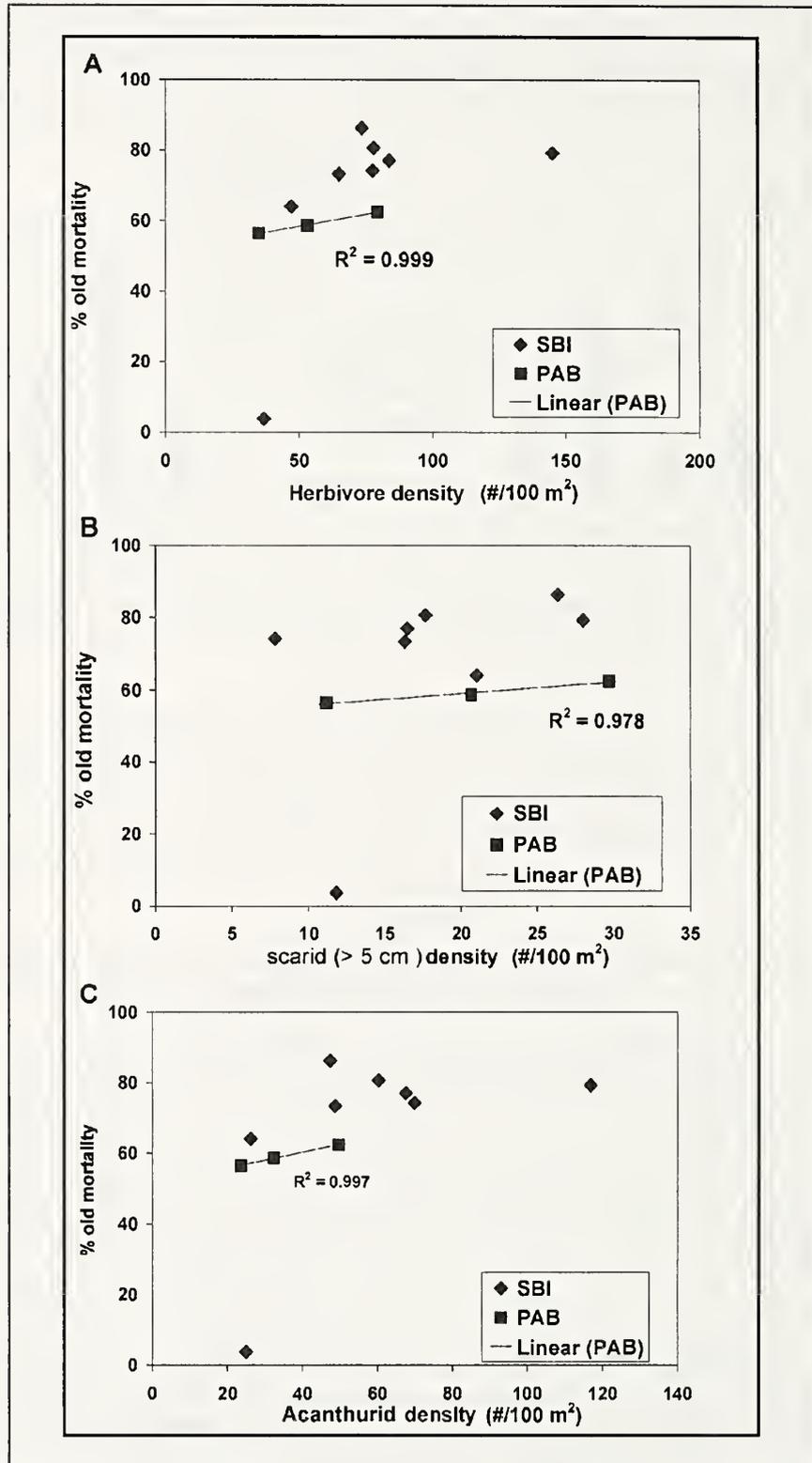
As in our SAB surveys, grunts, surgeonfishes and damselfishes were the most common families found by Telles (1998) when the data from all five islands are combined (Fig. 8). The SAB fish community also seems to be representative of the archipelago's five fringing reefs at the species level, given the similarities in the relative frequency of Telles' (1998) 20 target taxa in the two studies (Table 4). *Haemulon aurolineatum* is the most abundant of these species in both surveys.



**Figure 5.** Regression plots between mean herbivore density (no. individuals/100 m<sup>2</sup>) and macroalgal index (A) by site, (B) for six of the SAB sites (with the significant regression line), in the Abrolhos, Brazil.

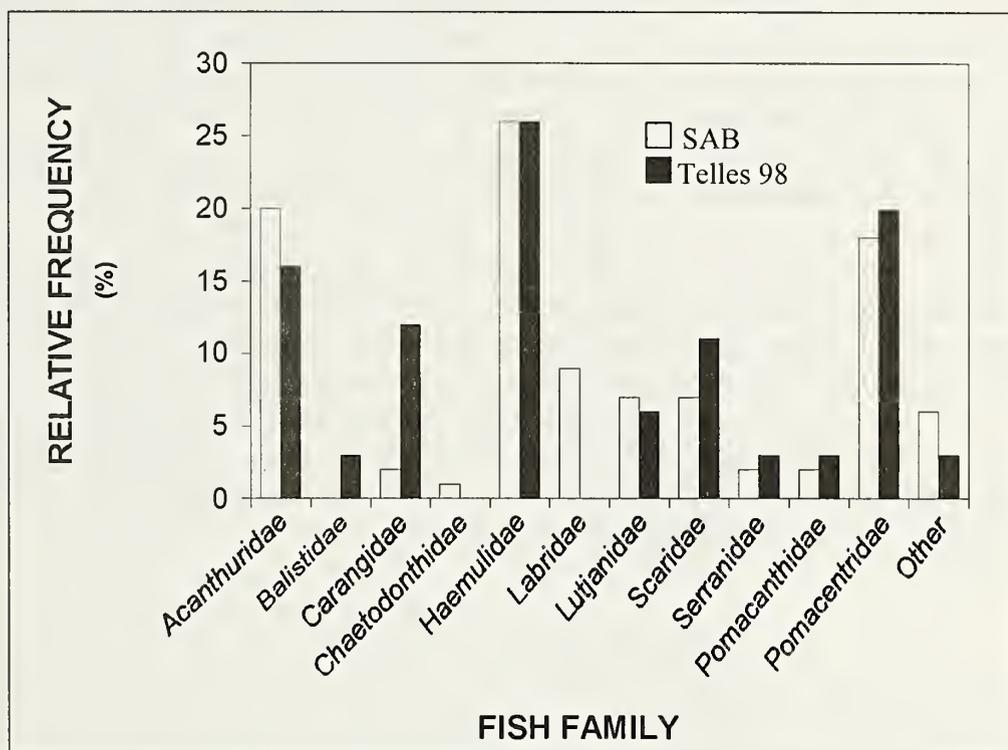


**Figure 6.** Regression plot between total fish species number and mean density of stony corals ( $\geq 25$  cm diameter)/10 m, by site in the Abrolhos, Brazil. Regression line shown for the PAB chapeirões sites is not significant.



**Figure 7.** Regression plots between old partial colony mortality of stony corals and density of (A) key herbivores, (B) scarids  $\geq 5$  cm, (C) acanthurids, by site in the Abrolhos. Regression lines shown for the PAB chapeirões sites are significant.

However, there are marked differences, at least at the family level, between the fish communities of the SAB fringing reef and the PAB chapeirões (Figs. 2, 3). The positive (but non-significant) relationship between the number of fish species and the density of large stony corals in the chapeirões might be explained by an attendant increase in the variety of habitats and niches associated with the latter. Like the large chapeirões, massive colonies of *Mussismilia braziliensis* are mushroom shaped. Their upper (illuminated) surfaces are larger in diameter than their shaded bases, which are dead and overgrown by crustose coralline algae, turf algae, macroalgae, and sponges. As the number of colonies increases, concomitant increases in the microtopographic complexity may enhance fish species richness by increasing the number of ecological interactions among stony corals, other benthic species, and fishes.



**Figure 8.** Percent relative frequency (number of fishes belonging to each family divided by the total number of fish counted) for 20 common fish taxa, by family, in the SAB fringing reef (this study) and all five offshore fringing reefs (Telles 1998) in the Abrolhos, Brazil. Other = *Caranx ruber*, *Caranx crissus*, *Sphyraena barracuda*.

Local nutrient enrichment is possibly responsible for the unexpectedly high algal biomass at the two fringing reef sites in which the inverse relationship elsewhere found between herbivores and macroalgae at SAB is lacking (Fig. 5). One of these sites (SAB5) is located near an area known as Bird's Cemetery where dead bodies of masked booby (*Sula dactylatra*), brown booby (*Sula leucogaster*), and magnificent frigatebird (*Fregata magnificens*) are frequently found. The other site (SAB8) is seaward of the seven houses occupied by Navy and Park personnel where sewage either seeps from septic tanks or is discharged directly into the ocean.

Relative to the chapeirões, snapper densities in the SAB fringing reef are strikingly low (Fig. 2) and the size-frequency distributions of herbivores are shifted toward smaller individuals (Fig. 4). As these patterns are not what we would expect in an area lacking direct human exploitation, tourists may simply chase carnivores away and/or illegal fishing (by tourists or artisanal fishermen) may occur at night.

Recent partial mortality of large ( $\geq 25$  cm in diameter) stony corals was rare and bore no relationship to fish densities. In the chapeirões, however, there seems to be a relationship between the densities of parrotfishes and surgeonfishes and old partial-colony mortality. Herbivorous fishes graze intensively on the dead surfaces of stony corals and may occasionally destroy some live tissues. Schools of the endemic greenbeak parrotfish, *Scarus trispinosus*, and of surgeonfishes are commonly seen engaged in this activity (personal observations of all co-authors).

The species richness of the common and AGRRA-listed reef fishes and the sizes of key herbivores and carnivores was higher in the PAB than off SAB, reinforcing our interpretation from the benthos assessment that the chapeirões are in better condition overall than the fringing reef (Kikuchi et al., this volume). The role of tourism in depressing carnivore, and possibly parrotfish, density in the SAB fringing reef can only be tested by assessing other areas of the AMNP, such as the islands of Sueste and Guarita, where tourist activities are prohibited.

If Telles' (1998) data are any guide, carnivorous snappers and groupers are slightly more common in the five islands as a whole than off SAB. Anthropogenic effects are possibly reflected in the AGRRA fish indicators in what is supposed to be a protected fringing reef. To test this hypothesis is one of the aims of future AGRRA assessments in the AMNP. We also plan to survey fish communities in its coastal reefs where artisanal and recreational harvesting is known to be intense. In sum, park management practices should be evaluated, and some changes may be needed to conserve the endemic species and other fishes in the Abrolhos Marine National Park.

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Table 1. Site information for AGRRA fish surveys in the Abrolhos, Brazil.

Site name	Site code	Relative exposure/ Reef type	Latitude (° S)	Longitude (° W)	Survey date(s)	Depth (m)	Depth >25 cm stony corals <sup>1</sup> (#/10 m)	Height (cm)	% live stony coral cover (mean ± sd) <sup>1</sup>	10 m fish transects (#)	Fish species (#)	
											All common	AGRRA
<b>Santa Bárbara Island</b>												
South 1	SAB1	exposed/fringing	17 58.89	38 41.71	Mar 28 00	4	3	14.0	3.5 ± 1.3	30	34	18
South 2	SAB2	exposed/fringing	17 57.82	38 42.06	Mar 29 00	5.5	4	23.5	6.5 ± 4.3	30	37	18
South 3	SAB3	exposed/fringing	17 57.92	38 42.22	Mar 30 00	5.5	5.5	20.0	8.0 ± 6.3	30	13	17
Leeward 1	SAB4	sheltered/fringing	17 57.75	38 42.40	Mar 31-Apr 1 00	3.5	10.5	20.5	7.5 ± 5.0	30	35	15
Leeward 12	SAB5	sheltered/fringing	17 57.85	38 42.20	Apr 1-2 00	4.5	12	17.0	12.0 ± 4.9	30	37	18
North 3	SAB6	exposed/fringing	17 57.60	38 42.11	Apr 4-5 00	3.5	6	26.0	7.0 ± 5.2	30	28	19
North 2	SAB7	exposed/fringing	17 57.68	38 41.98	Apr 5-6 00	4.5	12.5	25.0	11.5 ± 6.1	30	22	16
North 1	SAB8	exposed/fringing	17 58.04	38 41.54	Apr 9-10 00	4	7	23.0	7.0 ± 4.3	30	27	18
<b>Parcel dos Abrolhos</b>												
Chapeirões 2	PAB2	exposed/chapeirões	17 59.00	38 40.04	Mar 21 00	6.5	11	26.5	20.0 ± 10.5	30	39	16
Chapeirões 3	PAB3	exposed/chapeirões	17 57.52	38 40.17	Mar 22 00	6.5	11	30.5	23.0 ± 8.1	30	38	14
Chapeirões 4	PAB4	exposed/chapeirões	17 58.08	38 39.34	Mar 23 00	6.5	12	30.0	22.0 ± 7.5	30	40	16

<sup>1</sup>From Kikuchi et al. (this volume)

Table 2. Density (mean  $\pm$  standard deviation) of AGRRA fishes and macroalgal index, by site in the Abrolhos, Brazil

Site code	Herbivores (#/100 m <sup>2</sup> )			Carnivores (#/100 m <sup>2</sup> )			Macroalgal Index <sup>2</sup>
	Acanthuridae	Scaridae $\geq 5$ cm	Haemulidae $\geq 5$ cm	Lutjanidae	Serranidae <sup>1</sup>		
SAB1	48.67 $\pm$ 41.88	16.33 $\pm$ 12.04	62.33 $\pm$ 59.77	10.50 $\pm$ 7.29	2.33 $\pm$ 1.41		36
SAB2	26.17 $\pm$ 19.80	21.00 $\pm$ 12.82	72.17 $\pm$ 105.58	16.33 $\pm$ 8.34	3.83 $\pm$ 3.05		49
SAB3	25.00 $\pm$ 13.72	11.83 $\pm$ 9.01	42.83 $\pm$ 40.47	13.00 $\pm$ 18.46	2.00 $\pm$ 2.05		69
SAB4	47.50 $\pm$ 37.16	26.33 $\pm$ 17.72	55.00 $\pm$ 41.20	9.50 $\pm$ 7.62	2.83 $\pm$ 2.23		34
SAB5	60.33 $\pm$ 39.98	17.67 $\pm$ 15.46	79.00 $\pm$ 72.48	17.00 $\pm$ 8.88	1.50 $\pm$ 3.19		94
SAB6	69.83 $\pm$ 61.66	7.83 $\pm$ 7.20	45.17 $\pm$ 31.84	15.17 $\pm$ 20.27	1.33 $\pm$ 1.53		<1
SAB7	67.50 $\pm$ 36.47	16.50 $\pm$ 21.03	74.83 $\pm$ 71.88	10.83 $\pm$ 14.70	2.67 $\pm$ 3.44		16
SAB8	117.00 $\pm$ 92.26	28.00 $\pm$ 30.73	32.67 $\pm$ 56.33	5.50 $\pm$ 4.97	1.00 $\pm$ 1.41		133
PAB2	49.83 $\pm$ 40.40	29.67 $\pm$ 12.79	3.50 $\pm$ 5.18	40.00 $\pm$ 29.06	0.67 $\pm$ 0.86		7
PAB3	23.67 $\pm$ 24.57	11.17 $\pm$ 12.67	2.67 $\pm$ 1.61	9.83 $\pm$ 5.47	0.83 $\pm$ 0.88		6
PAB4	32.33 $\pm$ 31.14	20.67 $\pm$ 12.40	41.50 $\pm$ 66.58	38.83 $\pm$ 32.46	1.83 $\pm$ 2.28		5

<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.<sup>2</sup>From Kikuchi et al. (this volume)

Table 3. Twenty-five most frequently sighted fishes and the mean densities for species seen in belt transect surveys in the Abrolhos, Brazil.

Scientific name	Sighting frequency (%) <sup>2</sup>	Density (#/100 m <sup>2</sup> )
<i>Abudefduf saxatilis</i>	100	28.95
<i>Stegastes spp.</i>	100	23.77
<i>Acanthurus chirurgus</i>	100	20.06
<i>Acanthurus coeruleus</i>	100	17.79
<i>Ocyurus chrysurus</i>	100	16.88
<i>Scarus trispinosus</i> <sup>1</sup>	100	11.77
<i>Sparisoma spp.</i>	100	7.9
<i>Anisotremus virginicus</i>	100	4.89
<i>Chaetodon striatus</i>	100	3.33
<i>Holocentrus adscensionis</i>	100	2.21
<i>Acanthurus bahianus</i>	96	13.77
<i>Pomacanthus paru</i>	96	2.98
<i>Mycteroperca bonaci</i>	96	1.86
<i>Gramma braziliensis</i>	96	1.59
<i>Haemulon plumieri</i>	92	5.32
<i>Balistes vetula</i>	88	0.83
<i>Haemulon aurolineatum</i>	83	35.68
<i>Halichoeres radiatus</i>	79	1.61
<i>Elacatinus figaro</i>	75	5.68
<i>Holocanthus ciliaris</i>	75	1.05
<i>Haemulon parra</i>	71	9.73
<i>Scarus zelindae</i>	71	1.48
<i>Caranx chrysus</i>	67	1.36
<i>Caranx ruber</i>	50	0.86
<i>Caranx bartholomaei</i>	46	1.02

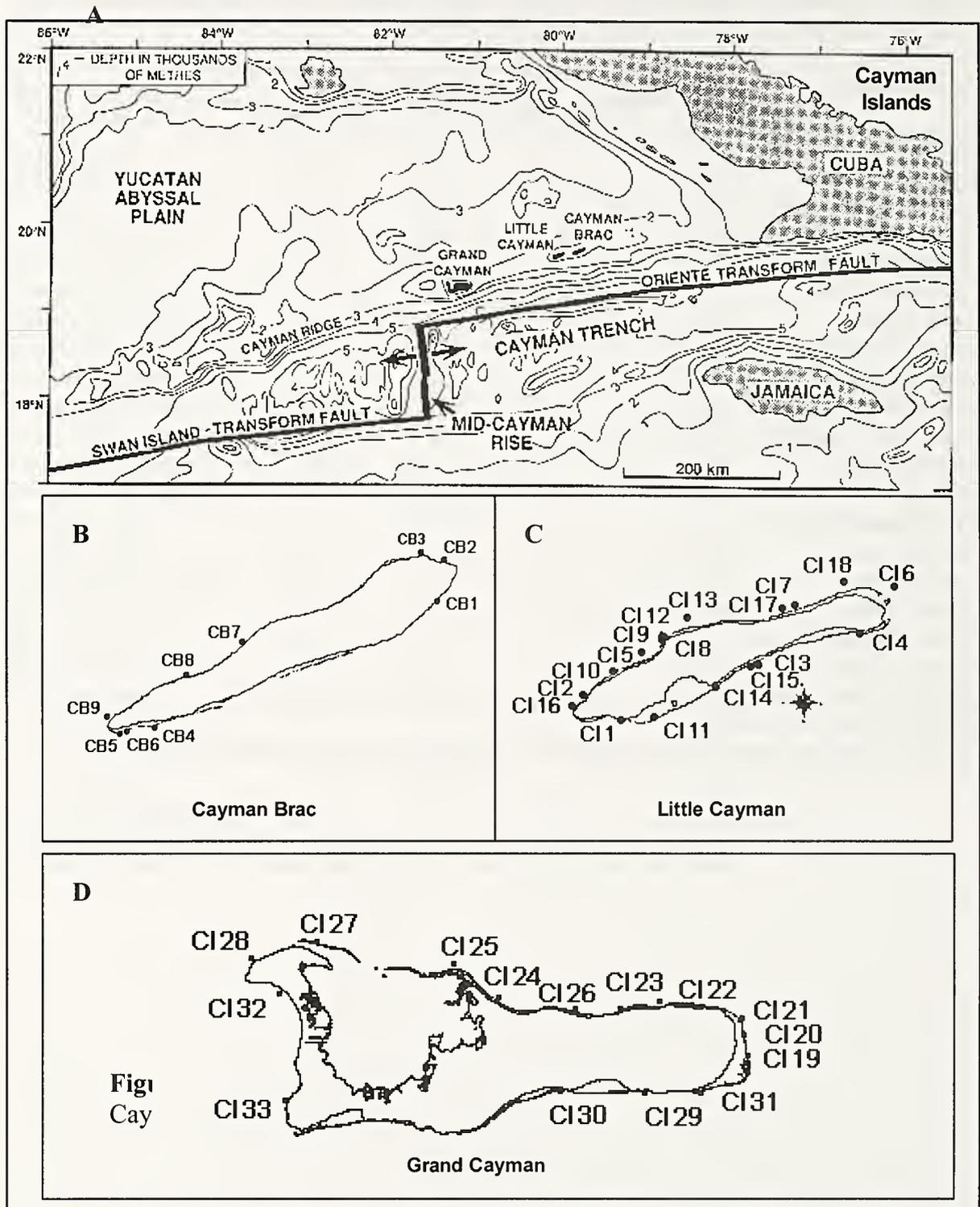
<sup>1</sup>formerly *Scarus coelestinus*.<sup>2</sup>Sighting frequency (%) = percentage of dives in which the taxon was recorded.

Table 4. Relative frequency of the most common fishes in the Abrolhos Archipelago (Telles, 1998) and in Santa Bárbara Island (this survey).

Scientific name	Relative frequency (%) <sup>2</sup>	
	(Telles, 1998)	(this survey)
<i>Haemulon aurolineatum</i>	19	14
<i>Stegastes variabilis</i>	13	9
<i>Acanthurus chirurgus</i>	11	8
<i>Scarus trispinosus</i> <sup>1</sup>	8	4
<i>Abudefduf saxatilis</i>	7	11
<i>Chaetodon striatus</i>	5	1
<i>Caranx latus</i>	5	0
<i>Ocyurus chrysurus</i>	4	7
<i>Acanthurus coeruleus</i>	4	7
<i>Pomacanthus arcuatus</i>	3	0.2
<i>Anisotremus virginicus</i>	3	2
<i>Haemulon parra</i>	3	4
<i>Sparisoma spp.</i>	3	3
<i>Mycteroperca bonaci</i>	3	1
<i>Balistes vetula</i>	3	0.3
<i>Lutjanus jocu</i>	2	0.3
<i>Caranx ruber</i>	2	0.3
<i>Holocentrus adscensionis</i>	1	1
<i>Kyphosus sectatrix</i>	1	4
<i>Sphyraena barracuda</i>	1	0.1

<sup>1</sup>formerly *Scarus coelestinus*.

<sup>2</sup>Relative frequency = number of individuals/taxon relative to the total number of individuals.



**Figure 1.** AGRRRA survey sites in Grand Cayman and Little Cayman, Cayman Islands. See Table 1 for site codes.

# STATUS OF CORAL REEFS OF LITTLE CAYMAN, GRAND CAYMAN AND CAYMAN BRAC, BRITISH WEST INDIES, IN 1999 AND 2000 (PART 1: STONY CORALS AND ALGAE)

BY

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and ROBERT GRAIFMAN<sup>4</sup>

## ABSTRACT

A benthic assessment of the isolated Cayman Islands was completed at 42 sites. Major changes in the reef community structure were documented by comparison with earlier studies. *Acropora palmata* and *A. cervicornis*, once abundant as shallow framework builders, were uncommon. Diseased stony corals were seen in >90% of the study sites, with the highest averages in Little Cayman, especially at Bloody Bay which is one of the most highly regulated marine parks in the Cayman Islands. The *Montastraea annularis* species complex accounted for two-thirds of the diseased corals which, along with other massive species, were affected largely by white-plague disease. Recent partial-colony mortality was particularly high in Grand Cayman. However, small- to intermediate-sized (<1.5 m diameter) colonies and recruits of reef framework builders (including the *M. annularis* complex) suggest a strong potential for population regeneration. Algal competition generally did not appear to be a problem for stony corals, and bleaching was insignificant, yet more prevalent, in the deeper (>10 m) sites.

## INTRODUCTION

The Cayman Islands lie in the middle of the Caribbean Sea, about 240 km south of Cuba. They are comprised of three small (12–18 km long) low-lying, limestone islands with Little Cayman (LC) 125 km northeast of Grand Cayman (GC) and just 7.5 km to the southwest of Cayman Brac (CB). LC has a permanent resident population of about 130, and remains relatively undeveloped with four small dive

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lodges and one 40-room resort. There are 2,200 residents and three dive resorts on CB. In comparison, GC has a resident population of 36,000, and numerous large resort hotels, condominiums, and dive operators along the seven-mile beach area on the western side of the island. Georgetown Harbor is an important port for cruise ships.

The Cayman Islands are protected from open-ocean long-fetch Atlantic waves by the Antilles Island Arc, Hispaniola, and Cuba. Trade winds blow predominantly from the northeast but shift periodically to the east and southeast. The eastern side of GC receives the highest wind and wave energy, whereas the western sides of all three islands are least exposed. LC and CB are oriented northeast to southwest, so the southern sides tend to be more exposed to prevailing winds and waves. Polar fronts push winter storms south during the winter months, and winds around the islands shift to blow from the north, developing into Noreasters and Norwesters.

Two shallow-sloping submarine terraces occur around the islands with a few exceptions (e.g. Bloody Bay on LC is missing the lower terrace) (Rigby and Roberts, 1976; Blanchon et al., 1997). The upper terrace extends from the shoreline to a mid-shelf scarp at depths of 8–15 m and a lower terrace from the base of this scarp to the edge of the shelf at 15–20 m. Spectacular shelf-edge “walls” occur around all three islands. Fringing reefs and boulder ramparts are nearly continuous around LC and GC (except off the western side of GC) and occur along the south side (western third) of CB. In many locations, the boulder rampart (in 0.5–5 m water depth) was formerly a reef crest dominated by *Acropora palmata* on the upper terrace along the windward coasts (east, south, north of GC and LC, south of CB). High-relief (to 12 m) spur-and-groove or buttress reefs have developed at several sites at the edge of the upper terrace in the more exposed (eastern, northeastern, southeastern) sides of the islands. At “more protected” windward locations on the northern and southern coasts, low- to medium-relief (1–3 m) spurs and grooves are found at the edge of the upper terrace. In leeward locations (western side of GC; Bloody Bay on LC), where a reef crest is lacking, there are narrow, low-to-medium relief spurs and grooves or shelf-edge reefs with poorly developed spurs and grooves, although a few high-relief, elongated spurs occur along the upper terrace in Bloody Bay.

Episodes of coral bleaching were reported in the Cayman Islands in the late 1980's, 1995, and in 1998 when nearly 80% of the corals bleached on GC (LC and CB were not surveyed) (Timothy Austin, personal communication). The last major storms to impact the Caymans were Hurricanes Allen in 1980, Gilbert in 1988, and Mitch in 1998. Hurricanes Allen and Mitch reportedly had minimal impact underwater but Gilbert damaged the shallow reefs, particularly breaking small colonies of *Acropora palmata*, at many locations around GC (Blanchon and Jones, 1997; Timothy Austin, personal communication). The extent of the impacts by these storms remains largely undocumented on LC and CB; however, boulder ramparts on many of the beaches of all three islands and reef-crest zones that are largely biodetrital provide historic evidence of the continuous impact by major storms.

A Marine Conservation Law was passed in the Cayman Islands in 1979. Marine park areas (15 km<sup>2</sup>), replenishment zone areas (15 km<sup>2</sup>), an environmental zone (17 km<sup>2</sup>) in the North Sound of GC, and animal sanctuaries/RAMSAR sites were designated in 1986 (Fig. 1A, B). Conservation regulations have not prevented

overfishing of conch and lobster, nor have they adequately protected grouper spawning grounds from fishing or turtle nesting sites from coastal development. Residents (for at least five years) can apply for licenses to use spearguns and seine nets and to capture turtles if they can prove these activities are a cultural necessity. A total of 500 speargun licenses have been issued in the Cayman Islands. Growth in the hotel industry has resulted in an influx of workers from other Caribbean countries and has resulted in increased use of fish traps (pots).

Recent amendments made to the Marine Conservation Legislation in the Cayman Islands include instituting a season for conch and limiting the catch to five per person (from 15) and 10 per boat (from 20). Grouper spawning sites are now protected, and a catch limit of 12 per boat and a minimum size of 12 inches have been set. Individuals normally resident in Cayman are permitted to line fish in these areas. Lobster catch limits have been reduced to three per person and six per boat per day, and the length of the closed season has been extended. Fish pots are now limited to a maximum of two per household, and they must now be licensed and tagged. Only Caymanians are granted fish pot licenses.

Beginning in the early 1980's, the Cayman Islands Department of Environment has installed 159 permanent moorings on GC, 56 on LC, and 54 on CB. Diving has historically been focused along the western and northwestern coasts of GC where some dive sites receive over 15,000 visitors per year. Most divers on LC visit only one location, Bloody Bay Marine Park, where there are 22 moorings over a 3 km distance. With increased human usage (fishing, coastal development, mangrove destruction, diving, etc.), concern about the survival of Cayman's coral reefs has been growing. The survey reported here was designed to rapidly assess the present status of these important resources with the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocols (Manfrino, et al., 2000) and to determine recent changes in reef community structure and condition in the various oceanographic settings by comparison with earlier accounts (as summarized by Rigby and Roberts, 1976; Roberts, 1994; Logan, 1994; Hunter, 1994). Evaluating the current distribution of *Acropora palmata* and *A. cervicornis* was a major concern because of the extensive distribution previously reported. Other objectives were to compare reefs off the more populous GC with those of its sister islands and to strategically survey sites both inside and outside of marine protected areas and those with both high diver pressure and low or no diver pressure. The Cayman Islands lie in the middle of the Caribbean Sea, far from most continental and many human influences, and thus should provide an important point of comparison for other AGRRA surveys throughout the region. A detailed report on the results of the fish assessments completed during this study can be found in Pattengill-Semmens and Semmens (this volume).

## METHODS

A total of 11 scientists and graduate assistants conducted AGRRA benthos surveys on LC and GC in June 1999 and on CB in June 2000. As relatively little information existed for LC, its major reef types were initially characterized during a reconnaissance around the entire island using a standard manta-towing technique. We

recorded locations where coral cover was high and where we could potentially survey typical reefs in the recommended water depths for the AGRRA protocol. Except for the southwest coast of GC, the representative sites that were studied on all sides of the islands were strategically located both inside and outside of marine parks and replenishment zones. At LC there was a higher concentration of sites in heavily dived areas (Bloody Bay, Jackson Point). Published maps and aerial photographic images made available from the Cayman Islands Department of Environment 1994 photo series and input from knowledgeable boat captains were essential aides in site selection. All scientists participated in two days of training to standardize measuring techniques and identification of species. Humann (1993) was our primary identification guide.

Following AGRRA protocol Version 2.2 guidelines (see Appendix One, this volume), two zones were examined: the shallow (1-5 m) *Acropora palmata* reef crest, and the generally well-developed framework reefs in depths of ~5-16 m on the upper terrace. *Acropora palmata* in the reef-crest zone was sparse, or completely dead, and although several sites were investigated, only one (CI11) was surveyed. Forty-one surveys were conducted on the upper terrace. The minimum size of surveyed stony corals was 10 cm and coral size was recorded to the nearest 10 cm. *Agaricia* was not identified to the species level. Damselfish in the vicinity of the surveyed corals were counted. All tiny corals ( $\leq 2$  cm) were included in the counts of "recruits." Sediment was removed from crustose coralline algae before estimating their abundance in the quadrats. Algal abundances at CB were estimated using the AGRRA Version 3.1 protocols given in Appendix One, except they included turf algae.

For the purposes of analysis, the upper terrace sites were arbitrarily divided by depth into 18 "shallower" reefs that were located in  $\leq 10$  m and 23 "deeper" reefs that were  $> 10$  m deep. Few geomorphological or habitat differences were associated with these depth divisions.

## RESULTS

A total of 5,531 stony corals (all  $\geq 10$  cm in diameter) in 486 transects and 2,407 algal quadrats were assessed during surveys off LC (18 sites) and GC (15 sites) in June, 1999, and off CB (9 sites) in June, 2000 (Table 1). Average live stony coral cover varied between ~14% in CB and 23% in LC (Table 1).

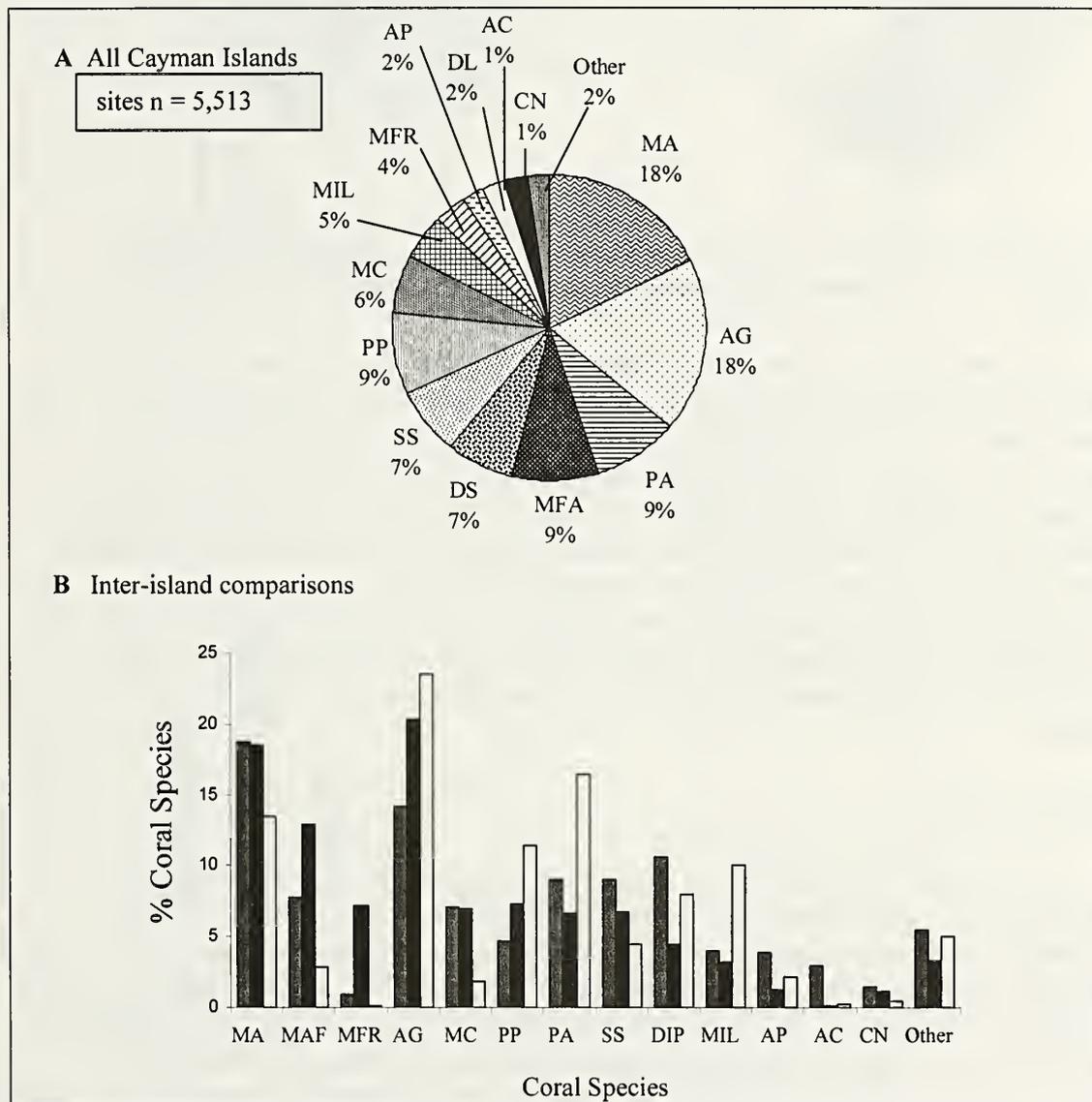
### Stony Coral Species

Thirty-three scleractinian and hydrocoral species with colonies of at least 10 cm diameter were encountered along the transect lines. However, roving surveys indicated a total of 37 species in the vicinity of the survey sites (including species characteristically not growing larger than 10 cm).

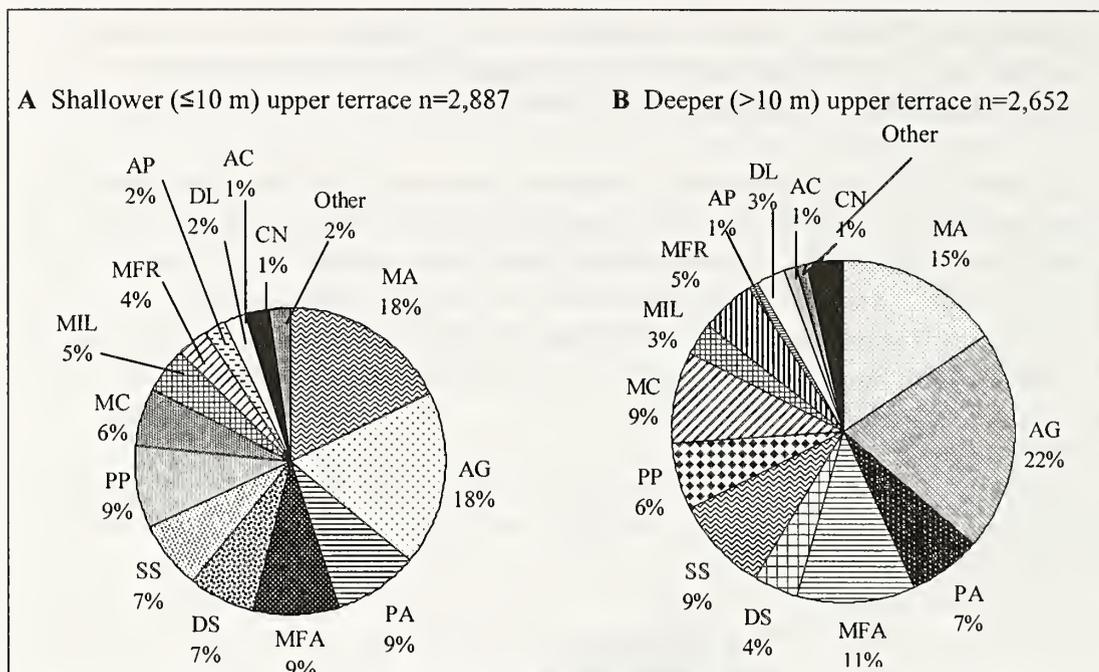
*Reef crests.* *Acropora palmata*, the once-dominant shallow reef-crest coral (Rigby and Roberts, 1976), was found only in rare patches just seaward of what is now a rubble rampart. Extensive groves of "standing dead" colonies (completely dead

but still in growth position) and “stumps” occurred in the windward reefs and off the north sides of GC and LC. Only one windward site on LC (CI11) was found with a zone of live *A. palmata* that was extensive enough to survey.

*Upper terrace reefs.* Stony corals ( $\geq 10$  cm in diameter) off all three Cayman Islands were dominated by the *Montastraea annularis* species complex and *Agaricia* spp. (Fig. 2A). Inter-island comparisons indicate that *Montastraea annularis*, *M. faveolata* and *M. franksi* were numerically predominant in LC ( $>38\%$ ) and GC ( $\sim 28\%$ ), but *Agaricia* was predominant in CB (Fig. 2B). The *Montastraea annularis* complex generally dominated the shallower reefs while *Agaricia* spp. were predominant in the deeper reefs (Fig. 3A, B).



**Figure 2.** (A) Species composition and mean relative abundance of all stony corals ( $\geq 10$  cm diameter) in all survey sites off the Cayman Islands. (B) Interisland comparison of 14 stony coral species. AC = *Acropora cervicornis*, AP = *A. palmata*, AG = *Agaricia agaricites*, CN = *Colpophyllia natans*, DIP = *Diploria* spp., DL = *D. labyrinthiformis*, DS = *D. strigosa*, MIL = *Millepora* spp., MA = *M. annularis*, MC = *M. cavernosa*, MAF = *M. faveolata*, MFR = *M. franksi*, PA = *Porites astreoides*, PP = *P. porites*, SS = *Siderastrea siderea*.



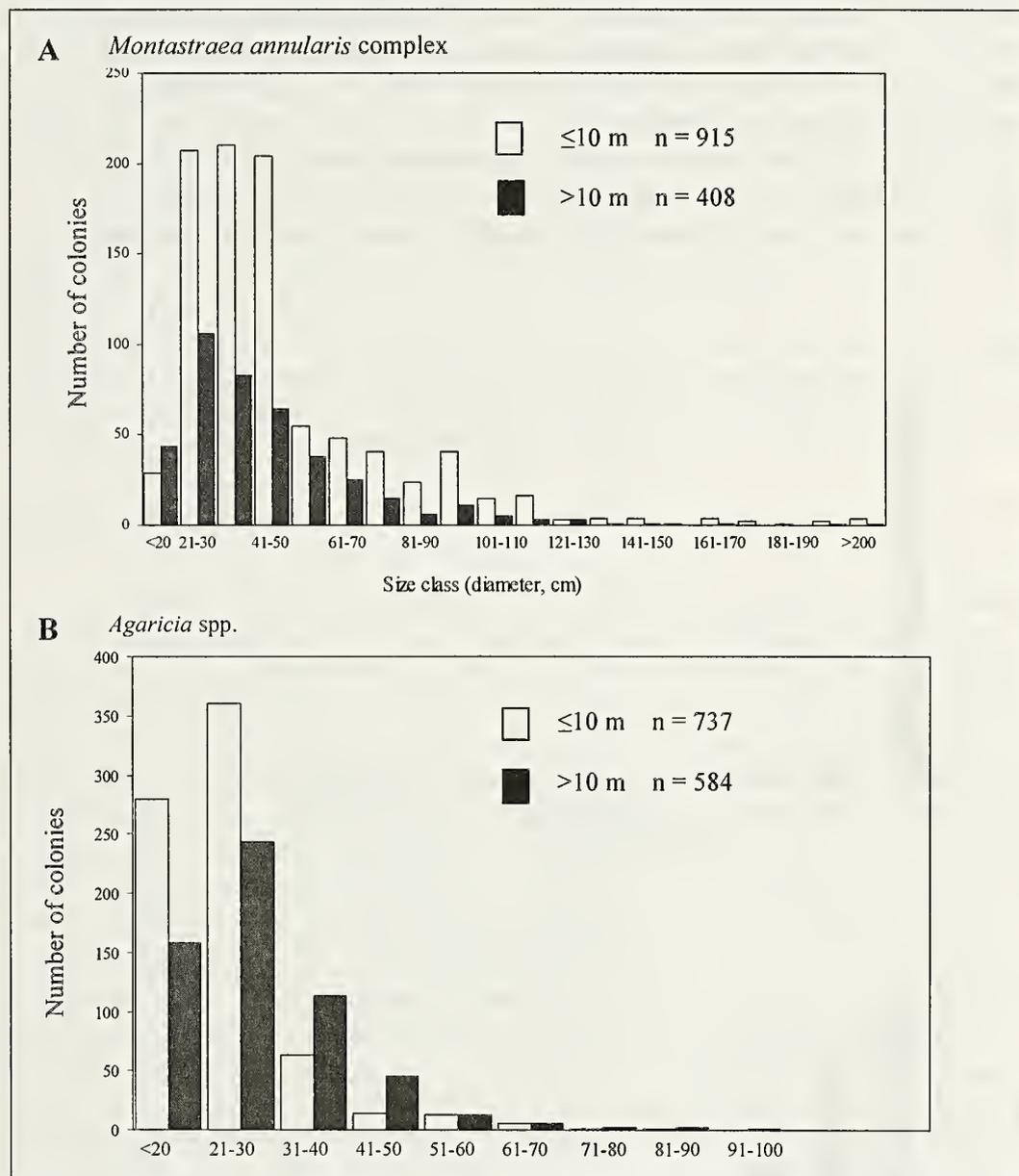
**Figure 3.** Species composition and mean relative abundance of all stony corals ( $\geq 10$  cm diameter) in (A) shallower ( $\leq 10$  m), (B) deeper ( $>10$  m) upper terrace reefs off the Cayman Islands. See Figure 2 for species codes.

*A. palmata* was present in five of the six most exposed windward reefs off GC (CI19, CI20, CI21, CI29, CI 31) and in one protected windward reef along its northern coast (C124). It occurred in three of seven windward reefs off LC (CI3, CI11, CI15) and four of five windward reefs off CB (CB2, CB4, CB5, CB6). Live corals were generally less common than standing dead colonies. Although thickets of *Acropora cervicornis* were seen more commonly landward of the reef crests, at least a few small ( $<1$  m diameter) colonies were present in the shallower parts of the spur-and-groove formations at most (10/15) reefs off GC. *A. cervicornis* was rare in LC and CB. Piles of broken branches and dead, intact thickets were found in spur-and-groove zones, on hardgrounds, and generally at locations where some live colonies occurred.

### Stony Coral Size

The average diameter of the  $\geq 10$  cm stony corals (Table 2) was essentially identical off LC ( $\sim 45$  cm) and GC ( $\sim 42$  cm) but smaller off CB ( $\sim 32$  cm). Average coral height was somewhat greater off LC (nearly 30 cm), where *Montastraea faveolata* was numerically more abundant than off the other two islands (Table 2). The largest corals were *Montastraea faveolata* (with heights as great as 320 cm and diameters up to 400 cm), but there were also large (up to 390 cm tall) colonies of *Colpophyllia natans*, *Diploria clivosa*, *M. annularis*, and *Acropora palmata*. The diameters of live *A. palmata*, however, were all smaller ( $<1.5$  m) than the standing dead colonies (1.5–2 m in diameter). Surveyed *M. annularis* occurred most frequently in the 21–60 cm diameter size range (Fig. 4A) and colonies that were more than 1 m in diameter were more common in LC and GC than in CB. Colonies of *Agaricia* spp.

were most abundant in the 21–30 cm class size, and rarely exceeded 70 cm in diameter (Fig. 4B). No correlation between coral size (height or diameter, all  $\geq 10$  cm in diameter) and water depth or level of exposure to prevailing winds was identified (Tables 2 and 3).



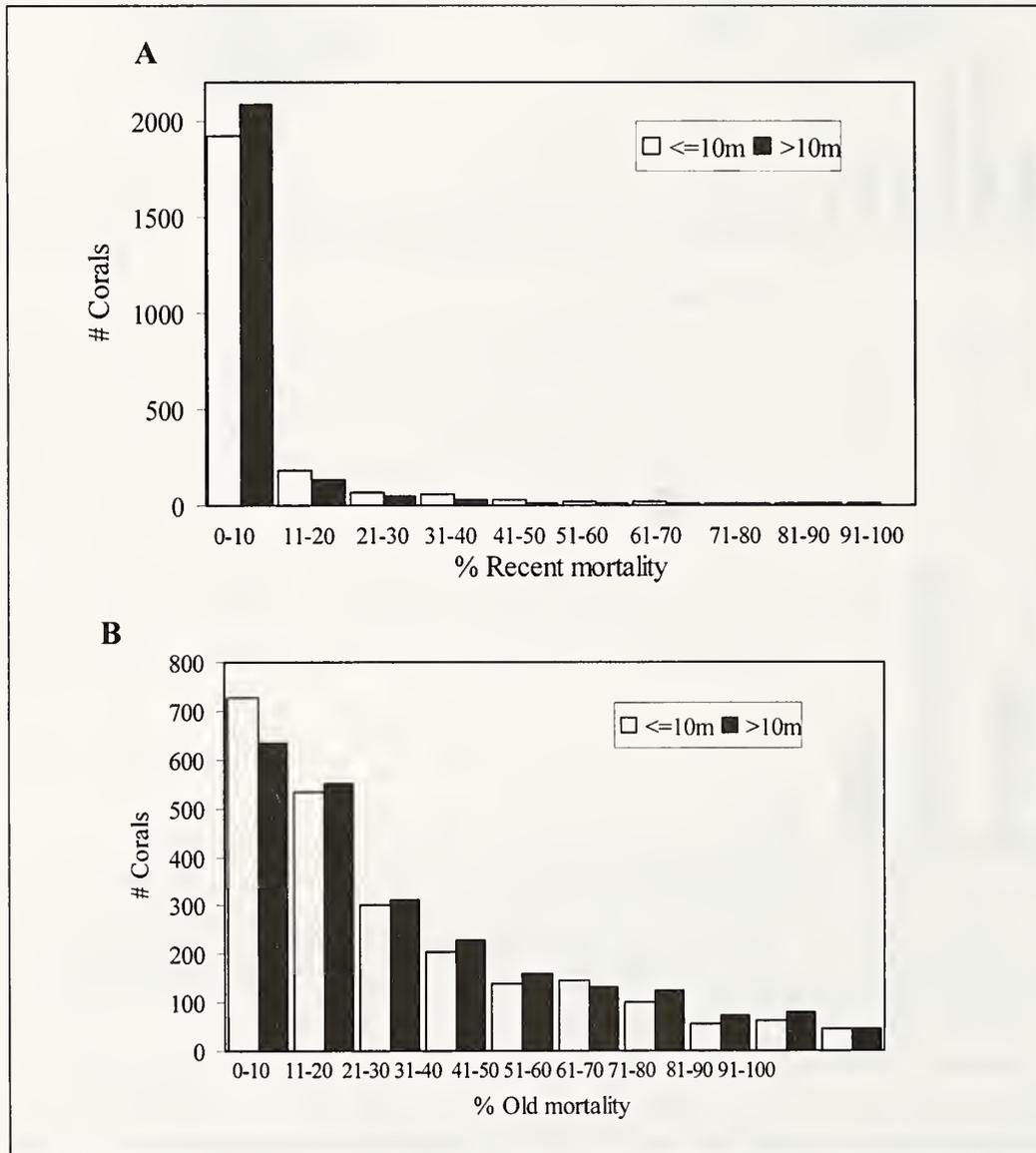
**Figure 4.** Size-frequency distribution of colonies ( $\geq 10$  cm diameter) of (A) *Montastraea annularis*, *M. faveolata* and *M. franksi*, (B) *Agaricia* spp. in shallower ( $\leq 10$  m) and deeper ( $> 10$  m) upper terrace reefs off the Cayman Islands.

### Stony Coral Diseases and Bleaching

Diseased stony corals were seen in over 90% (38/42) of the surveyed reefs (Table 2). Overall, the lowest incidence was at CB (mean = 2.3% diseased in 2000), while LC had the highest (mean = 5.5% diseased in 1999). Incidences of disease were

higher off the leeward sides of all three islands, affecting an average 6.5% of the leeward colonies versus 1.5 % of those off the windward coasts (Table 2). Heavily dived areas (CI 8,9,10, 22, 23, 27, 32, 33, CB 6,7) had high disease frequencies. For example, the mean percentages of diseased corals were more than double at two of the most popular dive sites (Cemetery Reef and Sunset House) on the western (leeward) side of GC than on its other coasts (Table 2). The highest percentage (mean >24%) of diseased corals was found at the Meadows site in Bloody Bay Marine Park off LC, and the average percentage of diseased colonies in Bloody Bay (10.5%) was nearly double that of the island as a whole.

More than 85% of these diseases were white syndromes (*sensu* Peters, 1997), i.e., white plague (WP) (presumably Type I, as subsequent visits indicated the disease was moving slowly across the coral tissues), which affected most species of massive



**Figure 5.** Frequency of (A) recent and (B) old partial colony mortality of stony corals ( $\geq 10$  cm diameter), by depth ( $\leq 10$ m and  $> 10$  m), in upper terrace reefs off the Cayman Islands.

corals except *Montastraea franksi*, and white-patch (white-pox) and white-band (WBD) diseases in *Acropora* (for descriptions of coral diseases see Peters, 1997). Black-band disease (BBD) and yellow-band (yellow-blotch) disease (YBD) only accounted, on average, for 11.4% and 3.4%, respectively, of the diseased corals off LC and GC, and neither were present in any of the transects or in random surveys off CB. No other diseases were noted in the study areas. Sixty-eight percent of the corals affected by disease were in the *Montastraea annularis* species complex, whereas only 3% were species of *Agaricia*. YBD was only found in *Montastraea annularis* and *M. franksi*; however, three colonies of *Porites astreoides* showed signs of a disease resembling YBD. BBD occurred preferentially in species of massive corals.

Regionally, the incidence of coral bleaching was low. The percent of stony corals that were bleached averaged higher off LC (2.4%) and GC (1.7%) in 1999 than off CB (0.3%) the following year. Bleached corals were twice as common in the deeper upper-terrace reefs off all three islands as in the shallower reefs. The non-algal-gardening yellowtail damselfish (*Microspathodon chrysurus*) was the most abundant damselfish in the immediate vicinity of the surveyed stony corals. In LC, 141 individuals/2,542 corals (5.5%) were counted; corresponding values for GC were 84 fish/2,162 corals (3.9%), and 19 fish/954 corals (1.99%) for CB.

### Stony Coral Mortality

Recent partial-colony mortality (hereafter recent mortality) was in the 0-10% range for most stony corals ( $\geq 10$ cm diameter). Recent mortality in GC in 1999 (5.0%) was more than double that in LC (2.1%); corresponding values were even lower (1.3%) in CB in 2000 (Fig. 5A; Table 2). Overall, areas with the highest abundance of disease were also areas with high recent mortality. Old partial mortality (hereafter old mortality) averaged 21.3% of these colony surfaces, and there were no significant depth-related trends or differences between the islands and no differences related to exposure (Fig. 5B; Table 2). Apart from the Bloody Bay Marine Park area, "standing dead" corals were uncommon off LC, but they occurred at all but two of the 15 GC reefs and all but one of the nine CB reefs (Table 2).

Stony corals in about half of the GC (8/15) and LC (8/18) reefs but only a few (2/9) of those in CB had a total (= recent + old) partial mortality exceeding 25% of their colony surfaces. Corals in leeward settings generally had higher overall mortality percentages than those in more exposed windward orientations (Table 2). However, the only GC locations that had total mortality percentages of less than 20% were off the eastern end of the island. The highest percentages of recent (10.5%) and total (36%) mortality were recorded at Isabel's Reef, a relatively new dive site off the northern GC coast.

High (>20%) to very high (>50%) percentages of recently dead colony surfaces in all three islands were most commonly found in *Montastraea annularis*, *Diploria strigosa* and *D. labyrinthiformis*. In the CB and LC reefs, *Agaricia* spp. had high percentages of recent mortality. Additional species with high-to-very high percentages of recent mortality off LC and GC included *M. faveolata* and *M. cavernosa*. In GC, *Acropora cervicornis*, *Colpophyllia natans* and *Siderastrea siderea* also commonly had high-to-very high percentages (10–60%) of recently dead

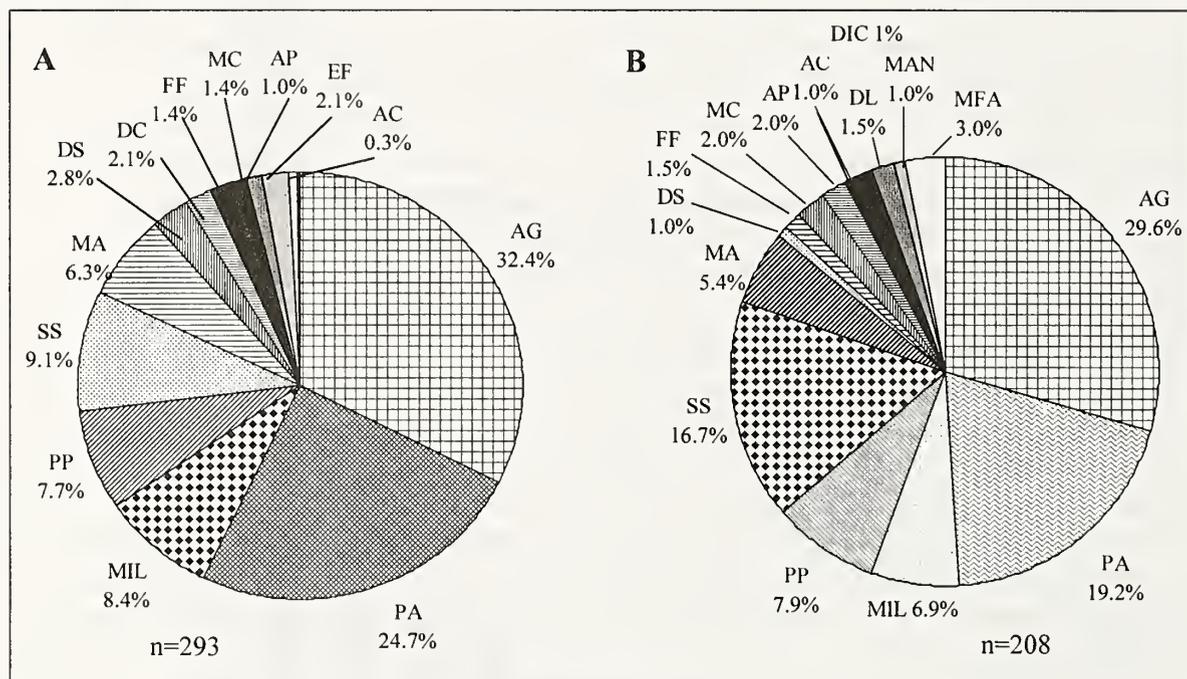
colony surfaces. On LC, recent mortality was also high (>20%) in *M. franksi* (which was less abundant on CB and GC) and in *Porites porites*.

### Stony Coral Recruitment

Recruitment averaged 4.2 corals/m<sup>2</sup> ( $= .26 \pm .65$  recruits/.0625 m<sup>2</sup>), was patchy in all reefs, and followed no trends related to exposure to higher or lower wave energy (Table 4). Recruitment was slightly higher in GC than in LC and CB. The most common recruits in the upper-terrace reefs were *Agaricia spp.* and *Porites astreoides* (Fig. 6A, B). Although there was little difference overall in the species composition of the recruits at these two depths, *Siderastrea siderea* was nearly twice as common in the deeper sites as in the shallow sites. Small colonies of *Montastraea annularis* were found at all depths; recruits of *M. faveolata* were also present in the deeper reefs.

### Algal Relative Abundance

The relative abundance of turf algae was low, averaging 13.1% off CB to 22.4% off LC. The relative abundance of crustose coralline algae was highest in GC (mean = 53.5%) and lowest in CB (mean = 28%), whereas macroalgae showed the opposite pattern (30.5% versus 59%, respectively). Crustose coralline algal



**Figure 6.** Species composition and mean relative abundance of all stony coral recruits ( $\leq 2$  cm diameter) in (A) shallower ( $\leq 10$  m) and (B) deeper ( $> 10$  m) upper terrace reefs off the Cayman Islands. AG = *Agaricia agaricites*, AC = *Acropora cervicornis*, AP = *A. palmata*, DIC = *Dichocoenia*, DC = *Diploria clivosa*, DL = *D. labyrinthiformis*, DS = *D. strigosa*, EF = *Eusmilia fastigiata*, FF = *Favia fragum*, MAN = *Manicina areolata*, MIL = *Millepora spp.*, MA = *Montastraea annularis*, MC = *M. cavernosa*, MFA = *M. faveolata*, PA = *Porites astreoides*, PP = *P. porites*, SS = *Siderastrea siderea*.

abundance was slightly higher in the deeper sites off GC and LC, while macroalgal abundance was higher on average in the deeper sites off all three islands. Turf algae were less abundant at the deeper sites in GC and LC. Because of taller growth, the average macroalgal index (relative abundance of macroalgae x macroalgal height) was nearly three times larger in CB than in GC (Table 4), and was also regionally higher in the deeper reefs. The highest macroalgal index value was found in a fairly remote reef (CB3) in a leeward/protected windward location on the northeast side of CB. Macroalgae were less abundant and macroalgal index values were lower in the more exposed windward sides of all three islands, except in LC where the macroalgal indices were slightly lower off the leeward side. The windward side of GC, however, clearly had the least macroalgae and the lowest macroalgal indices (which brought down the overall average for GC as well) (Tables 3, 4). *Dictyota* spp., *Styopodium zonale*, *Lobophora variegata*, and *Sargassum hystrix* were the most common macroalgae, especially on the southern sides of the islands. In some reefs, macroalgal encroachment and overgrowth, with resultant mortality of stony corals, was observed.

*Diadema antillarum* were only seen in transects at two reefs (one each shallower and deeper) on CB in 2000 and, even during roving dives, none were observed in the study areas off either GC or LC.

## DISCUSSION

Comparisons with earlier studies of the coral communities in the Cayman Islands indicate that many changes have occurred over the past 30 years. The shallow *Acropora palmata* zone, as described for the 1970's and 1980's (Rigby and Roberts, 1976; Logan, 1994), no longer exists. In 1988, Hurricane Gilbert heavily impacted the north side of GC and destroyed much of the *A. palmata*-dominated fringing reef (Blanchon et al., 1997; T. Austin, personal communication). The predominance of standing dead colonies of *A. palmata* in many of the upper-terrace study sites on GC (CI 19, 20, 21, 24, 29, 31) is indicative of prior, major mass mortality, possibly from white-band disease or, in part, from one or more of the earlier mass bleaching events. Tossed, dead fragments that were being consolidated by *Millepora* and crustose coralline algae, and a few small (<1 m in diameter) intact colonies of *A. palmata*, were all that remained in the shallow fringing reef crests on the north side of GC, where we had found higher densities during previous surveys in 1997 (Manfrino, unpublished data). The colonies in this area, growing close to sea level, appear to be periodically smashed by the passage of major storms and were probably affected by Hurricane Mitch in 1998.

Off all three islands, and at all depths and habitats in which they occurred, colonies of *A. palmata* and of *A. cervicornis* were affected by WBD and white-patch disease. Earlier studies indicate that WBD severely impacted *A. cervicornis* on the GC fore reef in the 1980s (Woodley et al., 1997). Nevertheless, these coral species appear to us to have the potential for strong regeneration in the Cayman Islands.

Overall, since reefs with the highest abundance of disease were generally the reefs with high recent mortality, we presume that the latter was the result of diseases, particularly WP, WBB, and white patch. It is difficult to account for the observation

that LC had a higher percentage of diseased corals, but lower recent mortality values, than GC. One possible explanation is that outbreaks of disease in GC had occurred earlier that year, or in the year prior to our surveys, and was only at the time affecting the LC reefs. The higher occurrence of diseases in LC during June 1999 was possibly due to the higher percentage of *Montastraea faveolata* in the outer terrace reefs as WP had preferentially impacted the *Montastraea annularis* species complex. Surveys that continued on LC in 2000 and 2001 indicated outbreaks of disease activity with a trend of decreasing incidence of WP in 2001, then a major increase since January 2002 (further increasing in the 2002 summer months) that is causing a major mortality on the reefs around Little Cayman (Manfrino, unpublished data). The low occurrence of WP and low recent mortality (and the low live stony coral cover) on CB in the 2000 survey also supports a theory of waves of disease having previously swept through the CB reefs.

Differences between the islands, between the different depths, and between the different island settings were documented. The islands differed in several ways. In CB, stony corals were smaller, live stony coral cover and recent mortality were lowest, as were incidences of disease and bleaching, but there was a higher percentage of standing dead coral than had been found the previous year off the other two islands. In GC there was higher recent and total mortality, higher abundance of crustose coralline algae, slightly higher coral recruitment, and lower macroalgal abundance and macroalgal index values (especially low off the windward GC coast). In LC, stony coral size and cover were greatest and though standing dead coral was lowest, the incidences of bleaching and disease were highest (especially in the Bloody Bay Marine Park where recent mortality was also high). Fish abundance was also higher in LC than in GC (see Pattengill-Semmens and Semmens, this volume). Disease and recent and total mortality were high in popular dive sites off GC and LC, which also happen to be in leeward settings. In general, mean coral diameter and height were greater in the shallower depths off all three islands. Diseased stony corals were more common but bleaching was less abundant in the shallower depths. The relative abundance of macroalgae was higher in the deeper reefs.

No relationship between recruitment and level of exposure was found around any of the islands. Higher recruitment and coral regeneration off GC may occur because the higher abundance of crustose coralline algae in the windward reefs provides a more suitable surface for coral recruits.

The difference in the overall condition of reefs within protected and unprotected areas was unexpected; stony corals in dive sites at Bloody Bay Marine Park, the most highly regulated park in the Cayman Islands, were undergoing a major mortality event in 1999, and this area had amongst the highest disease counts of all three islands.

Although this survey did not include sites in any of the Cayman's shelf-edge reefs, a limited number of wall dives made at Bloody Bay during June, 1999 indicated that bleaching was no longer significant in the deeper reefs. However, high levels of macroalgal overgrowth (up to 100%) on some of the stony corals suggested that a wave of recent mortality may have occurred in the aftermath of the 1998 bleaching event.

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Table 1. Site information for AGRRA stony coral and algal surveys in the Cayman Islands.

Site	Site Code <sup>1</sup>	Reef Type <sup>2</sup>	Relative Exposure <sup>3</sup>	Level of Protection <sup>4</sup>	Latitude (°'N)	Longitude (°'W)	Survey date	Depth (m)	Benthic transects (#)	stony corals (#/10m)	% live stony coral cover (mean ± sd)	
<b>Little Cayman</b>												
Meadows	CI8	Patch	Leeward	Park	19 41.510	80 04.130	Jun 10 99	6.0	16	14	37.0 ± 11.5	
Berges Meredith	CI17	S&G	Leeward	Open	19 42.470	80 00.495	Jun 15 99	9.0	10	10	22.5 ± 4.5	
Jigsaw Puzzle	CI2	High S&G	Leeward	Open	19 39.983	80 06.390	Jun 6 99	10.5	13	10	27.0 ± 9.5	
Joy's Joy	CI10	Shelf edge	Leeward	Park	19 40.690	80 05.575	Jun 10 99	12.0	14	11	19.0 ± 6.5	
Mixing Bowl-Three Fathom Wall	CI5	Shelf edge	Leeward	Park	19 41.096	80 04.700	Jun 8 99	12.5	15	13	29.0 ± 12.0	
Nancy's Cup of Tea	CI9	Shelf edge	Leeward	Park	19 41.639	80 04.137	Jun 10 99	12.5	13	10.5	21.5 ± 14.0	
Rock Bottom Wall	CI13	S&G	Leeward	Open	19 42.057	80 03.421	Jun 13 99	12.5	12	12	22.5 ± 10.5	
Paul's Anchor	CI12	Shelf edge	Leeward	Park	19 41.661	80 04.181	Jun 13 99	13.0	12	11.5	17.0 ± 12.0	
Lighthouse	CI16	S&G	Leeward	Open	19 39.702	80 06.728	Jun 15 99	14.0	10	10.5	25.5 ± 5.5	
Penguin's Leap	CI7	Hardpan	Leeward	Open	19 42.551	80 00.487	Jun 9 99	16.0	16	11	16.5 ± 8.0	
Wreck	CI18	Hardpan	Prot Windward	Outside Rep	19 42.996	79 58.921	Jun 15 99	11.0	10	10	15.0 ± 5.5	
Main Channel- East Side	CI11	Fringing	Windward	Open	19 39.412	80 04.368	Jun 11 99	3.0	12	10	16.0 ± 5.5	
Grundy's Gardens	CI1	S&G	Windward	Park	19 39.421	80 05.321	Jun 6 99	9.0	12	7.5	37.5 ± 11.5	
Disneyland	CI3	S&G	Windward	Outside Rep	19 49.831	80 01.374	Jun 7 99	10.0	14	11	25.5 ± 6.5	
Charles Bay	CI4	S&G	Windward	Open	19 41.628	79 58.459	Jun 7 99	11.0	17	8.5	17.5 ± 6.5	
No Name	CI15	S&G	Windward	Outside Rep	19 40.628	80 01.562	Jun 14 99	11.0	12	11	26.5 ± 8.5	
Black Tip Tunnels	CI6	S&G	Windward	Outside Rep	19 42.847	79 57.470	Jun 9 99	12.5	17	9.5	15.5 ± 7.5	
Lucas's Ledge	CI14	S&G	Windward	Outside Rep	19 40.155	80 02.595	Jun 14 99	13.5	10	13.5	24.5 ± 6.0	
<b>All Little Cayman (mean ± standard deviation)</b>								<b>11.0</b>	<b>13</b>	<b>10.8</b>	<b>23.1 ± 8.4</b>	
<b>Grand Cayman</b>												
Cemetery Reef	CI32	Patch/S&G	Leeward	Park	19 21.917	81 23.726	Jun 22 99	9.0	13	11.5	17.5 ± 5.0	
Sunset House	CI33	Patch/S&G	Leeward	Park	19 17.172	81 23.463	Jun 22 99	9.0	12	12	22.5 ± 12.0	
Hepp's Mini Wall	CI28	Patch/S&G	Leeward	Park	19 23.126	81 24.992	Jun 20 99	11.5	12	14	22.0 ± 5.5	
Delila's Delight	CI24	S&G	Prot Windward	Open	19 21.518	81 14.801	Jun 19 99	7.0	10	14.5	23.0 ± 13.0	
Babylon	CI23	S&G	Prot Windward	Outside Rep	19 21.200	81 09.842	Jun 18 99	9.5	11	13.5	18.0 ± 3.5	
Isabel's Reef	CI22	High S&G	Prot Windward	Open	19 21.460	81 08.145	Jun 18 99	10.5	11	12	24.5 ± 7.5	
Bear's Paw	CI27	S&G	Prot Windward	Outside Rep	19 23.854	81 21.617	Jun 20 99	10.5	12	11.5	15.5 ± 3.5	
Queen's Throne	CI25	Hardpan	Prot Windward	Outside Rep	19 22.818	81 17.493	Jun 19 99	12.0	12	11.5	14.5 ± 3.0	

Table 1. continued.

Site name	Site Code <sup>1</sup>	Reef Type <sup>2</sup>	Relative Exposure <sup>3</sup>	Level of Protection <sup>4</sup>	Latitude (° N)	Longitude (° W)	Survey date	Depth (m)	Benthic transects (#)	stony corals ≥25 cm (#/10m)	% live stony coral cover (mean ± sd)
Casey's Reef	CI26	S&G	Prot Windward	Outside Rep	19 21.202	81 11.746	Jun 19 99	14.0	12	10	12.5 ± 4.0
East End Reef	CI19	High S&G	Windward	Open	19 19.058	81 04.484	Jun 17 99	6.5	12	12	23.0 ± 9.5
Playing Field	CI31	High S&G	Windward	Open	19 17.565	81 06.318	Jun 21 99	7.5	12	12	17.5 ± 4.5
Kaho's Reef	CI20	High S&G	Windward	Open	19 20.002	81 04.596	Jun 17 99	9.0	11	12	18.0 ± 4.5
Kelly's Caverns	CI29	High S&G	Windward	Outside Rep	19 17.476	81 08.772	Jun 21 99	9.0	12	13.7	22.0 ± 6.5
Snapper Hole	CI21	High S&G	Windward	Open	19 20.634	81 04.676	Jun 18 99	20.0	12	13	24.0 ± 10.5
Breakers	CI30	S&G	Windward	Open	19 17.507	81 12.069	Jun 21 99	12.5	10	12.5	22.5 ± 4.5
<b>All Grand Cayman (mean ± standard deviation)</b>								<b>10.5</b>	<b>11.5</b>	<b>12.4</b>	<b>19.8 ± 6.5</b>
<i>Cayman Brac</i>											
Greenhouse Reef	CB7	High Patch/S&G	Leeward	Open	19 43.215	79 49.700	Jun 25 00	7.0	7	24	16.0 ± 4.5
Patch Reef	CB8	Patch	Leeward	Park	19 41.935	79 52.539	Jun 25 00	8.0	7	15	11.5 ± 4.0
End of Island	CB9	Low S&G	Leeward	Open	19 41.271	79 53.645	Jun 25 00	9.5	10	8	9.0 ± 2.5
Bert Brothers	CB3	S&G	Lee/Prot Wind	Open	19 45.491	79 44.457	Jun 22 00	16.5	8	12	14.0 ± 4.0
Pillars of Hercules	CB2	Fallen Blocks/S&G	Windward	Open	19 45.261	79 43.824	Jun 22 00	4.0	9	11	21.5 ± 5.5
Sargent Major Reef	CB6	S&G	Windward	Park	19 40.844	79 52.969	Jun 23 00	8.0	9	12	16.0 ± 7.5
Lighthouse Reef	CB5	S&G	Windward	Park	19 40.792	79 53.249	Jun 23 00	9.5	8	13	14.5 ± 5.5
Peter's Anchor	CB1	Fallen Blocks	Windward	Open	19 44.831	79 43.416	Jun 22 00	10.5	10	9	10.5 ± 4.5
Pillar Coral Reef	CB4	S&G	Windward	Park	19 40.933	79 52.187	Jun 23 00	10.5	9	12.5	16.0 ± 7.0
<b>All Cayman Brac (mean ± standard deviation)</b>								<b>9.5</b>	<b>8.5</b>	<b>12.8</b>	<b>14.3 ± 5.0</b>
<b>All Cayman Islands (mean ± standard error)</b>								<b>10.5</b>	<b>11.5</b>	<b>11.7</b>	<b>19.7 ± 6.9</b>

<sup>1</sup>CI = LC (for sites 1-18) or GC (for sites 19-33) in Pattengill-Semmens and Semmens (this volume).

<sup>2</sup>S & G = spur and groove

<sup>3</sup>Prot. = protected; Lee = leeward; Wind. = windward

<sup>4</sup>Rep = Replenishment Zone

Table 2. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 10$  cm diameter), by site in the Cayman Islands.

Site name	Site code	Relative exposure	Depth (m)	Stony corals (#)	Stony corals			Partial-colony mortality (%)			Stony corals (%)		
					Diameter (cm)	Height (cm)		Recent	Old	Total	Standing dead	Bleached	Diseased
<b>Little Cayman</b>													
Meadows	CI8	Lee	6.0	221	66.0 $\pm$ 45.5	43.0 $\pm$ 35.5	7.0 $\pm$ 16.5	23.0 $\pm$ 23.5	30.0 $\pm$ 26.5	0.5	0	24.5	
Berges Meredith	CI17	Lee	9.0	102	52.0 $\pm$ 30.0	32.0 $\pm$ 25.5	2.0 $\pm$ 4.0	22.5 $\pm$ 23.0	24.0 $\pm$ 24.5	0	1.0	9.0	
Jigsaw Puzzle	CI2	Lee	10.5	128	43.0 $\pm$ 23.5	31.0 $\pm$ 21.5	1.0 $\pm$ 7.0	20.0 $\pm$ 20.5	20.5 $\pm$ 21.5	1.0	0	1.5	
Joy's Joy	CI10	Lee	12.0	157	35.0 $\pm$ 23.0	23.5 $\pm$ 17.5	5.0 $\pm$ 14.0	19.0 $\pm$ 22.5	24.0 $\pm$ 25.0	0	2.5	12.5	
Mixing Bowl-Three Fathom Wall	CI5	Lee	12.5	196	40.5 $\pm$ 20.5	29.5 $\pm$ 21.0	0.5 $\pm$ 2.5	23.0 $\pm$ 23.5	23.5 $\pm$ 23.5	0	1.0	0.5	
Nancy's Cup of Tea	CI9	Lee	12.5	138	43.0 $\pm$ 31.5	28.0 $\pm$ 23.0	2.5 $\pm$ 7.5	30.0 $\pm$ 27.0	32.0 $\pm$ 28.0	1.5	3.5	12.5	
Rock Bottom Wall	CI13	Lee	12.5	142	41.5 $\pm$ 28.5	23.5 $\pm$ 18.0	1.0 $\pm$ 4.5	26.5 $\pm$ 26.0	27.5 $\pm$ 26.5	0	7.0	3.0	
Paul's Anchor	CI12	Lee	13.0	139	34.5 $\pm$ 18.0	23.5 $\pm$ 16.5	2.0 $\pm$ 5.0	24.5 $\pm$ 25.0	26.5 $\pm$ 27.0	0	1.5	5.0	
Lighthouse	CI16	Lee	14.0	106	46.0 $\pm$ 26.0	25.5 $\pm$ 22.0	1.5 $\pm$ 8.5	15.5 $\pm$ 18.5	17.0 $\pm$ 20.5	0	5.5	3.0	
Penguin's Leap	CI7	Lee	16.0	174	35.5 $\pm$ 18.5	26.5 $\pm$ 16.0	1.5 $\pm$ 4.5	33.5 $\pm$ 28.0	35.0 $\pm$ 29.0	0	3.5	3.0	
Wreck	CI18	Prot Wind	11.0	99	42.5 $\pm$ 19.5	26.5 $\pm$ 16.0	6.5 $\pm$ 20.5	20.5 $\pm$ 21.5	27.0 $\pm$ 25.5	1.0	3.0	8.0	
Main Channel- East Side	CI11	Wind	3.0	122	38.0 $\pm$ 35.0	17.0 $\pm$ 20.0	0.5 $\pm$ 3.0	14.0 $\pm$ 23.5	14.5 $\pm$ 24.0	0	0	4.0	
Grundy's Gardens	CI1	Wind	9.5	90	80.0 $\pm$ 71.5	59.0 $\pm$ 66.5	1.0 $\pm$ 4.5	23.5 $\pm$ 22.5	24.0 $\pm$ 22.5	0	1.0	0	
Disneyland	CI3	Wind	10.0	152	48.0 $\pm$ 39.5	32.0 $\pm$ 26.0	1.0 $\pm$ 6.5	20.5 $\pm$ 26.5	27.5 $\pm$ 26.5	0	2.5	0.5	
Charles Bay	CI4	Wind	11.0	146	40.0 $\pm$ 35.0	27.5 $\pm$ 34.0	0.5 $\pm$ 3.0	17.0 $\pm$ 24.0	17.5 $\pm$ 23.5	0	2.0	0.5	
No Name	CI15	Wind	11.0	130	54.0 $\pm$ 49.0	30.5 $\pm$ 31.5	2.0 $\pm$ 6.0	17.5 $\pm$ 21.5	19.5 $\pm$ 24.5	0	2.5	3.0	
Black Tip Tunnels	CI6	Wind	12.5	162	36.5 $\pm$ 29.0	25.0 $\pm$ 24.5	1.0 $\pm$ 6.5	25.5 $\pm$ 28.0	26.5 $\pm$ 28.5	0.5	5.0	2.0	
Lucas's Ledge	CI14	Wind	13.5	135	42.5 $\pm$ 30.5	27.0 $\pm$ 22.0	1.5 $\pm$ 6.5	15.0 $\pm$ 20.50	16.5 $\pm$ 21.5	0	2.0	6.0	
<b>All Little Cayman (mean <math>\pm</math> standard deviation)</b>			<b>11.0</b>	<b>141.1</b>	<b>45.4 <math>\pm</math> 31.4</b>	<b>29.5 <math>\pm</math> 25.4</b>	<b>2.1 <math>\pm</math> 7.2</b>	<b>21.7 <math>\pm</math> 23.6</b>	<b>24.1 <math>\pm</math> 25.0</b>	<b>0.2</b>	<b>2.4</b>	<b>5.5</b>	
<b>Grand Cayman</b>													
Cemetery Reef	CI32	Lee	9.0	151	45.5 $\pm$ 29.0	28.0 $\pm$ 29.5	9.0 $\pm$ 18.5	27.5 $\pm$ 26.5	36.5 $\pm$ 32.5	2.0	0	10.5	
Sunset House	CI33	Lee	9.0	144	49.0 $\pm$ 31.0	30.5 $\pm$ 28.0	7.5 $\pm$ 16.5	22.5 $\pm$ 23.5	30.0 $\pm$ 27.5	0	0	11.0	
Hepp's Mini Wall	CI28	Lee	11.5	170	37.5 $\pm$ 26.5	23. $\pm$ 21.00	3.5 $\pm$ 11.5	19.0 $\pm$ 22.5	22.5 $\pm$ 25.5	0	2.0	2.0	
Delila's Delight	CI24	Prot Wind	7.0	143	44.5 $\pm$ 33.5	25.5 $\pm$ 24.5	5.5 $\pm$ 16.5	24.0 $\pm$ 27.5	29.5 $\pm$ 30.5	3.0	3.5	3.5	
Babylon	CI23	Prot Wind	9.5	151	43.5 $\pm$ 24.5	22.0 $\pm$ 15.0	6.5 $\pm$ 16.0	28.0 $\pm$ 28.5	35.0 $\pm$ 31.0	0.5	0.5	5.5	
Isabel's Reef	CI22	Prot Wind	10.5	132	47.5 $\pm$ 25.5	24.5 $\pm$ 15.5	10.5 $\pm$ 22.5	26.0 $\pm$ 26.5	36.0 $\pm$ 31.5	2.5	4.0	7.0	
Bear's Paw	CI27	Prot Wind	10.5	136	43.0 $\pm$ 28.5	22.0 $\pm$ 23.0	6.5 $\pm$ 16.5	26.0 $\pm$ 27.0	32.5 $\pm$ 33.0	6.0	3.0	7.5	
Queen's Throne	CI25	Prot Wind.	12.0	137	36.5 $\pm$ 18.0	20.5 $\pm$ 14.0	5.0 $\pm$ 14.5	23.0 $\pm$ 25.5	28.0 $\pm$ 29.5	1.5	2.0	0.5	

Table 2. continued.

Site name	Site code	Relative exposure	Depth (m)	Stony corals		Partial-colony mortality (%)			Stony corals (%)			
				(#)	Diameter (cm)	Height (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased
Casey's Reef	CI26	Prot Wind	14.0	122	34.5 ± 13.5	17.5 ± 11.5	3.5 ± 13.5	20.5 ± 23.5	24.0 ± 26.5	1.5	5.0	1.0
East End Reef	CI19	Wind	7.0	145	36.0 ± 30.5	18.5 ± 24.5	1.5 ± 9.0	16.5 ± 21.5	18.0 ± 23.0	2.0	0	4.0
Playing Field	CI31	Wind	7.5	144	43.0 ± 37.0	23.5 ± 23.0	2.5 ± 7.5	19.0 ± 24.5	21.5 ± 27.0	0.5	0	1.5
Kaho's Reef	CI20	Wind	9.0	131	33.0 ± 21.0	17.0 ± 13.5	2.0 ± 9.5	15.5 ± 21.5	17.5 ± 23.0	1.5	1.5	3.0
Kelly's Caverns	CI29	Wind	9.0	164	45.5 ± 38.0	27.0 ± 25.5	3.5 ± 13.5	20.0 ± 24.5	23.5 ± 26.5	2.5	2.0	0.0
Snapper Hole	CI21	Wind	10.0	155	47.5 ± 33.5	25.0 ± 20.5	2.0 ± 6.5	18.5 ± 28.5	20.5 ± 29.5	4.5	1.5	2.0
Breakers	CI30	Wind	12.5	127	44.0 ± 30.0	21.0 ± 19.5	6.0 ± 21.0	24.0 ± 25.0	30.0 ± 29.5	1.5	1.0	1.0
<b>All Grand Cayman (mean ± standard deviation)</b>			<b>10.0</b>	<b>143.5</b>	<b>41.8 ± 28.0</b>	<b>23.0 ± 20.6</b>	<b>5.0 ± 14.1</b>	<b>22.0 ± 25.1</b>	<b>27.0 ± 28.5</b>	<b>2.0</b>	<b>1.7</b>	<b>4.0</b>
<i>Cayman Brac</i>												
Greenhouse Reef	CB7	Lee	7.0	168	31.5 ± 21.0	25.0 ± 16.0	2.5 ± 8.0	29.5 ± 32.0	34.0 ± 32.0	6.0	0	4.0
Patch Reef	CB8	Lee	8.0	106	28.5 ± 32.0	24.0 ± 28.0	1.0 ± 2.5	31.5 ± 29.0	32.5 ± 29.0	3.0	0	6.5
End of Island	CB9	Lee	9.5	79	25.0 ± 14.0	17.0 ± 13.5	0.2 ± 1.5	19.5 ± 26.0	19.5 ± 26.0	2.5	1.5	2.5
Bert Brothers	CB3	Lee/ Prot Wind	16.5	94	27.0 ± 18.5	21.0 ± 13.0	2.5 ± 6.5	18.0 ± 23.5	20.5 ± 25.0	1.0	0	4.5
Pillars of Hercules	CB2	Wind	4.0	97	38.5 ± 32.0	20.5 ± 14.5	0.1 ± 1.0	10.0 ± 14.5	10.0 ± 14.5	0	0	0
Sergeant Major Reef	CB6	Wind	8.0	107	34.5 ± 35.0	24.0 ± 24.5	0.2 ± 1.0	18.5 ± 29.5	18.5 ± 30.0	6.5	0	1.0
Lighthouse Reef	CB5	Wind	9.5	103	38.5 ± 34.5	29.5 ± 26.5	1.5 ± 5.0	14.5 ± 25.0	16.0 ± 26.0	4.0	0	1.0
Peter's Anchor	CB1	Wind	10.5	89	29.0 ± 23.0	14.5 ± 11.5	2.5 ± 11.5	10.5 ± 17.0	13.0 ± 20.5	1.0	1.0	0
Pillar Coral Reef	CB4	Wind	10.5	112	37.0 ± 32.5	29.5 ± 25.0	2.0 ± 4.5	22.5 ± 30.5	24.0 ± 31.5	4.5	0	1.0
<b>All Cayman Brac (mean ± standard deviation)</b>			<b>9.0</b>	<b>106.1</b>	<b>32.3 ± 27.0</b>	<b>22.7 ± 19.2</b>	<b>1.3 ± 4.6</b>	<b>19.3 ± 25.3</b>	<b>20.9 ± 26.1</b>	<b>3.2</b>	<b>0.3</b>	<b>2.3</b>
<b>All Cayman (mean ± standard error)</b>			<b>10.5</b>	<b>134.4</b>	<b>41.4 ± 29.5</b>	<b>25.7 ± 22.5</b>	<b>3.0 ± 9.0</b>	<b>21.2 ± 24.5</b>	<b>24.3 ± 26.5</b>	<b>1.5</b>	<b>1.7</b>	<b>4.2</b>

Table 3. Comparison of benthos data (mean  $\pm$  standard deviation, where appropriate) in leeward, windward and protected windward settings and at all shallower ( $\leq 10\text{m}$ ) and all deeper ( $> 10\text{m}$ ) sites, in Little Cayman, Grand Cayman and Cayman Brac.

Parameter Island <sup>1</sup>	Setting <sup>2</sup>					Depth	
	Leeward	Windward	Leeward + Protected Windward	Protected Windward	Protected windward + Windward	Shallower	Deeper
<b>Diameter, stony corals (cm)</b>							
LC	43.5 $\pm$ 26.5	48.5 $\pm$ 40.0	43.5	42.5	47.5	61.5 $\pm$ 46.5	41.0 $\pm$ 26.5
GC	44.0 $\pm$ 29.0	41.5 $\pm$ 31.5	42.5	41.5	41.5	42.5 $\pm$ 30.5	34.5 $\pm$ 13.5
CB	28.0 $\pm$ 21.5	35.5 $\pm$ 31.5				33.0 $\pm$ 28.0	31.0 $\pm$ 24.5
<b>Height, stony corals (cm)</b>							
LC	28.5 $\pm$ 21.5	31.5 $\pm$ 32.0	28.5	26.5	30.5	41.5 $\pm$ 38.5	26.5 $\pm$ 22.0
GC	27.0 $\pm$ 26.0	22.0 $\pm$ 21.0	24.0	22.0	22.0	24.0 $\pm$ 23.0	17.5 $\pm$ 11.5
CB	21.5 $\pm$ 17.5	23.5 $\pm$ 20.5				23.5 $\pm$ 20.5	21.5 $\pm$ 16.5
<b>Diseased stony corals (%)</b>							
LC	7.5	2.5	7.5	8.0	3.0	8.5	4.5
GC	8.0	2.0	5.5	4.0	3.0	5.0	1.0
CB	4.5	0.5				2.5	1.5
<b>Bleached stony corals (%)</b>							
LC	2.5	2.0	2.5	3.0	2.5	1.0	3.0
GC	0.5	1.0	2.0	3.0	2.0	1.0	5.0
CB	0.5	0.2				0.2	0.5
<b>Recruitment, stony corals (#/.0625m<sup>2</sup>)</b>							
LC	0.2 $\pm$ 0.6	0.2 $\pm$ 0.6	0.3	0.5	0.3	0.3 $\pm$ 0.8	0.2 $\pm$ 0.5
GC	0.3 $\pm$ 0.6	0.3 $\pm$ 0.5	0.3	0.3	0.3	0.3 $\pm$ 0.6	0.3 $\pm$ 0.6
CB	0.2 $\pm$ 0.3	0.2 $\pm$ 0.3				0.2 $\pm$ 0.3	0.2 $\pm$ 0.4
<b>Recent partial mortality, stony corals (%)</b>							
LC	2.5 $\pm$ 7.5	1.0 $\pm$ 5.0	3.0	6.5	1.5	2.5 $\pm$ 8.0	2.0 $\pm$ 7.5
GC	6.5 $\pm$ 15.5	3.0 $\pm$ 11.	6.5	6.0	4.5	5.0 $\pm$ 13.5	3.5 $\pm$ 13.5
CB	1.5 $\pm$ 4.5	1.0 $\pm$ 4.5				1.0 $\pm$ 3.0	2.0 $\pm$ 7.5
<b>Old partial mortality, stony corals (%)</b>							
LC	23.5 $\pm$ 23.5	19.0 $\pm$ 24.0	21.5	23.5	19.0	22.5 $\pm$ 24.0	22.0 $\pm$ 23.5
GC	23.0 $\pm$ 24.0	19.0 $\pm$ 24.0	24.0	24.5	21.5	21.5 $\pm$ 24.5	20.5 $\pm$ 23.5
CB	24.5 $\pm$ 27.5	15.0 $\pm$ 23.5				20.5 $\pm$ 26.0	17.0 $\pm$ 23.5
<b>Standing dead, stony corals (%)</b>							
LC	0.5	0.1	0.3	1.0	0.2	0.1	0.3
GC	0.5	2.0	2.0	2.5	2.5	1.5	1.5
CB	3.0	3.0				3.5	2.0
<b>Relative abundance, crustose coralline algae (%)</b>							
LC	39.0 $\pm$ 22.0	35.5 $\pm$ 22.5	39.0	40.0	36.0	34.5 $\pm$ 25.0	39.0 $\pm$ 21.0
GC	36.5 $\pm$ 20.5	63.5 $\pm$ 20.5	46.5	51.5	57.5	51.0 $\pm$ 21.0	55.5 $\pm$ 19.5
CB	30.5 $\pm$ 22.0	26.5 $\pm$ 25.0				31.5 $\pm$ 24.0	21.5 $\pm$ 22.0
<b>Relative abundance, macroalgae (%)</b>							
LC	42.0 $\pm$ 21.0	36.5 $\pm$ 21.0	42.0	44.0	37.5	29.0 $\pm$ 20.0	44.0 $\pm$ 21.5
GC	39.5 $\pm$ 26.0	15.5 $\pm$ 19.0	39.0	39.0	27.0	27.5 $\pm$ 22.0	32.0 $\pm$ 20.5
CB	62.0 $\pm$ 22.0	56.0 $\pm$ 26.0				55.5 $\pm$ 24.5	65.5 $\pm$ 24.0
<b>Macroalgal Index</b>							
LC	83.9	93.1	82.2	65.1	89.6	73.6	90.7
GC	55.2	19.8	54.8	54.7	37.2	53.1	23.1
CB	175.0	92.7				100.3	197.3
<b>Relative abundance, turf algae (%)</b>							
LC	19.5 $\pm$ 19.5	27.5 $\pm$ 22.5	19.0	16.0	26.5	36.5	17.0 $\pm$ 18.5
GC	24.0 $\pm$ 25.0	21.0 $\pm$ 19.0	14.5	9.5	15.5	21.5 $\pm$ 21.5	12.0 $\pm$ 16.0
CB	7.5 $\pm$ 11.5	17.5 $\pm$ 19.0				13.0 $\pm$ 16.0	13.0 $\pm$ 15.5

<sup>1</sup>LC = Little Cayman; GC = Grand Cayman; CB = Cayman Brac

<sup>2</sup>None of the sites at CB are considered protected leeward with the possible exception of the Bert Brothers location which is a marginal protected leeward location. Only one site on LC would be considered protected windward (Wreck).

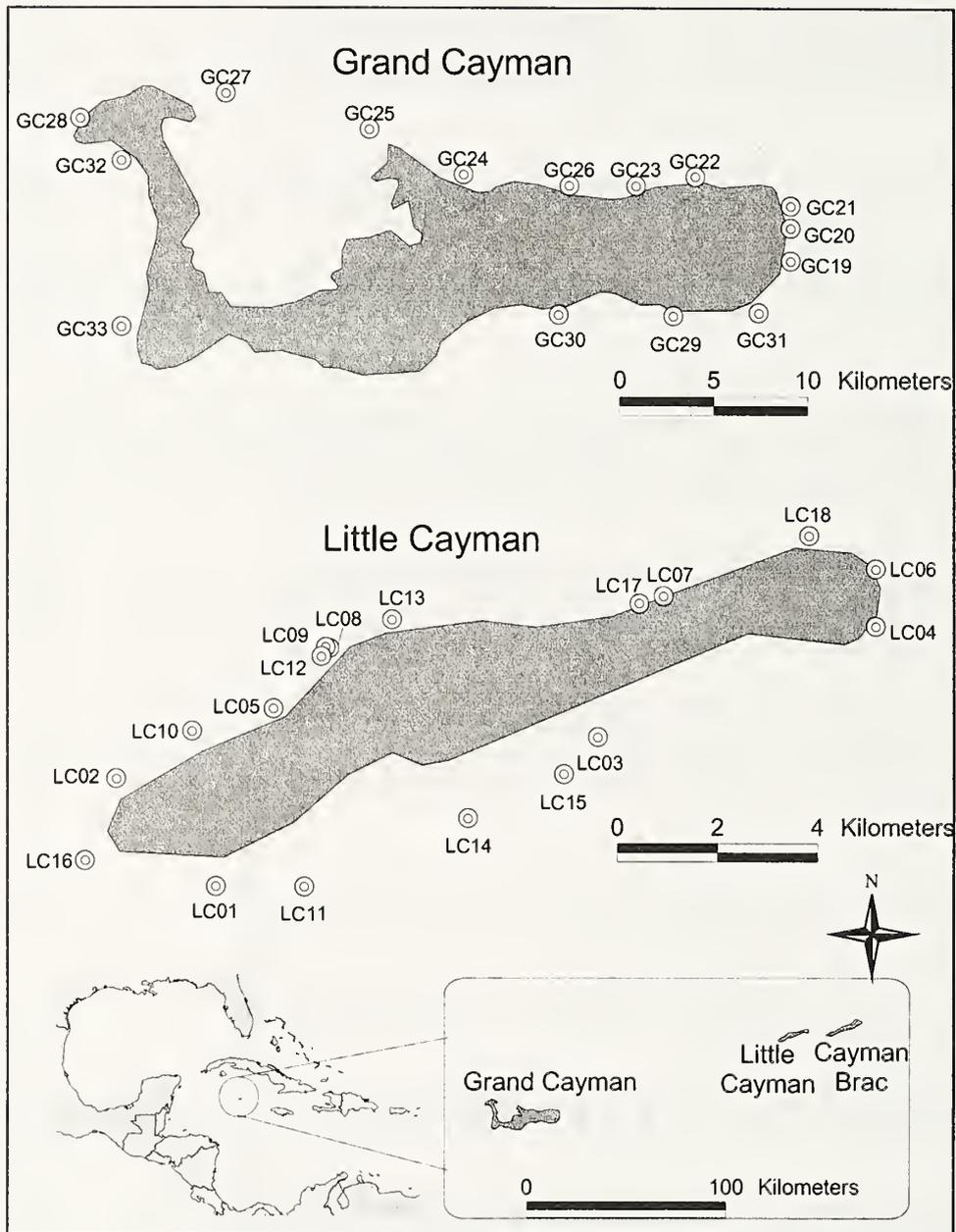
Table 4. Algal characteristics, density of stony coral recruits and *Diadema anillarum* (mean  $\pm$  standard deviation), by site in the Cayman Islands.

Site Name	Site Code	Relative Exposure <sup>1</sup>	Depth (m)	Quadrats (#)	Relative Abundance (%)			Macroalgal		Recruits (#/0625m <sup>2</sup> )	<i>Diadema</i> (#/100m <sup>2</sup> )
					Macroalgae	Turf algae	Crustose coralline algae	Height (cm)	Index <sup>2</sup>		
<b><i>Little Cayman</i></b>											
Meadows	CI8	Lee	6	75	23.0 $\pm$ 23.0	41.5 $\pm$ 29.5	35.5 $\pm$ 29.0	2 $\pm$ 1	36	0.4 $\pm$ 1.2	0
Berges Meredith	CI17	Lee	9	50	47.5 $\pm$ 24.0	16.0 $\pm$ 19.5	37.0 $\pm$ 23.0	2 $\pm$ 1	95	0.3 $\pm$ 0.7	0
Jigsaw Puzzle	CI2	Lee	10.5	67	28.0 $\pm$ 20.5	29.0 $\pm$ 21.0	43.5 $\pm$ 22.0	2 $\pm$ 1	62	0.4 $\pm$ 1.1	0
Joy's Joy	CI10	Lee	12	70	41.0 $\pm$ 20.5	15.0 $\pm$ 16.0	44.0 $\pm$ 24.5	2 $\pm$ 1	76	0.2 $\pm$ 0.4	0
Mixing Bowl-Three Fathom Wall	CI5	Lee	12.5	67	36.5 $\pm$ 17.5	21.0 $\pm$ 21.0	42.5 $\pm$ 19.0	3 $\pm$ 2	98	0.3 $\pm$ 0.5	0
Nancy's Cup of Tea	CI9	Lee	12.5	60	50.5 $\pm$ 20.5	13.0 $\pm$ 14.5	36.5 $\pm$ 19.5	2 $\pm$ 1	83	0.1 $\pm$ 0.3	0
Rock Bottom Wall	CI13	Lee	12.5	60	45.5 $\pm$ 20.0	13.0 $\pm$ 16.5	41.5 $\pm$ 22.0	2 $\pm$ 1	94	0.3 $\pm$ 0.5	0
Paul's Anchor	CI12	Lee	13	60	41.5 $\pm$ 19.0	16.5 $\pm$ 19.5	42.0 $\pm$ 20.0	2 $\pm$ 1	80	0.2 $\pm$ 0.4	0
Lighthouse	CI16	Lee	14	50	58.5 $\pm$ 22.0	13.5 $\pm$ 17.5	28.5 $\pm$ 17.5	2 $\pm$ 1	102	0.1 $\pm$ 0.5	0
Penguin's Leap	CI7	Lee	16	79	47.0 $\pm$ 20.5	14.5 $\pm$ 17.5	38.5 $\pm$ 23.5	2 $\pm$ 2	116	0.1 $\pm$ 0.4	0
Wreck	CI18	Prot Wind	11	50	44.0 $\pm$ 21.1	16.0 $\pm$ 18.5	40.0 $\pm$ 18.5	1 $\pm$ 1	65	0.5 $\pm$ 0.7	0
Main Channel- East Side	CI11	Wind	3	59	8.0 $\pm$ 12.0	59.0 $\pm$ 30.5	32.5 $\pm$ 31.0	1 $\pm$ 1	8	0.4 $\pm$ 0.7	0
Grundy's Gardens	CI1	Wind	9	60	38.0 $\pm$ 24.0	36.5 $\pm$ 24.5	25.5 $\pm$ 20.0	3 $\pm$ 2	112	0.1 $\pm$ 0.4	0
Disneyland	CI3	Wind	10	75	29.0 $\pm$ 17.0	30.0 $\pm$ 21.0	41.0 $\pm$ 22.0	3 $\pm$ 1	76	0.4 $\pm$ 0.8	0
Charles Bay	CI4	Wind	11	89	39.0 $\pm$ 24.5	24.0 $\pm$ 21.5	37.0 $\pm$ 20.0	3 $\pm$ 3	115	0.2 $\pm$ 0.5	0
No Name	CI15	Wind	11	60	49.0 $\pm$ 24.0	14.5 $\pm$ 21.0	37.0 $\pm$ 21.5	2 $\pm$ 1	120	0.1 $\pm$ 0.3	0
Black Tip Tunnels	CI6	Wind	12.5	85	46.0 $\pm$ 23.5	18.5 $\pm$ 20.0	35.5 $\pm$ 22.0	2 $\pm$ 1	115	0.2 $\pm$ 0.5	0
Lucas's Ledge	CI14	Wind	13.5	60	48.0 $\pm$ 22.0	11.0 $\pm$ 18.0	40.5 $\pm$ 22.5	2 $\pm$ 1	108	0.3 $\pm$ 0.9	0
<b>All Little Cayman (mean <math>\pm</math> standard error)</b>			<b>11</b>	<b>65.3</b>	<b>40.0 <math>\pm</math> 21.0</b>	<b>22.5 <math>\pm</math> 20.5</b>	<b>37.5 <math>\pm</math> 22.0</b>	<b>2 <math>\pm</math> 1</b>	<b>87</b>	<b>0.2 <math>\pm</math> 0.6</b>	<b>0.0</b>
<b><i>Grand Cayman</i></b>											
Cemetery Reef	CI32	Lee	9	65	47.0 $\pm$ 30.5	34.0 $\pm$ 30.0	19.0 $\pm$ 20.0	2 $\pm$ 1	91	0.2 $\pm$ 0.4	0
Sunset House	CI33	Lee	9.0	59	39.0 $\pm$ 24.5	27.5 $\pm$ 26.5	34.0 $\pm$ 19.5	1 $\pm$ 1	43	0.1 $\pm$ 0.3	0
Hepp's Mini Wall	CI28	Lee	11.5	65	32.0 $\pm$ 23.0	11.0 $\pm$ 18.0	56.5 $\pm$ 21.0	1 $\pm$ 1	32	0.5 $\pm$ 0.9	0
Delila's Delight	CI24	Prot Wind	7	50	25.5 $\pm$ 21.5	24.5 $\pm$ 25.0	50.0 $\pm$ 22.0	2 $\pm$ 1	52	0.6 $\pm$ 1.5	0
Babylon	CI23	Prot Wind	9.5	54	46.0 $\pm$ 28.5	11.0 $\pm$ 18.5	43.0 $\pm$ 21.0	2 $\pm$ 1	91	0.2 $\pm$ 0.5	0
Isabel's Reef	CI22	Prot Wind	10.5	44	42.0 $\pm$ 20.0	7.0 $\pm$ 17.0	51.0 $\pm$ 19.5	2 $\pm$ 1	69	0.3 $\pm$ 0.6	0
Bear's Paw	CI27	Prot Wind	10.5	60	39.0 $\pm$ 24.0	8.5 $\pm$ 16.0	52.0 $\pm$ 21.0	1 $\pm$ 1	46	0.2 $\pm$ 0.5	0
Queen's Throne	CI25	Prot Wind	12	59	41.0 $\pm$ 17.5	4.0 $\pm$ 12.5	55.0 $\pm$ 17.0	1 $\pm$ 1	39	0.3 $\pm$ 0.4	0

Table 4. continued.

Site Name	Site Code	Relative Exposure <sup>1</sup>	Depth (m)	Quadrats (#)	Relative Abundance (%)			Macroalgal		Recruits (#/.0625m <sup>2</sup> )	Diadema (#/100m <sup>2</sup> )
					Macroalgae	Turf algae	Crustose coralline algae	Height (cm)	Index <sup>2</sup>		
Casey's Reef	CI26	Prot Wind	14	60	39.5 ± 20.5	2.4 ± 5.5	58.0 ± 21.5	1 ± 1	39	0.3 ± 0.5	0
East End Reef	CI19	Wind	6.5	60	16.0 ± 20.5	14.0 ± 17.5	70.0 ± 22.0	1 ± 1	20	0.3 ± 0.6	0
Playing Field	CI31	Wind	7.5	60	8.5 ± 14.0	26.5 ± 20.5	65.0 ± 20.0	1 ± 1	5	0.2 ± 0.4	0
Kaho's Reef	CI20	Wind	9	56	28.5 ± 25.5	10.5 ± 15.5	61.0 ± 21.5	2 ± 1	47	0.4 ± 0.7	0
Kelly's Caverns	CI29	Wind	9	60	9.0 ± 13.0	23.5 ± 18.0	67.5 ± 19.5	1 ± 1	7	0.4 ± 0.5	0
Snapper Hole	CI21	Wind	20	60	18.5 ± 22.5	24.0 ± 20.5	57.5 ± 17.5	1 ± 1	26	0.2 ± 0.5	0
Breakers	CI30	Wind	12.5	50	13.5 ± 17.5	27.0 ± 22.5	59.5 ± 20.5	1 ± 1	13	0.2 ± 0.5	0
<b>All Grand Cayman (mean ± standard error)</b>			<b>10.5</b>	<b>57.5</b>	<b>30.5 ± 21.5</b>	<b>17.0 ± 19.0</b>	<b>53.5 ± 20.5</b>	<b>1 ± 1</b>	<b>42</b>	<b>0.3 ± 0.6</b>	<b>0.0</b>
<b>Cayman Brac</b>											
Greenhouse Reef	CB7	Lee	7	35	64.0 ± 24.0	15.5 ± 20.0	20.5 ± 19.0	2 ± 1	147	0	0
Patch Reef	CB8	Lee	8	35	75.0 ± 14.0	4.0 ± 6.0	21 ± 14.5	2 ± 1	165	0.2 ± 0.4	0
End of Island	CB9	Lee	9.5	45	34.5 ± 27.0	5.3 ± 8.5	60.5 ± 31.5	1 ± 1	21	0.2 ± 0.0	0
Bert Brothers	CB3	Lee	16.5	40	75.0 ± 22.0	5.0 ± 10.5	20.0 ± 22.5	5 ± 5	368	0.3 ± 0.7	0
Pillars of Hercules	CB2	Wind	4	45	38.0 ± 28.0	29.5 ± 28.0	33.0 ± 28.5	2 ± 1	61	0.1 ± 0.0	9
Sargent Major Reef	CB6	Wind	8	45	54.5 ± 30.0	8.0 ± 14.0	37.5 ± 31.5	1 ± 1	71	0.4 ± 0.5	0
Lighthouse Reef	CB5	Wind	9.5	40	67 ± 24.0	16.0 ± 19.0	16.5 ± 20.0	1 ± 3	94	0.2 ± 0.6	0
Peter's Anchor	CB1	Wind	10.5	45	51.5 ± 25.5	24.0 ± 20.0	24.5 ± 23.5	3 ± 1	134	0.2 ± 0.5	1
Pillar Coral Reef	CB4	Wind	10.5	40	69.5 ± 24.0	10.5 ± 15.5	20.0 ± 20.5	2 ± 1	104	0.2 ± 0.0	0
<b>All Cayman Brac (mean ± standard error)</b>			<b>9.5</b>	<b>41.1</b>	<b>59.0 ± 24.5</b>	<b>13.0 ± 15.5</b>	<b>28.0 ± 23.5</b>	<b>2 ± 2</b>	<b>129</b>	<b>0.2 ± 0.3</b>	<b>1.1</b>
<b>All Cayman (mean ± standard error)</b>			<b>10.5</b>	<b>56.9</b>	<b>41.0 ± 22.0</b>	<b>18.0 ± 17.5</b>	<b>41.5 ± 21.5</b>	<b>2 ± 1</b>	<b>81</b>	<b>0.2 ± 0.5</b>	<b>0.2</b>

Lee = leeward; Wind = windward; Prot Wind = protected windward; Prot Lee = protected leeward



**Figure 1.** AGRRA survey sites in Grand Cayman and Little Cayman, Cayman Islands. See Table 1 for site codes.

# STATUS OF CORAL REEFS OF LITTLE CAYMAN AND GRAND CAYMAN, BRITISH WEST INDIES, IN 1999 (PART 2: FISHES)

BY

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## ABSTRACT

The fish assemblages at 33 sites around the islands of Grand Cayman and Little Cayman were assessed in June 1999 for the Atlantic and Gulf Rapid Reef Assessment initiative using belt transects and Roving Diver Technique surveys. A comprehensive species list, with 58 new records, was compiled for the Cayman Islands based on these data and survey data from the Reef Environmental Education Foundation database. In general, the reefs on Little Cayman appeared to support larger and more individual fishes than those of Grand Cayman. A multidimensional scaling ordination plot showed no clear island pattern but did reveal that the windward or leeward location of each site was an important factor affecting fish community composition. All but two sites followed a pattern of distinct windward and leeward clusters, and these clusters also correlated to macroalgal abundance. The relationship between macroalgal abundance and herbivore density was analyzed and significant correlations were found with surgeonfishes (Acanthuridae) and parrotfishes (Scaridae) using multiple regression.

## INTRODUCTION

Fishes have the potential to provide sensitive indices of reef health. Certain predatory fish species dominate the top of coral reef food webs, hence their density reflects a vast number of human and natural disturbances from habitat alteration to direct exploitation (Ferreira et al., 1998). Similarly, the presence and abundance of herbivorous fishes affect algal composition and cover (Ogden and Lobel, 1978).

In response to concerns about the widespread deterioration of reef condition in the Caribbean basin, the Atlantic and Gulf Rapid Reef Assessment (AGRRA) initiative was designed to provide a regional perspective using a standardized methodology. The rapid assessment protocol is focused on three main components of the reef community: stony corals, fish, and algae. As part of this initiative, the reefs of Grand Cayman (GC) and Little Cayman (LC) were assessed in June 1999.

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The Cayman Islands are a British Crown Colony located in the western Caribbean. The three islands lie between 19° 15' and 19° 45' N latitude and between 79° 44' and 81° 27' W longitude (Fig. 1). GC is the largest and most populous. LC lies approximately 145 km to its east-northeast and is about 10 km from Cayman Brac. The three islands are limestone, horst-and-graben structures associated with the Cayman Ridge (Jones, 1994). Freshwater is scarce and the islands lack rivers and streams. The fringing reefs that surround most of the islands contain shallow reef crests (rubble ramparts) as well as mid-shelf and shelf-edge fore reefs (Blanchon and Jones, 1997). These fringing reefs are particularly well-developed on the windward (eastern and southern) coasts of both islands. Other submerged benthic habitats include seagrass beds and mangrove fringes.

The level of human disturbance on GC is significantly greater than on LC, which is relatively remote and undeveloped. Anthropogenic impacts on GC reefs include habitat destruction from anchors and increased suspended sediment load from dredging and mangrove removal. Fishing pressure is considerably greater on GC than around LC. Five spawning aggregations of Nassau grouper (*Epinephelus striatus*) have been heavily harvested (during the 2002 spawning season, all but one had been depleted). Five hundred local residents are licensed to snorkel with spearguns. Fish pots (Antillean Z-traps) probably represent the biggest threat to the fish communities of both islands (personal observations).

An extensive marine park system was established in the Cayman Islands in 1986. Reefs in marine park and replenishment zone areas are protected from fish traps, spearguns, anchoring, and line fishing, although line fishing from shore and beyond the drop-off (shelf edge) is allowed. The Cayman Islands' Department of the Environment maintains a system of 257 permanent mooring buoys throughout the three islands.

The benthos of the Cayman Islands has been well studied, including descriptions of the coral communities, reef status, and analysis of spatial patterns (Roberts, 1988; Logan, 1994; Roberts, 1994). In contrast, apart from descriptions of Nassau grouper spawning aggregations (Colin et al., 1987; Tucker et al., 1993), there are few scientific descriptions of its reef fishes. However, Burgess et al.'s (1994) taxonomic review of collection expeditions contained an annotated list of 381 species known to occur in the Cayman Islands, including the endemic y-lined blenny (*Starksia y-lineata*) described by Gilbert (1965).

Since 1994, fish sighting and relative abundance data have been collected around the Cayman Islands as part of the Reef Environmental Education Foundation (REEF) Fish Survey Project, an ongoing volunteer monitoring effort. REEF volunteers use the Roving Diver Technique (RDT) (Schmitt and Sullivan, 1996) and the survey data are maintained in a publicly-accessible database. By the end of 2001, the REEF database contained over 40,000 surveys from over 2,000 sites, including approximately 2,200 surveys from the Cayman Islands.

This paper describes the fish assemblages of the Cayman Islands using the 1999 AGRRA data for GC and LC, along with REEF data from the two islands collected between 1994 and 2001. An updated species list and comparisons between islands and among sites are provided. The relationship between herbivorous fishes and macroalgal abundance is also investigated.

## METHODS

In June 1999, AGRRA fish and benthos surveys were simultaneously conducted at 15 sites on GC and 18 sites on LC (Fig. 1, Table 1). Sites were chosen by a mixed representative/strategic strategy: 12 were on the windward sides of the islands and 21 were on their leeward sides (the southwest side of GC was underrepresented). Six sites on LC and three on GC were located within marine park or replenishment zone areas. The benthic component is reported by Manfrino et al. (this volume). To assess the fishes, the AGRRA protocol Version 2.1 was used (Appendix One, this volume). At each site, a team of three (occasionally two) divers conducted at least 10 2 m x 30 m belt transects. Counts of serranids (groupers) were restricted to species of *Epinephelus* and *Mycteroperca*; scarids (parrotfishes) and haemulids (grunts) less than 5 cm in length were not tallied. Each diver also conducted a 45-60 minute RDT survey at each site. All fieldwork was undertaken between 9:00 a.m. and 3:00 p.m. Field identifications were based on Humann (1994), Stokes (1980), and Robins et al. (1986).

The fish transect data were entered into a custom AGRRA Excel spreadsheet. REEF provided the RDT data in ASCII format. Using the transects as replicates, the average density (#/100 m<sup>2</sup>) and size (cm) of each species and family were calculated for each site. Analyses were done at the regional (GC versus LC) and site levels, incorporating reef location (windward, leeward) and benthic parameters when appropriate. The average density and size of each species and family were compared between regions using a t-test after testing the data for normality. Due to confounding factors such as differences in use (e.g., recreation, harvest) and hydrographic features, comparisons between protected (marine park) and unprotected sites were not attempted. The site data were used in a hierarchical cluster analysis using Pearson's similarity index. The similarity matrix was generated using log-transformed density values for each species documented in at least three (10%) of the sites; the other 22 rare species were eliminated (per Grossman et al., 1982). A two-dimensional multidimensional scaling (MDS) ordination plot was also generated using the similarity matrix.

The transect data were also used to investigate interactions between the fish assemblages and the benthic community. This preliminary investigation was focused on herbivore/algae interactions. A regression was calculated on the densities of parrotfish and surgeonfish against percent absolute macroalgal abundance in quadrats (hereafter macroalgal abundance) and height at each site. Other coral factors (percent live coral cover, average colony height, percent diseased colonies) and environmental (windward/leeward) parameters were also plotted against each fish family. All values were transformed prior to regression (transformations were log+1 for fish density and algal height and arcsine of the square root for proportions).

The RDT survey data provided species lists, frequency of occurrence, and relative abundance estimates. Percent sighting frequency (%SF) for each species was the percentage of all dives in which the species was recorded. An estimate of abundance was calculated as: abundance score = D x %SF, where the density score (D) for each species was a weighted average index based on the frequency of observations in different abundance categories. Density score was calculated as:

$$D = ((n_S \times 1) + (n_F \times 2) + (n_M \times 3) + (n_A \times 4)) / (n_S + n_F + n_M + n_A)$$
 where  $n_S$ ,  $n_F$ ,  $n_M$ , and  $n_A$  represented the number of times each abundance category (Single, Few, Many,

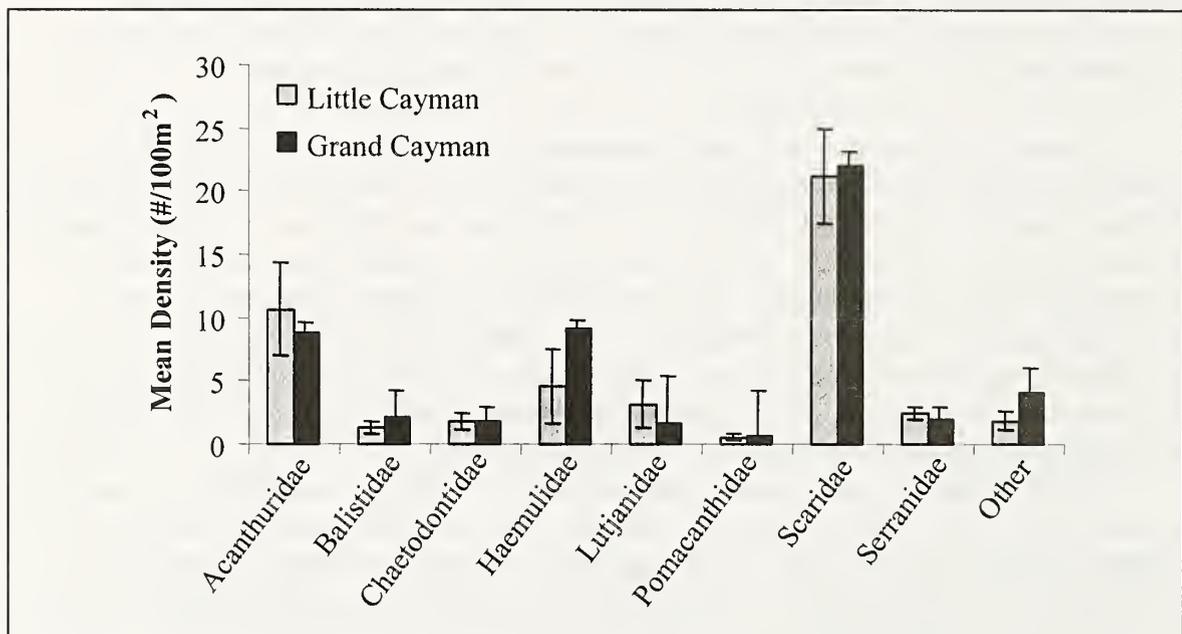
Abundant) was assigned for a given species. The RDT data were pooled and compared by island using the Wilcoxon Sign Rank test. Only species that were seen in at least 10% of the RDT AGRRA surveys were included in the analysis (103 species), reducing the effect of rare species (Grossman et al., 1982). SYSTAT 7.0 was used for all the analyses.

All expert-level REEF data from GC and LC, including the RDT data collected during the AGRRA expedition, were used to compile a species list of reef fishes for the Cayman Islands (REEF, 2001).

## RESULTS

A total of 341 transects (142 - GC; 199 - LC) and 79 RDT surveys (32- GC and 47- LC) documenting 173 species were conducted at 33 reefs (Table 1). The RDT survey data were added to the existing REEF database. The total number of species recorded by REEF experts on the Cayman Islands between 1994 and 2001 was 275 (Appendix A, this paper). When compared with Burgess et al.'s (1994) ichthyofaunal list, the REEF survey data added 58 new species records for a total of 423 reef fishes documented on the Cayman Islands (five freshwater species, 10 deepwater (>300 m) species, and a misidentification (*Stegastes mellis*) listed by Burgess et al. (1994) were not included in this tally). The 25 most common species, according to %SF in the REEF database, are noted in Table 2.

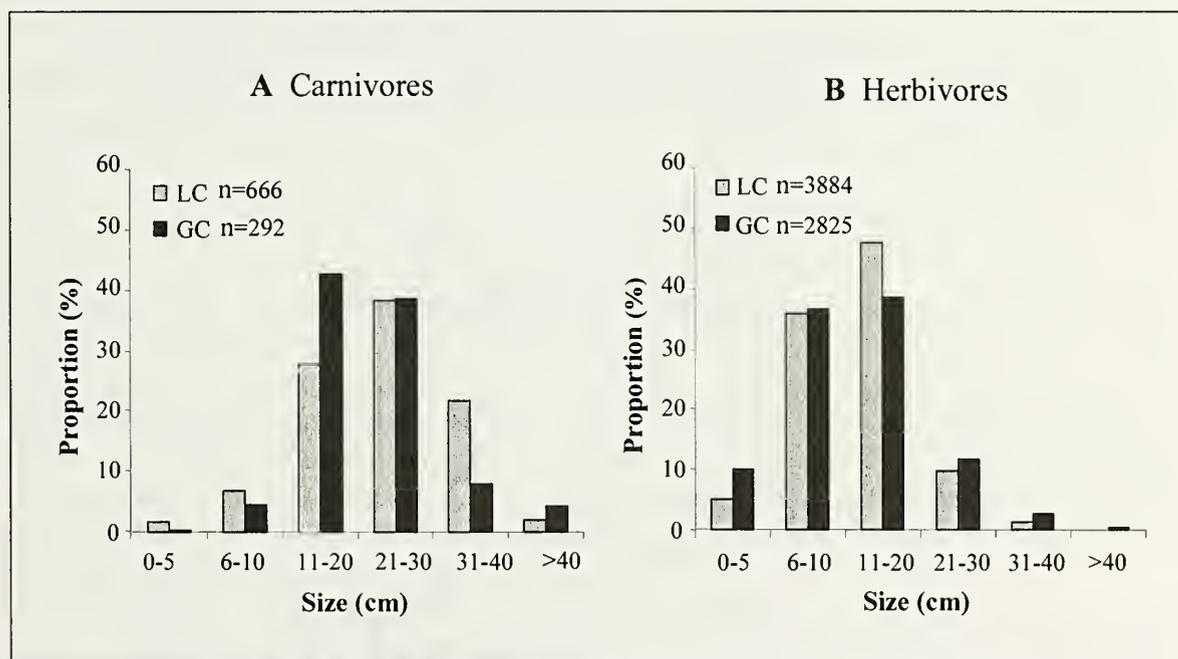
Parrotfish (Scaridae) was the most abundant family recorded during the belt transects (Fig. 2). Average density of snapper (Lutjanidae) on LC was approximately twice that of GC reefs. Size frequency distributions of carnivores (select grouper genera and all snappers) and herbivores [parrotfish  $\geq 5$  cm, surgeonfish (Acanthuridae), and the



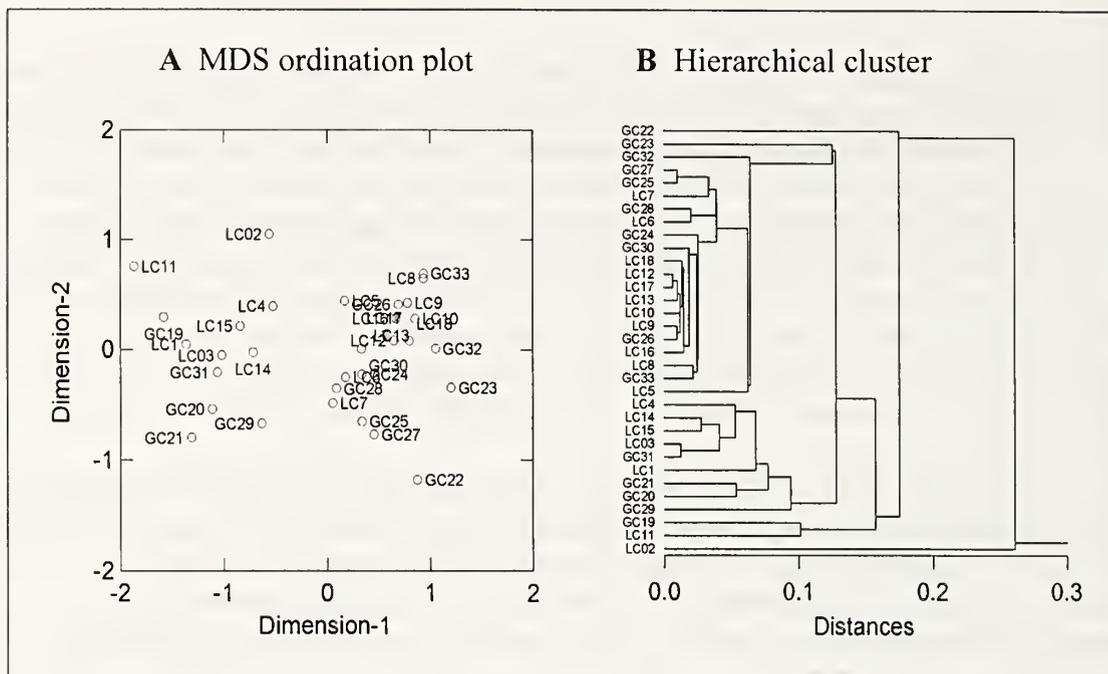
**Figure 2.** Mean fish density (no. individuals/100 m<sup>2</sup> ± sd) for AGRRA fishes in Grand Cayman and Little Cayman. Other = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Microspathodon chrysurus*, *Sphyraena barracuda*.

yellowtail damselfish *Microspathodon chrysurus*] are shown in Figure 3. Approximately 75% of the carnivores were less than 30 cm in length, and 85% of the herbivores were less than 20 cm in length. T-tests on these data showed that the average density and size for most species and families did not differ between islands. However, many species were reported in RDT surveys with greater than average abundance on the LC reefs (Wilcoxon Sign Rank  $p < 0.0005$ ). In particular, the sighting frequencies of six species of large groupers were considerably greater in LC (Table 3; Wilcoxon Sign Rank  $p < 0.05$ ). Exceptions included yellowtail snapper (*Ocyurus chrysurus*) and sergeant major (*Abudefduf saxatilis*), two species that become abundant when fed regularly by divers. Fish feeding is much more commonplace on GC reefs (Burgess et al., 1994; personal observations).

Site comparisons at the assemblage level showed no clear, intra-island groupings. However, two distinct clusters were obvious in the MDS plot (Fig. 4A) and, to a lesser extent, in the cluster diagram (Fig. 4B). The only environmental characteristics significantly related to fish density were reef location (windward/leeward) and macroalgal abundance. The windward (high-wave exposure) or leeward/protected windward (low wave-exposure) location of the sites was an important factor in the MDS cluster for all but two of the sites (LC02 and GC30). Leeward sites also had significantly higher macroalgal abundance than windward sites (45% versus 31%, respectively; F-test  $p < 0.001$ ; multiple  $R = 0.560$ ).

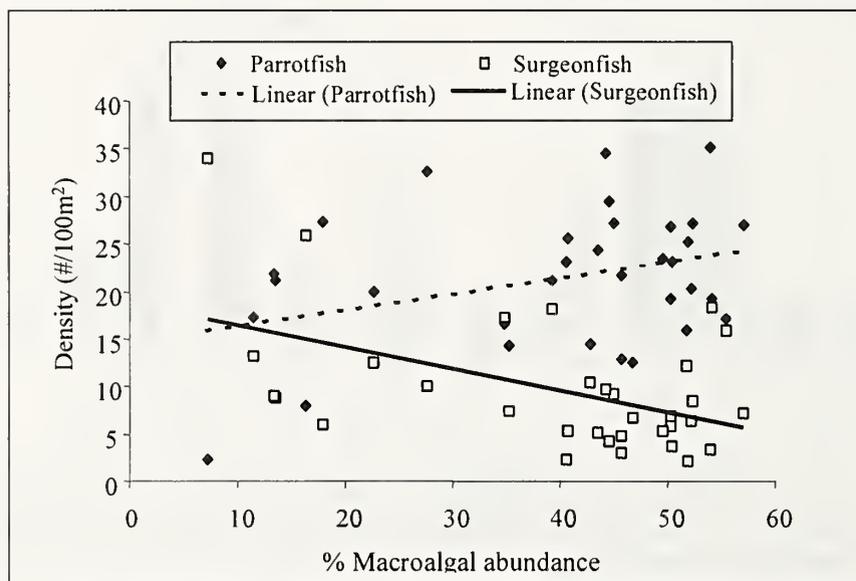


**Figure 3.** Size frequency distribution of (A) carnivores (all lutjanids, select serranids) and (B) herbivores (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) in GC (Grand Cayman) and LC (Little Cayman). Total number of individuals counted (n) is given.



**Figure 4.** (A) MDS ordination plot (left cluster is windward) and (B) hierarchical cluster analysis of AGRR reef fish transect data in GC (Grand Cayman) and LC (Little Cayman).

Surgeonfish density showed an inverse relationship with macroalgal abundance ( $p < 0.01$ ;  $r^2 = 0.209$ ), whereas parrotfish density was positively related to macroalgal abundance ( $p < 0.01$ ;  $r^2 = 0.215$ ) (Fig. 5). Adding macroalgal height to a multiple regression significantly improved the relationships with macroalgal abundance for parrotfish ( $p < 0.001$ ;  $r^2 = 0.413$ ) and surgeonfish ( $p = 0.001$ ;  $r^2 = 0.367$ ) densities. A strong inverse relationship between parrotfish and surgeonfish densities was also found ( $p < 0.001$ ;  $r^2 = 0.384$ ).



**Figure 5.** Regression plot between mean parrotfish density ( $\blacklozenge$ ) and mean surgeonfish density ( $\square$ ) (no. individuals/100m<sup>2</sup>) and mean absolute macroalgal abundance by site in the Cayman Islands.

## DISCUSSION

The reefs of the Cayman Islands support relatively diverse and abundant fish assemblages. This richness is probably a result of several factors including high local habitat diversity, a significant (34%) area of coastal reserves, and a reef system that is generally in fair condition (Manfrino et al., this volume). However, significant differences were revealed between GC and LC, most likely a result of the greater anthropogenic impacts on GC reefs. Higher harvest pressure on GC was reflected in the lower density and size of large groupers, parrotfishes and snappers (Table 4) and lower sighting frequencies of large groupers (Table 3). Analyses of RDT data indicated that regardless of commercial importance, the average abundance of most fish species was higher on LC, hence other factors, such as coastal development and water pollution, may also adversely impact fish communities on GC.

The site-level transect density data correlated most strongly with relative wave exposure (Fig. 4A,B). Macroalgae were significantly less abundant overall on windward (high-wave exposure) sites than on leeward and protected windward (low-wave exposure) sites, where parrotfish were the most abundant fishes in the transects. It is clear, however, that macroalgal abundance does not by itself adequately explain site-level assemblage composition, given that LC sites had only slight differences between wave-exposed and non wave-exposed sites (Manfrino et al., this volume). The correlation between fish communities at sites with similar wave exposure highlights the effect of physical parameters on fish assemblage structure, and should be taken into consideration in future analyses of fish data for the Cayman Islands.

In a simple system, one might expect the presence and density of herbivorous species to be negatively correlated with algal abundance and height. In other words, a site with many herbivorous fish would have relatively low algal abundance due to grazing. Our analysis at the site level indicates that this expectation holds true for surgeonfish. However, the inverse is evident in parrotfish. This implies either or both of the following: 1) there is a direct or indirect interaction between parrotfish and surgeonfish, or, more generally 2) the dynamic spatial and temporal characteristics of reef fish confound simple relationships between resource availability and fish abundance. Recent work on stoplight parrotfish (*Sparisoma viride*) indicates that whereas there are few, if any, direct interactions between surgeonfish and parrotfish, the use of space on the reef by individual fish is complex (territorial behavior, depth partitioning based on social grouping), and varies as a function of social status and intraspecific interactions (van Rooij et al., 1996a; van Rooij et al., 1996b). Clearly, more research is needed to understand the use of space by reef fish if accurate conclusions are to be drawn from relationships between fish abundance and benthic conditions.

One of the crucial tasks that scientists face in implementing a "reef health scale" using AGRRA data is to determine exactly what indicators within the collected data track health. An additional challenge lies in assessing how to evaluate and analyze the broad and complementary set of information collected on fishes, stony corals, and algae. Results from this paper and others in this volume will provide valuable insight on these issues. Due to the inherently complex nature of coral reef communities, the manner in which AGRRA data will dictate a scale of reef condition is most certainly also complex. The negative relationship between surgeonfish and parrotfish at the site level is a good

example of how community complexity may confound seemingly logical indicators of reef health such as herbivore biomass. Given our results, it is possible that Cayman Island reefs with similar herbivore biomass constituted by predominately different taxa may reflect dramatic differences in benthic conditions. The disparity between grouper abundance between the transect and RDT data and the dramatic increase in species reported in the Cayman Islands that resulted from the RDT surveys (18% based on the published list by Burgess et al., 1994) highlights the importance of using the two complementary visual fish-survey methods.

Because certain fish species dominate the top of coral reef food webs, a baseline of fish community composition and richness provides a useful tool for future assessment of reef health, given that a change in reef communities at lower trophic levels will most likely result in changes in the reef fish community composition (Choat, 1991; Jones et al., 1991). Additionally, because fish tend to be the most charismatic group of reef community members, changes in their community are most likely to be noticed and documented.

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Table 1. Site information for AGRRA fish surveys in Little Cayman and Grand Cayman Islands.

Name <sup>1</sup>	Site	Protection <sup>2</sup>	Location <sup>3</sup>	Reef Type <sup>4</sup>	Latitude ° ' N	Longitude ° ' W	Survey date	Depth (m)	% live stony coral cover (mean ± sd) <sup>5</sup>	30m fish transects (#)	RDT surveys (#) <sup>6</sup>	RDT fish species (#)
<b>Little Cayman</b>												
Jigsaw Puzzle	LC02	open	lee	High S&G	19 39.983'	80 06.390'	6-Jun-99	10.5	27.0 ± 9.5	11	4	96
Mixing Bowl	LC05	park	lee	Shelf edge	19 41.096'	80 04.700'	8-Jun-99	12.4	29.0 ± 12.0	12	2	105
Black Tip Tunnels	LC06	replen	lee	S&G	19 42.847'	79 57.470'	9-Jun-99	12.4	15.5 ± 7.5	12	2	74
Penguin's Leap	LC07	open	lee	Hardpan	19 42.551'	80 00.487'	9-Jun-99	16.1	16.5 ± 8.0	12	2	64
Meadows	LC08	park	lee	Patch	19 41.510'	80 04.130'	10-Jun-99	18.4	37.0 ± 11.5	14	3	88
Nancy's Cup of Tea	LC09	park	lee	Shelf edge	19 41.639'	80 04.137'	10-Jun-99	12.5	21.5 ± 14.0	14	3	87
Joy's Joy	LC10	park	lee	Shelf edge	19 40.690'	80 05.575'	10-Jun-99	12.1	19.0 ± 6.5	14	3	82
Paul's Anchor	LC12	park	lee	Shelf edge	19 41.661'	80 04.181'	13-Jun-99	12.9	17.0 ± 12.0	9	3	73
Rock Bottom Wall	LC13	open	lee	S&G	19 42.057'	80 03.421'	13-Jun-99	12.6	22.5 ± 10.5	10	2	70
	LC16	open	lee	S&G	19 39.702'	80 06.728'	15-Jun-99	14	25.5 ± 5.5	10	3	76
	LC17	open	lee	S&G	19 42.470'	80 00.495'	15-Jun-99	8.8	22.5 ± 4.5	10	3	71
Grundy's Gardens	LC18	replen	pro wind	Hardpan	19 42.996'	79 58.921'	15-Jun-99	10.8	15.0 ± 5.5	10	3	62
Disneyland	LC01	park	wind	S&G	19 39.421'	80 05.321'	6-Jun-99	9.4	37.5 ± 11.5	10	3	80
Charles Bay	LC03	replen	wind	S&G	19 49.831'	80 01.374'	7-Jun-99	9.9	25.5 ± 6.5	11	2	73
Main Channel East	LC04	open	wind	S&G	19 41.628'	79 58.459'	7-Jun-99	10.7	17.5 ± 6.5	11	2	76
Lucas's Ledge	LC11	open	wind	Fringing	19 39.412'	80 04.368'	11-Jun-99	2.8	16.0 ± 5.5	9	1	30
	LC14	replen	wind	S&G	19 40.155'	80 02.595'	14-Jun-99	13.6	24.5 ± 6.0	10	3	68
	LC15	replen	wind	S&G	19 40.628'	80 01.562'	14-Jun-99	10.9	26.5 ± 8.5	10	3	65
<b>Grand Cayman</b>												
Hepp's Mini Wall	GC28	park	lee	Patch/S&G	19 23.126'	81 24.992'	20-Jun-99	11.5	22.0 ± 5.5	5	1	35
Cemetery Reef	GC32	park	lee	Patch/S&G	19 21.917'	81 23.726'	22-Jun-99	9	17.5 ± 4.9	9	2	74
Sunset House	GC33	park	lee	Patch/S&G	19 17.172'	81 23.463'	22-Jun-99	9.1	22.5 ± 12.0	12	3	74
Isabel's Reef	GC22	open	pro wind	High S&G	19 21.460'	81 08.145'	18-Jun-99	10.6	24.5 ± 7.5	10	2	52
Babylon	GC23	replen	pro wind	S&G	19 21.200'	81 09.842'	18-Jun-99	9.5	18.0 ± 3.5	10	2	51
Delila's Delight	GC24	open	pro wind	S&G	19 21.518'	81 14.801'	19-Jun-99	7.1	23.0 ± 13.0	10	2	56
Queen's Throne	GC25	replen	pro wind	Hardpan	19 22.818'	81 17.493'	19-Jun-99	12.1	14.5 ± 3.0	10	2	61
	GC26	replen	pro wind	S&G	19 21.202'	81 11.746'	19-Jun-99	13.9	12.5 ± 4.0	10	2	49
	GC27	replen	pro wind	S&G	19 23.854'	81 21.617'	20-Jun-99	10.7	15.5 ± 3.5	5	1	46
Bear's Paw	GC19	open	wind	High S&G	19 19.058'	81 04.484'	17-Jun-99	6.7	23.0 ± 9.5	10	2	38
	GC20	open	wind	High S&G	19 20.002'	81 04.596'	17-Jun-99	8.9	18.0 ± 4.5	10	2	49

Table 1, Continued

Name	Site <sup>1</sup>	Protection <sup>2</sup>	Location <sup>3</sup>	Reef Type <sup>4</sup>	Latitude °N	Longitude °W	Survey date	Depth (m)	% live stony coral cover (mean ± sd) <sup>5</sup>	30m fish transects (#)	RDT surveys (#) <sup>6</sup>	RDT fish species (#)
Snapper Hole	GC21	open	wind	High S&G	19 20.634'	81 04.676'	18-Jun-99	10.1	24.0 ± 10.5	10	3	72
Breakers	GC30	open	wind	S&G	19 17.507'	81 12.069'	21-Jun-99	12.5	22.5 ± 4.5	10	2	59
Playing Fields	GC31	open	wind	High S&G	19 17.565'	81 06.318'	21-Jun-99	7.5	17.5 ± 4.5	10	3	71

<sup>1</sup>Site name given if it corresponds to a known Cayman Island Department of Environment buoy.

<sup>2</sup>Park = Marine Park Area; open = no protection; replen = Replenishment Zone Area

<sup>3</sup>lee = leeward; wind = windward; pro wind = protected windward

<sup>4</sup>Reef types follow Manfrino et al. (this volume), S&G = spur and groove, High S&G = high profile spur and groove.

<sup>5</sup>Cover values from benthic AGRRA transects (Manfrino et al., this volume)

<sup>6</sup>RDT = Roving Diver Technique

Table 2. Twenty-five most frequently sighted fish species on the Cayman Islands. Data (Sighting Frequency and Density Score) were compiled from the REEF database, using expert sightings from 1994 through 2001 (N=670 RDT Surveys).

Scientific name	Common name	Sighting frequency (%)	Density score <sup>1</sup>
<i>Acanthurus coeruleus</i>	Blue Tang	98	2.8
<i>Thalassoma bifasciatum</i>	Bluehead	97	3.3
<i>Chromis cyanea</i>	Blue Chromis	97	3.8
<i>Stegastes partitus</i>	Bicolor Damselfish	96	3.6
<i>Sparisoma viride</i>	Stoplight Parrotfish	96	2.7
<i>Caranx ruber</i>	Bar Jack	95	2.3
<i>Chromis multilineata</i>	Brown Chromis	95	3.4
<i>Gramma loreto</i>	Fairy Basslet	94	3.6
<i>Chaetodon capistratus</i>	Four-eye Butterflyfish	94	2.1
<i>Sparisoma aurofrenatum</i>	Redband Parrotfish	93	2.7
<i>Halichoeres garnoti</i>	Yellowhead Wrasse	92	2.7
<i>Canthigaster rostrata</i>	Sharpnose Puffer	92	2.1
<i>Epinephelus cruentatus</i>	Graysby	92	2.1
<i>Lutjanus apodus</i>	Schoolmaster	91	2.4
<i>Ocyurus chrysurus</i>	Yellowtail Snapper	89	2.5
<i>Haemulon flavolineatum</i>	French Grunt	89	2.3
<i>Scarus croicensis</i>	Striped Parrotfish	89	2.4
<i>Scarus taeniopterus</i>	Princess Parrotfish	88	2.5
<i>Clepticus parrae</i>	Creole Wrasse	87	3.7
<i>Melichthys niger</i>	Black Durgon	87	2.8
<i>Mulloidichthys martinicus</i>	Yellow Goatfish	84	2.4
<i>Holacanthus tricolor</i>	Rock Beauty	84	1.9
<i>Stegastes dienaecus</i>	Longfin Damselfish	82	2.3
<i>Epinephelus fubus</i>	Coney	81	2.1
<i>Stegastes planifrons</i>	Threespot Damselfish	80	2.2

<sup>1</sup>See Methods for definition of Density score.

Table 3. Mean percent sighting frequency of select groupers during AGRRA roving diver surveys in LC and GC, Cayman Islands.

Scientific name	Common name	Sighting frequency (%)	
		LC	GC
<i>Mycteroperca bonaci</i>	Black Grouper	7%	3%
<i>Epinephelus striatus</i>	Nassau Grouper	69%	9%
<i>Epinephelus guttatus</i>	Red Hind	44%	21%
<i>Mycteroperca tigris</i>	Tiger Grouper	50%	12%
<i>Mycteroperca venenosa</i>	Yellowfin Grouper	22%	0%
<i>Mycteroperca interstitialis</i>	Yellowmouth Grouper	6%	3%

Table 4. Density and length (mean  $\pm$  standard deviation) of AGRRA fishes, and macroalgal index values for LC and GC, Cayman Islands.

Island	Herbivores		Carnivores		Macroalgal index <sup>2</sup>	
	Acanthuridae	Scaridae ( $\geq 5$ cm)	Haemulidae ( $\geq 5$ cm)	Lutjanidae Serranidae <sup>1</sup>		
LC density (#/100m <sup>2</sup> )	10.67 $\pm$ 7.22	21.20 $\pm$ 7.56	4.59 $\pm$ 5.98	3.17 $\pm$ 3.61	2.38 $\pm$ 0.96	105
LC length (cm)	11.85 $\pm$ 2.08	15.87 $\pm$ 2.67	19.45 $\pm$ 2.58	27.78 $\pm$ 3.42	19.11 $\pm$ 3.02	
GC density (#/100m <sup>2</sup> )	8.35 $\pm$ 6.66	22.13 $\pm$ 7.04	8.72 $\pm$ 10.04	1.86 $\pm$ 2.04	2.00 $\pm$ 0.67	32
GC length (cm)	12.02 $\pm$ 1.60	15.33 $\pm$ 2.19	19.21 $\pm$ 1.36	28.58 $\pm$ 7.29	18.95 $\pm$ 1.92	

<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.<sup>2</sup>Macroalgal index = absolute macroalgal abundance x canopy height.

Appendix A. Cayman Islands Species List. Data compiled from the REEF database, using expert sightings from 1994 through 2001. A total of 670 expert surveys (32- Cayman Brac; 258- Little Cayman; 380- Grand Cayman) reported 276 species. For each species, percent sighting frequency (%SF) and density score (DEN) are given. Fifty-eight species previously unreported from the Cayman Islands are listed and indicated by an asterisk (\*).

Scientific Name	Common Name	SF%	DEN	Scientific Name	Common Name	SF%	DEN
<b>Acanthuridae</b>	<b>Surgonfishes</b>			<b>Belontiidae</b>	<b>Needlefishes</b>		
<i>Acanthurus bahianus</i>	Ocean Surgeonfish	84%	2.3	* <i>Polybelone argalus</i>	Keeltail Needlefish	0.1%	2.0
<i>Acanthurus chirurgus</i>	Doctofish	39%	2.0	<i>Tylosurus crocodilus</i>	Houndfish	3%	1.6
<i>Acanthurus coeruleus</i>	Blue Tang	98%	2.9	<b>Blenniidae</b>	<b>Blennies (Combtooth)</b>		
<b>Apogonidae</b>	<b>Cardinalfishes</b>			<i>Entomacrodus nigricans</i>	Pearl Blenny	0.3%	1.0
* <i>Apogon affinis</i>	Bigtooth Cardinalfish	0.1%	2.0	<i>Ophioblennius atlanticus</i>	Redlip Blenny	32%	1.9
* <i>Apogon aurolineatus</i>	Bridle Cardinalfish	0.4%	1.6	* <i>Parablennius marmoratus</i>	Seaweed Blenny	1%	1.3
<i>Apogon binotatus</i>	Barred Cardinalfish	17%	2.0	<b>Bothidae</b>	<b>Flounders (Lefteye)</b>		
<i>Apogon lachneri</i>	Whitestar Cardinalfish	17%	2.0	<i>Bothus ocellatus</i>	Eyed Flounder	0.3%	1.0
<i>Apogon maculatus</i>	Flamefish	14%	1.7	<i>Bothus lunatus</i>	Peacock Flounder	12%	1.2
* <i>Apogon pillionatus</i>	Broadsaddle Cardinalfish	SO <sup>1</sup>		<b>Callionymidae</b>	<b>Dragonets</b>		
<i>Apogon planifrons</i>	Pale Cardinalfish	1%	1.8	<i>Paradiplogrammus bairdi</i>	Lancer Dragonet	3%	1.3
<i>Apogon pseudomaculatus</i>	Twospot Cardinalfish	2%	1.6	<b>Carangidae</b>	<b>Jacks</b>		
<i>Apogon quadrisquamatus</i>	Sawcheek Cardinalfish	1%	1.8	<i>Alectis ciliaris</i>	African Pompano	SO <sup>1</sup>	
<i>Apogon towsendi</i>	Belted Cardinalfish	26%	2.2	<i>Caranx bartholomaei</i>	Yellow Jack	3%	1.6
<i>Astrapogon puncticulatus</i>	Blackfin Cardinalfish	0.4%	2.0	<i>Caranx crysos</i>	Blue Runner	1%	1.4
<i>Phaeoptyx pigmentaria</i>	Dusky Cardinalfish	3%	2.3	<i>Caranx hippos</i>	Crevalle Jack	2%	1.9
* <i>Phaeoptyx xenus</i>	Sponge Cardinalfish	11%	1.6	<i>Caranx latus</i>	Horse-Eye Jack	33%	2.3
<b>Aulostomidae</b>	<b>Trumpetfishes</b>			<i>Caranx lugubris</i>	Black Jack	8%	1.4
<i>Aulostomus maculatus</i>	Trumpetfish	69%	1.7	<i>Caranx ruber</i>	Bar Jack	95%	2.3
<b>Balistidae</b>	<b>Leatherjackets</b>			<i>Elagatis bipinnulata</i>	Rainbow Runner	0.4%	1.3
<i>Aluterus scriptus</i>	Scrawled Filefish	9%	1.1	<i>Scomberomorus regalis</i>	Cero	18%	1.3
<i>Balistes vetula</i>	Queen Triggerfish	34%	1.3	<i>Trachinotus falcatus</i>	Permit	3%	1.2
<i>Cantherhines macrocerus</i>	Whitespotted Filefish	22%	1.4	<i>Trachinotus goodei</i>	Palometa	0.3%	2.0
<i>Cantherhines pultus</i>	Orangespotted Filefish	13%	1.3	<b>Carcharhinidae</b>	<b>Sharks (Requiem)</b>		
<i>Canthidermis sufflamen</i>	Ocean Triggerfish	11%	1.3	* <i>Carcharhinus limbatus</i>	Blacktip Shark	0.3%	1.5
<i>Melichthys niger</i>	Black Durgon	87%	2.8	* <i>Carcharhinus perezii</i>	Reef Shark	2%	1.1
<i>Monacanthus tockeri</i>	Slender Filefish	8%	1.4	<b>Chaenopsidae</b>	<b>Blennies (Tube)</b>		
				<i>Acanthemblemaria aspera</i>	Roughhead Blenny	11%	1.5

## Appendix A, Continued

Scientific Name	Common Name	SF%	DEN	Scientific Name	Common Name	SF%	DEN
<b>Chaenopsidae (cont.)</b>	<b>Blennies (Tube)</b>			<b>Gerreidae (cont.)</b>	<b>Mojarra</b>		
<i>*Acanthemblemaria chaplini</i>	Papillose Blenny	0.1%	1.0	<i>*Eucinostomus gula</i>	Silver Jenny	0.1%	2.0
<i>Acanthemblemaria maria</i>	Secretary Blenny	16%	1.6	<i>Eucinostomus jonesi</i>	Slender Mojarra	0.1%	2.0
<i>Emblemaria pandionis</i>	Sailfin Blenny	14%	1.7	<i>Gerres cinereus</i>	Yellowfin Mojarra	11%	1.9
<i>Emblemaropsis</i> sp.	Darkhead Blenny	4%	1.2	<b>Gobiesocidae</b>	<b>Clingfishes</b>		
<i>Lucayablenniuss zingaro</i>	Arrow Blenny	19%	1.5	<i>Arcos rubiginosus</i>	Red Clingfish	1%	1.4
<b>Chaetodontidae</b>	<b>Butterflyfishes</b>			<i>Gobiesox punctulatus</i>	Stippled Clingfish	0.1%	1.0
<i>Chaetodon aculeatus</i>	Longsnout Butterflyfish	26%	1.5	<b>Gobiidae</b>	<b>Gobies</b>		
<i>Chaetodon capistratus</i>	Four-eye Butterflyfish	94%	2.1	<i>Coryphopterus</i>	Masked/Glass Goby	82%	3.7
<i>Chaetodon ocellatus</i>	Spotfin Butterflyfish	38%	1.7	<i>personatus/hyalinus</i>			
<i>Chaetodon sedentarius</i>	Reef Butterflyfish	1%	1.1	<i>Coryphopterus dicrus</i>	Colon Goby	3%	1.2
<i>Chaetodon striatus</i>	Banded Butterflyfish	73%	1.8	<i>Coryphopterus eidolon</i>	Pallid Goby	20%	1.6
<b>Cirrihitidae</b>	<b>Hawkfishes</b>			<i>Coryphopterus glaucofraenum</i>	Bridled Goby	63%	2.3
<i>Amblycirrhitis pinos</i>	Redspotted Hawkfish	29%	1.3	<i>Coryphopterus lipernes</i>	Peppermint Goby	46%	1.9
<b>Congridae</b>	<b>Eels (Conger)</b>			<i>Gnatholepis thompsoni</i>	Goldspot Goby	69%	2.4
<i>Heteroconger halis</i>	Brown Garden Eel	22%	3.5	<i>Gobionellus saepepallens</i>	Dash Goby	1%	1.4
<b>Dactylopteridae</b>	<b>Flying Gurnards</b>			<i>*Gobiosoma chancei</i>	Shortstripe Goby	3%	1.7
<i>Dactylopterus volitans</i>	Flying Gurnard	0.3%	1.5	<i>Gobiosoma dilepsis</i>	Orangesided Goby	28%	1.8
<b>Dasyatidae</b>	<b>Rays (Sting)</b>			<i>Gobiosoma evelynae</i>	Sharknose Goby	20%	1.9
<i>Dasyatis americana</i>	Southern Stingray	23%	1.3	<i>Gobiosoma genie</i>	Cleaning Goby	49%	2.4
<b>Echeneidae</b>	<b>Remoras</b>			<i>Gobiosoma horsti</i>	Yellowline Goby	45%	1.9
<i>Echeneis naucrates</i>	Sharksucker	5%	1.1	<i>Gobiosoma louisae</i>	Spotlight Goby	6%	1.7
<b>Elopidae</b>	<b>Tarpon</b>			<i>Gobiosoma multifasciatum</i>	Greenbanded Goby	6%	2.0
<i>Megalops atlanticus</i>	Tarpon	22%	2.2	<i>Gobiosoma pallens</i>	Semiscald Goby	4%	1.3
<b>Ephippidae</b>	<b>Spadefishes</b>			<i>*Gobiosoma saucrum</i>	Leopard Goby	0.3%	1.5
<i>*Chaetodipterus faber</i>	Atlantic Spadefish	1%	1.2	<i>*Gobiosoma xanthiprora</i>	Yellowprowl Goby	0.1%	2.0
<b>Exocoetidae</b>	<b>Flyingfishes/Halfbeaks</b>			<i>*Ioglossus helenae</i>	Hovering Goby	2%	1.4
<i>*Hemiramphus balao</i>	Balao	0.1%	3.0	<i>*Microgobius carri</i>	Seminole Goby	0.1%	1.0
<i>*Hirundichthys speculiger</i>	Mirrorwing Flyingfish	SO <sup>1</sup>		<i>*Nes longus</i>	Orangespotted Goby	0.3%	1.5
<b>Fistulariidae</b>	<b>Cornetfishes</b>			<i>Priolepis hipolitii</i>	Rusty Goby	3%	1.2
<i>Fistularia tabacaria</i>	Bluespotted Cornetfish	1%	1.5	<i>Risor ruber</i>	Tusked Goby	1%	1.5
<b>Gerreidae</b>	<b>Mojarra</b>			<b>Grammatidae</b>	<b>Basslets</b>		
<i>*Eucinostomus melanopterus</i>	Flagfin Mojarra	0.3%	2.0	<i>*Gramma linki</i>	Yellowcheek Basslet	0.1%	1.0

Scientific Name	Common Name	SF%	DEN	Scientific Name	Common Name	SF%	DEN
<b>Grammatidae (cont.)</b>	<b>Basslets</b>			<b>Labridae (cont.)</b>	<b>Wrasses</b>		
<i>Gramma melacara</i>	Blackcap Basslet	34%	3.4	<i>Doratonotus megalepis</i>	Dwarf Wrasse	1%	1.0
<i>Lipogramma trilineatum</i>	Threeline Basslet	2%	1.4	<i>Halichoeres bivittatus</i>	Slippery Dick	42%	2.2
<b>Haemulidae</b>	<b>Grunts</b>			* <i>Halichoeres cyanocephalus</i>	Yellowcheek Wrasse	2%	1.1
<i>Anisotremus surinamensis</i>	Black Margate	3%	1.5	<i>Halichoeres garnoti</i>	Yellowhead Wrasse	93%	2.8
<i>Anisotremus virginicus</i>	Porkfish	1%	1.3	<i>Halichoeres maculipinna</i>	Clown Wrasse	43%	2.0
<i>Haemulon album</i>	White Margate	17%	1.5	* <i>Halichoeres pictus</i>	Rainbow Wrasse	16%	2.0
<i>Haemulon aurolineatum</i>	Tomtate	6%	2.3	* <i>Halichoeres poeyi</i>	Blackear Wrasse	0.1%	2.0
<i>Haemulon carbonarium</i>	Caesar Grunt	26%	2.3	<i>Halichoeres radiatus</i>	Puddingwife	19%	1.4
<i>Haemulon chrysargyreum</i>	Smallmouth Grunt	1%	2.5	* <i>Hemipteronotus martinicensis</i>	Rosy Razorfish	19%	2.2
<i>Haemulon flavolineatum</i>	French Grunt	89%	2.3	<i>Hemipteronotus splendens</i>	Green Razorfish	28%	1.9
* <i>Haemulon macrostomum</i>	Spanish Grunt	3%	1.7	<i>Lachnolaimus maximus</i>	Hogfish	44%	1.3
* <i>Haemulon melanurum</i>	Cottonwick	1%	1.7	<i>Thalassoma bifasciatum</i>	Bluehead	97%	3.3
<i>Haemulon parra</i>	Sailors Choice	18%	2.2	<b>Labrisomidae</b>	<b>Blennies (Scaly)</b>		
<i>Haemulon plumieri</i>	White Grunt	69%	1.8	<i>Acanthemblemaria spinosa</i>	Spinyhead Blenny	4%	1.4
<i>Haemulon sciurus</i>	Bluestriped Grunt	65%	2.0	<i>Chaenopsis limbaughi</i>	Yellowface Pikeblenny	1%	1.0
* <i>Haemulon striatum</i>	Striped Grunt	0.3%	2.0	* <i>Hemimblemaria simulus</i>	Wrasse Blenny	1%	1.2
<b>Holocentridae</b>	<b>Squirrelfishes</b>			* <i>Labrisomus filamentosus</i>	Quillfin Blenny	0%	2.0
<i>Holocentrus adscensionis</i>	Squirrelfish	57%	1.9	<i>Labrisomus gobio</i>	Palehead Blenny	0.1%	1.0
<i>Holocentrus coruscum</i>	Reef Squirrelfish	3%	1.7	* <i>Labrisomus kalisherae</i>	Downy Blenny	0.1%	1.0
<i>Holocentrus marianus</i>	Longjaw Squirrelfish	69%	2.0	<i>Malacocentrus aurolineatus</i>	Goldline Blenny	2%	1.7
<i>Holocentrus rufus</i>	Longspine Squirrelfish	68%	2.0	<i>Malacocentrus boehlkei</i>	Diamond Blenny	25%	1.5
<i>Holocentrus vexillarius</i>	Dusky Squirrelfish	19%	1.9	<i>Malacocentrus macropus</i>	Rosy Blenny	7%	1.5
<i>Myripristis jacobus</i>	Blackbar Soldierfish	54%	2.0	<i>Malacocentrus triangulatus</i>	Saddled Blenny	75%	2.3
<i>Plectrypops retrospinis</i>	Cardinal Soldierfish	2%	1.2	<i>Malacocentrus versicolor</i>	Barfin Blenny	0.1%	1.0
<b>Inermiidae</b>	<b>Bonnetmouths</b>			<i>Starksia nanodes</i>	Dwarf Blenny	0.1%	2.0
* <i>Emmelichthys atlanticus</i>	Bonnetmouth	0.4%	3.0	<b>Lutjanidae</b>	<b>Snappers</b>		
* <i>Inermia vittata</i>	Boga	17%	3.0	<i>Apsilus dentatus</i>	Black Snapper	0.6%	1.2
<b>Kyphosidae</b>	<b>Chubs</b>			<i>Lutjanus analis</i>	Mutton Snapper	59%	1.5
<i>Kyphosus sectatrix/incisor</i>	Bermuda/Yellow Chub	78%	2.4	<i>Lutjanus apodus</i>	Schoolmaster	91%	2.4
<b>Labridae</b>	<b>Wrasses</b>			<i>Lutjanus buccanella</i>	Blackfin Snapper	17%	2.0
<i>Bodianus pulchellus</i>	Spotfin Hogfish	1%	1.5	<i>Lutjanus cyanopterus</i>	Cubera Snapper	3%	1.2
<i>Bodianus rufus</i>	Spanish Hogfish	74%	1.9	<i>Lutjanus griseus</i>	Gray Snapper	4%	1.9
<i>Clepticus parrae</i>	Creole Wrasse	87%	3.7	<i>Lutjanus jocu</i>	Dog Snapper	10%	1.3

Scientific Name	Common Name	SF%	DEN	Scientific Name	Common Name	SF%	DEN
<b>Lutjanidae (cont.)</b>	<b>Snappers</b>			<b>Pemppheridae</b>	<b>Sweepers</b>		
<i>Lutjanus mahogoni</i>	Mahogany Snapper	69%	2.2	<i>Pemppheris schomburgki</i>	Glassy Sweeper	2%	2.2
<i>Lutjanus synagris</i>	Lane Snapper	2%	1.9	<b>Pomacanthidae</b>	<b>Angelfishes</b>		
<i>Ocyurus chrysurus</i>	Yellowtail Snapper	89%	2.5	<i>Centropyge argi</i>	Cherubfish	1%	1.2
<b>Matacanthidae</b>	<b>Tilefishes</b>			* <i>Holacanthus bermudensis</i>	Blue Angelfish	0.1%	1.0
<i>Malacanthus plumieri</i>	Sand Tilefish	74%	2.0	<i>Holacanthus ciliaris</i>	Queen Angelfish	35%	1.4
<b>Mobulidae</b>	<b>Ray (Mantas)</b>			<i>Holacanthus tricolor</i>	Rock Beauty	84%	1.9
<i>Manta birostris</i>	Manta	0.3%	1.0	<i>Pomacanthus arcuatus</i>	Gray Angelfish	39%	1.4
<b>Mullidae</b>	<b>Goatfishes</b>			<i>Pomacanthus paru</i>	French Angelfish	51%	1.3
<i>Mulloidichthys martinicus</i>	Yellow Goatfish	84%	2.4	<b>Pomacentridae</b>	<b>Damselfishes</b>		
<i>Pseudupeneus maculatus</i>	Spotted Goatfish	46%	1.7	<i>Abudefduf saxatilis</i>	Sergeant Major	49%	2.5
<b>Muranidae</b>	<b>Eels (Moray)</b>			<i>Abudefduf taurus</i>	Night Sergeant	0.4%	2.0
<i>Echidna catenata</i>	Chain Moray	0.1%	1.0	<i>Chromis cyanea</i>	Blue Chromis	97%	3.8
<i>Enchelycore carychroa</i>	Chestnut Moray	0.1%	1.0	* <i>Chromis insolata</i>	Sunshinefish	18%	2.2
<i>Gymnothorax funebris</i>	Green Moray	5%	1.0	<i>Chromis multilineata</i>	Brown Chromis	95%	3.4
<i>Gymnothorax militaris</i>	Goldentail Moray	4%	1.0	<i>Microspathodon chrysurus</i>	Yellowtail Damselfish	57%	2.0
<i>Gymnothorax moringa</i>	Spotted Moray	3%	1.2	<i>Stegastes dienaenus</i>	Longfin Damselfish	82%	2.3
<i>Gymnothorax vicinus</i>	Purplemouth Moray	0.3%	2.0	<i>Stegastes fuscus</i>	Dusky Damselfish	18%	1.8
<b>Myliobatidae</b>	<b>Rays (Eagle)</b>			<i>Stegastes leucostictus</i>	Beaugregory	35%	1.8
<i>Aetobatus narinari</i>	Spotted Eagle Ray	9%	1.1	<i>Stegastes partitus</i>	Bicolor Damselfish	96%	3.6
<b>Ogcocephalidae</b>	<b>Batfishes</b>			<i>Stegastes planifrons</i>	Threespot Damselfish	80%	2.2
<i>Ogcocephalus nasutus</i>	Shortnose Batfish	SO <sup>1</sup>		<i>Stegastes variabilis</i>	Cocoa Damselfish	17%	1.5
<b>Ophichthidae</b>	<b>Eels (Snake)</b>			<b>Priacanthidae</b>	<b>Bigeyes</b>		
<i>Myrichthys breviceps</i>	Sharptail Eel	0.3%	1.0	<i>Priacanthus arenatus</i>	Bigeye	0.4%	1.0
<b>Opistognathidae</b>	<b>Jawfishes</b>			<i>Priacanthus cruentatus</i>	Glasseye Snapper	9%	1.3
* <i>Opistognathus aurifrons</i>	Yellowhead Jawfish	42%	2.1	<b>Rhincodontidae</b>	<b>Sharks (Carpet)</b>		
* <i>Opistognathus macrognathus</i>	Banded Jawfish	0.3%	1.5	<i>Ginglymostoma cirratum</i>	Nurse Shark	5%	1.0
* <i>Opistognathus whitehursti</i>	Dusky Jawfish	0.3%	1.5	<b>Scaridae</b>	<b>Parrotfishes</b>		
<b>Ostraciontidae</b>	<b>Boxfishes</b>			<i>Cryptotomus roseus</i>	Bluelip Parrotfish	6%	1.9
<i>Lactophrys bicaudalis</i>	Spotted Trunkfish	30%	1.1	<i>Scarus coelestinus</i>	Midnight Parrotfish	8%	1.2
<i>Lactophrys polygona</i>	Honeycomb Cowfish	26%	1.2	<i>Scarus coeruleus</i>	Blue Parrotfish	2%	1.5
<i>Lactophrys quadricornis</i>	Scrawled Cowfish	1%	1.6	<i>Scarus croicensis</i>	Striped Parrotfish	89%	2.4
<i>Lactophrys trigonus</i>	Trunkfish	0.4%	1.3	<i>Scarus guacamaia</i>	Rainbow Parrotfish	15%	1.3
<i>Lactophrys triquetus</i>	Smooth Trunkfish	23%	1.2	<i>Scarus taeniopterus</i>	Princess Parrotfish	88%	2.5

Scientific Name	Common Name	SF%	DEN	Scientific Name	Common Name	SF%	DEN
<b>Scariidae (cont.)</b>	<b>Parrotfishes</b>			<b>Serranidae (cont.)</b>	<b>Sea Bases</b>		
<i>Scarus vetula</i>	Queen Parrotfish	59%	1.9	* <i>Liopropoma carmabi</i>	Candy Bass	1%	1.0
* <i>Sparisoma atomarium</i>	Greenblotch Parrotfish	41%	2.2	<i>Liopropoma mowbrayi</i>	Cave Bass	3%	1.3
<i>Sparisoma aurofrenatum</i>	Redband Parrotfish	93%	2.7	<i>Liopropoma rubre</i>	Peppermint Bass	19%	1.4
<i>Sparisoma chrysopteryum</i>	Redtail Parrotfish	62%	2.0	<i>Mycteroperca bonaci</i>	Black Grouper	11%	1.2
<i>Sparisoma radians</i>	Bucktooth Parrotfish	2%	2.2	<i>Mycteroperca interstitialis</i>	Yellowmouth Grouper	6%	1.1
<i>Sparisoma rubripinne</i>	Yellowtail Parrotfish	46%	2.0	* <i>Mycteroperca phenax</i>	Scamp	1%	1.2
<i>Sparisoma viride</i>	Stoplight Parrotfish	96%	2.7	<i>Mycteroperca tigris</i>	Tiger Grouper	54%	1.5
<b>Sciaenidae</b>	<b>Drums</b>			<i>Mycteroperca venenosa</i>	Yellowfin Grouper	18%	1.3
<i>Equetus acuminatus</i>	Highhat	1%	1.5	* <i>Paranithias furcifer</i>	Creole-fish	1%	2.0
<i>Equetus lanceolatus</i>	Jackknife Fish	0.1%	1.0	<i>Rypticus saponaceus</i>	Greater Soapfish	18%	1.2
<i>Equetus punctatus</i>	Spotted Drum	15%	1.2	* <i>Serranus baldwini</i>	Lantern Bass	11%	1.5
<i>Odontoscia dentex</i>	Reef Croaker	0.4%	1.6	<i>Serranus tabacarius</i>	Tobaccofish	53%	1.8
<b>Scorpiionidae</b>	<b>Scorpiionfishes</b>			<i>Serranus tigrinus</i>	Harlequin Bass	75%	1.9
<i>Scorpaena plumieri</i>	Spotted Scorpiionfish	6%	1.1	* <i>Serranus tortugarum</i>	Chalk Bass	7%	1.7
<i>Scorpaenodes caribbaeus</i>	Reef Scorpiionfish	1%	1.0	<b>Sparidae</b>	<b>Porgies</b>		
<b>Serranidae</b>	<b>Sea Bases</b>			<i>Calamus bajonado</i>	Jolthead Porgy	15%	1.2
<i>Epinephelus adscensionis</i>	Rock Hind	2%	1.1	<i>Calamus calamus</i>	Saucereye Porgy	27%	1.3
<i>Epinephelus cruentatus</i>	Graysby	92%	2.1	* <i>Calamus pennatula</i>	Pluma	1%	1.1
<i>Epinephelus fulvus</i>	Coney	81%	2.1	<b>Sphyraenidae</b>	<b>Barracudas</b>		
<i>Epinephelus guttatus</i>	Red Hind	27%	1.3	<i>Sphyraena barracuda</i>	Great Barracuda	49%	1.5
<i>Epinephelus itajara</i>	Goliath Grouper	0.6%	1.2	<i>Sphyraena picudilla</i>	Southern Sennet	1%	3.5
* <i>Epinephelus morio</i>	Red Grouper	0.3%	1.5	<b>Sphyrnidae</b>	<b>Sharks (Hammerhead)</b>		
<i>Epinephelus striatus</i>	Nassau Grouper	50%	1.5	* <i>Sphyrna lewini</i>	Scalloped Hammerhead	SO <sup>1</sup>	
<i>Hypoplectrus aberrans</i>	Yellowbelly Hamlet	4%	1.2	<b>Syngnathidae</b>	<b>Pipefishes/Seahorses</b>		
* <i>Hypoplectrus chlorurus</i>	Yellowtail Hamlet	0.3%	1.0	* <i>Acentronura dendritica</i>	Pipehorse	0.4%	1.6
<i>Hypoplectrus gummigutta</i>	Golden Hamlet	0.4%	1.0	* <i>Cosmocampus albirostris</i>	Whitnose Pipefish	0.1%	1.0
<i>Hypoplectrus guttavarius</i>	Shy Hamlet	21%	1.3	* <i>Hippocampus erectus</i>	Lined Seahorse	0.4%	1.0
<i>Hypoplectrus indigo</i>	Indigo Hamlet	9%	1.3	<b>Synodontidae</b>	<b>Lizardfishes</b>		
<i>Hypoplectrus nigricans</i>	Black Hamlet	14%	1.4	<i>Synodus intermedius</i>	Sand Diver	5%	1.1
<i>Hypoplectrus puella</i>	Barred Hamlet	64%	1.8	* <i>Synodus saurus</i>	Bluestriped Lizardfish	0.3%	1.0
* <i>Hypoplectrus sp.</i>	Masked Hamlet	2%	1.0	<i>Synodus synodus</i>	Red Lizardfish	0.4%	1.0
* <i>Hypoplectrus sp.</i>	Tan Hamlet	2%	1.0	<b>Tetrodontidae</b>	<b>Puffers</b>		
<i>Hypoplectrus unicolor</i>	Butter Hamlet	19%	1.3	<i>Canthigaster rostrata</i>	Sharpnose Puffer	92%	2.1

Scientific Name	Common Name	SF%	DEN	Scientific Name	Common Name	SF%	DEN
<b>Tetrodontidae (cont.)</b>				<b>Tripterygiidae</b>	<b>Blennies (Tripplefin)</b>		
* <i>Chilomycterus antennatus</i>	Bridled Burrfish	0.3%	1.0	<i>Enneanectes altivelis</i>	Lofty Triplefin	2%	1.0
<i>Diodon holocanthus</i>	Balloonfish	4%	1.0	<i>Enneanectes atrorus</i>	Blackedge Triplefin	0.1%	1.0
* <i>Diodon hystrix</i>	Porcupinefish	13%	1.1	<i>Enneanectes boehlkei</i>	Roughhead Triplefin	2%	1.4
<i>Sphoeroides spengleri</i>	Bandtail Puffer	5%	1.2	<i>Enneanectes pectoralis</i>	Redeye Triplefin	3%	1.1
<b>Torpedinidae</b>	<b>Rays (Torpedo Electric)</b>			<b>Urolophidae</b>	<b>Rays (Round)</b>		
* <i>Narcine brasiliensis</i>	Lesser Electric Ray	0.1%	2.0	<i>Urolophus jamaicensis</i>	Yellow Stingray	6%	1.2

<sup>1</sup>Species Only, those seen during REEF surveys with no abundance estimates.



**Plate 6A.** “Recent mortality,” as in this *Montastraea annularis* lobe, is defined as any non-living parts of the coral in which the corallite structures are white and either still intact or covered by a thin layer of algae or fine mud. (Photo Kenneth W. Marks)



**Plate 6B.** Recent mortality resulting from parrotfish bites, most commonly observed in the *Montastraea annularis* species complex (as shown) and *Colpophyllia natans*, is characterized by partial loss of the skeleton along with the overlying living tissues. (Photo Robert S. Steneck)

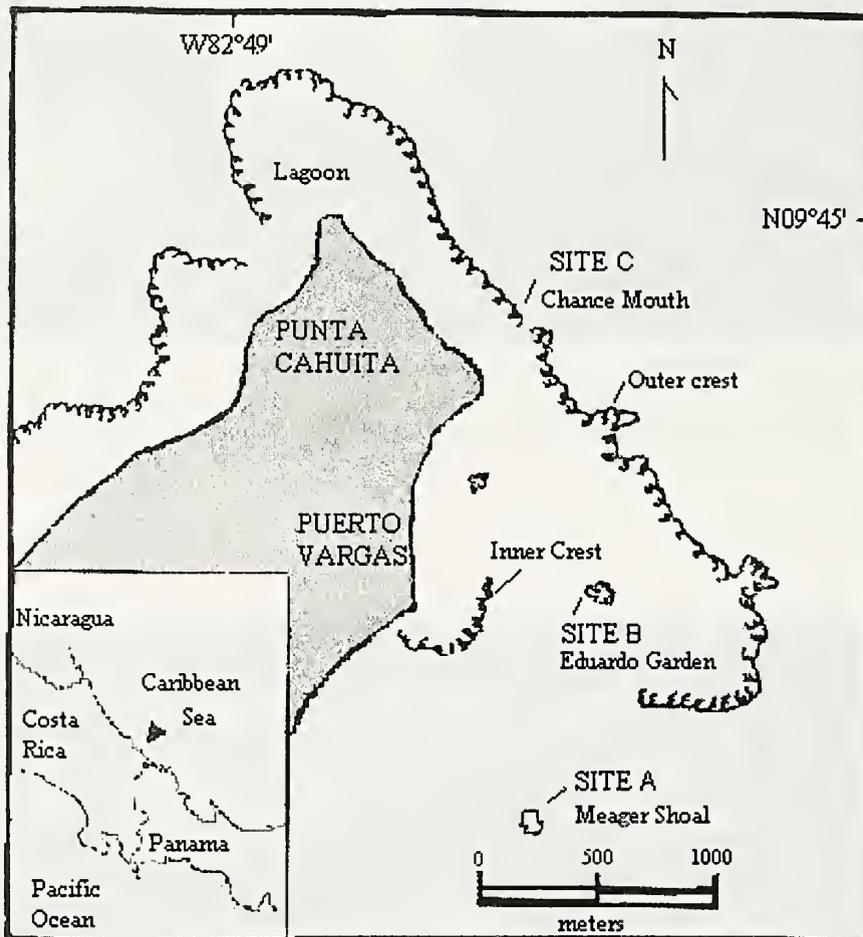


Figure 1. AGRRA survey sites in Cahuita, Costa Rica (modified from Cortés, 1998).

# A RAPID ASSESSMENT AT CAHUITA NATIONAL PARK, COSTA RICA, 1999 (PART 1: STONY CORALS AND ALGAE)

BY

ANA C. FONSECA E.<sup>1</sup>

## ABSTRACT

Live stony coral cover, which has deteriorated in Costa Rica's Cahuita National Park, was 2-3% in three reefs during October 1999. About eight percent of all "large" scleractinians ( $\geq 25$  cm in diameter) were diseased. The number of scleractinian species and the densities of recruits and *Diadema antillarum* were greatest in a carbonate hardground, where total (recent + old) partial-colony mortality of large scleractinians and macroalgal abundance were lowest. Colonies of *Acropora palmata* that were mostly dead were found only in a shallow patch reef where old partial-colony mortality was highest and where the largest corals were found. Recent partial-colony mortality was lowest in an outer fore-reef habitat. Recovery of deforested watersheds, the main source of the sediment stress in this and other reef systems on the Caribbean coast of Costa Rica, is urgently required.

## INTRODUCTION

Cahuita National Park, situated 35 km south of the city of Limón, has the largest fringing reef on the Caribbean coast of Costa Rica (Cortés, 1998). Its main outer reef, situated 1 km from the coast, arcs around the northern tip of Punta Cahuita, runs southeast about 5 km, and bends inshore towards Puerto Vargas (Fig. 1). A narrow spur-and-groove system reaches depths of 10 m on the outer fore reef. Several small patch reefs occur among meadows of seagrass (*Thalassia testudinum*, *Syringodium filiforme*), algae, and coral fragments in a 3 m-deep lagoon shoreward of the outer reef. Several shallow carbonate hardgrounds (7-10m) are located south of the main reef and offshore of Puerto Vargas beach. A small (500 m long) inner reef is located 100 m off the north end of the beach. Stony coral cover (measured by the chain transect method) averaged 40% across this inner reef in 1981 (Cortés and Risk, 1984).

Rainfall at Cahuita averages about 300 cm/year. The nearshore current, which flows from northwest to southeast, is strong, and wave energy is high, but the tidal amplitude is low (30-50 cm) (Cortés, 1998). Stony coral growth rates and diversity are low. As summarized by Cortés and Risk (1984), species inhabiting the Cahuita reefs are

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considered to be good sediment rejecters (e.g., *Siderastrea*, *Diploria*), or to have morphologies that inherently facilitate sediment removal (e.g., frondose *Agaricia agaricites*) and/or efficiently intercept light (e.g., foliaceous *Porites astreoides*).

Since 1970, 600 ha of the reef have been afforded partial protection by the creation of the Cahuita National Park (Cortés and Guzmán 1985). However, deforestation of the highlands and inappropriate agricultural practices, particularly on the coastal banana plantations, had already increased the input of noncarbonate (terrigenous) sediment and other pollutants in runoff, especially from Rio La Estrella. Physical damage to the reefs from visitors (trampling, diving, and anchoring) has also increased (Cortés, 1994). Both local inhabitants (who are allowed to fish with hook-and-line) and tourists walk through the shallow lagoon causing further disturbance to shallow habitats.

Other stressors have included the 40% mortality of stony corals, particularly *Acropora palmata* and *A. cervicornis*, during the 1983 El Niño-Southern Oscillation (ENSO) event and the near-demise of *Diadema antillarum* in 1983 and again in 1992. Shallow-water colonies of *Agaricia agaricites* and *Porites porites* were severely impacted by the 50 cm uplift of the coastline during the 1991 Limón earthquake (Cortés, 1994). By 1993 stony coral cover had fallen to 11% in 5 m at the Caribbean Coastal Marine Productivity (CARICOMP) monitoring site on the inner reef near Puerto Vargas (Cortés, 1994) where macroalgae were reported to be smothering stony corals and possibly affecting larval settlement. More recently, mild bleaching occurred in the reefs at Cahuita in 1998 (Garzón-Ferreira et al., 2000).

The purpose of this study was to characterize and update the condition of the coral and benthic algal communities in Cahuita National Park in October 1999 at the end of the millennium using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocol. Similar data were obtained the following year south of Cahuita close to the Panamanian border in the fringing fore reef off Manzanillo, where live stony coral cover was 1.9% in 1988 (Cortés, 1992). Results of the companion fish censuses are given in Fonseca and Gamboa (this volume).

## METHODS

Three sites (Fig. 1) that are each considered representative of a habitat type in Cahuita National Park were strategically selected for survey by two divers.

- Meager Shoal is a circular 10,000 m<sup>2</sup> hardground located south of the southern end of the outer reef. Seaward of Puerto Vargas beach, at an average depth of 7.0 m and about 1 km from the coast; it was chosen for its high coral diversity. The sediment that surrounds this bank is predominantly terrigenous muds and easily resuspended by currents.
- Eduardo Garden, a shallow (2 m) patch reef of 6,060 m<sup>2</sup> in the lagoon between the outer and inner reefs, is about 200 m from the coast. Constructed of largely dead colonies of *Acropora palmata*, it is known to be in better condition than other lagoonal sites, and local guides take tourists snorkeling here. Eduardo Garden was the

“outer patch” reef in Cortés and Risk’s (1985) investigation of siltation stress at Cahuita.

- Chance Mouth is a shallow (~5.5 m) spur-and-groove formation in the area of maximum topographic relief in the outer fringing reef. Located about 1 km from the coast, Chance Mouth was chosen because it is adjacent to the mouth of the only natural boat channel crossing the outer reef crest. The shallow fore reef that was surveyed at similar depths (4 m) off Manzanillo is a low-relief platform.

The following modifications were made to the AGRRRA Version 2.2 benthos protocol (see Appendix One, this volume). The maximum diameter of every stony coral covered by the transect line was recorded. Colonies of *Siderastrea* were only identified to genus due to difficulties with species identification in the western Caribbean. *Millepora* was not included in the surveys of individual corals. Coral sizes were measured to the nearest cm. Sediment was brushed gently off the substratum before estimating the abundance of crustose coralline algae. The abundance of sponges and other invertebrates in the algal quadrats was recorded. We used Humann (1993) for a field guide.

Species diversity ( $H'$ ), equity ( $J$ ) and Morisita similarity ( $M_i$ ) indices, based on maximum colony diameters, were estimated for all scleractinians under the transect lines. Coral diversity indices were compared using a Student t-test.

## RESULTS

### Stony Corals

The dominant “large” scleractinians ( $\geq 25$  cm in diameter) in the transects were *Siderastrea*, *Porites astreoides* and *Montastraea faveolata* in Meager Shoal, *Acropora palmata*, *Siderastrea* and *Diploria strigosa* in Eduardo Garden, and *Agaricia agaricites* and *Siderastrea* in Chance Mouth. Only *Siderastrea* was relatively abundant in all three sites. Species diversity differed significantly among sites ( $p < 0.0001$ ), being highest in Meager Shoal, which had twice as many large scleractinian taxa (eight versus four, respectively) as were present in each of the other two reefs (Table 1). The equity index was slightly higher in Eduardo Garden. The similarity of scleractinian taxa was somewhat greater between Meager Shoal and both Eduardo Garden ( $M_i = 0.48$ ) and Chance Mouth ( $M_i = 0.41$ ) than between Eduardo Garden and Chance Mouth ( $M_i = 0.19$ ).

Although the average density of large scleractinian corals in all three reefs was approximately 0.5 colonies/m (Table 1), the live stony coral cover (scleractinians and *Millepora*) was only 2-3%. The mean size (as maximum diameter and height) of the large corals was substantially greater in Eduardo Garden, where *A. palmata* and *Siderastrea* were especially large (Tables 2, 3), than in the other two reefs.

Recent partial-colony mortality (hereafter recent mortality), which averaged  $< 10\%$  in most surveyed scleractinians, was highest (mean=3%) at Meager Shoal. Estimates of old partial-colony mortality (hereafter old mortality) were more variable (~10-33%) with 34-78% of the colonies in Eduardo Garden and Meager Shoal, respectively, having less than 10% old mortality. The highest live:dead ratio (mean=7) was found in Meager Shoal.

Bleaching was only seen in Meager Shoal, where 2% of the large scleractinians were affected (Table 2). The percentage of large colonies affected by disease varied from 7-9%. White-plague disease affected proportionately more colonies of *Siderastrea*, *P. astreoides*, *D. strigosa* and *A. agaricites* in Chance Mouth and Meager Shoal than in Eduardo Garden. Black-band disease, found only in Chance Mouth, had infected one colony of *Siderastrea*. One colony of *Diploria strigosa* that appeared to have red-band disease was recorded in Eduardo Garden. The margins of less than five percent of the live scleractinians were being conspicuously overgrown by sediment-trapping turf algae in all reefs, or by macroalgae in Eduardo Garden and Chance Mouth.

Recruitment by scleractinians, primarily *Siderastrea* (12/18 recruits), was highest in Meager Shoal (Table 4). Recruits were rare in the other two reefs: one *Siderastrea* recruit was seen in Eduardo Garden, whereas two each *Agaricia agaricites* and *Dichocoenia stokesii* had recruited in Chance Mouth.

The following year at 4 m in the Manzanillo fore reef, where the mean live stony coral cover was 1.5%, large scleractinians averaged 32 cm in diameter (Tables 1, 2). Fifteen percent (7/47) of the surveyed corals were scored as diseased, including two colonies of *Siderastrea* with circular lesions ("white spots") of unknown origin. Only one recruit of *Porites astreoides* was found during the entire survey.

#### Algae and *Diadema antillarum*

Turf algae (and the cyanobacterium *Schizothrix*) constituted the predominant algal functional group in Meager Shoal (mean relative abundance = 67%) and were somewhat more abundant than crustose coralline algae in Eduardo Garden and Chance Mouth (Table 4). The relative abundance of macroalgae, of which the most common were *Dictyopteris*, *Sargassum* and *Halimeda*, varied from 14-25%. Macroalgal height and macroalgal indices (relative abundance of macroalgae x macroalgal height) were both substantially higher in Chance Mouth (means = 5.5 cm and 134, respectively) than elsewhere. Sponges and other invertebrates were less abundant in Eduardo Garden than in Chance Mouth and Meager Shoal. Tall macroalgae (mean=7.5 cm) predominated at Manzanillo in 2000 where the relative abundance of turf algae and crustose corallines were each less than 25% (Table 2).

The mean density of *Diadema antillarum* at Cahuita in 1999 was considerably greater in Meager Shoal than in the other two reefs (~60 versus 1-7 individuals/100m<sup>2</sup>, respectively). *Diadema antillarum* densities in Manzanillo averaged 10 individuals/100m<sup>2</sup> in 2000.

## DISCUSSION

The disparity in live stony coral cover between the 1999 AGRRA transects (<3% in three reefs) and those in the inner reef at the 5 m Caribbean CARICOMP site, which have averaged 11% since 1993 (Cortés, 1994; Fonseca, 1999, CARICOMP unpublished report), is due partially to natural differences among the reefs and partially to differing methodologies. AGRRA data are based on projected length under a line and

underestimate cover relative to values obtained by counting chain links draped over the contours of the substratum.

Live stony coral cover off Cahuita was lowest in Meager Shoal, where large ( $\geq 25$  cm in diameter) scleractinians were the most diverse, smallest in size, and had low partial-colony mortality values (Tables 1, 2, 3) and primarily laminar or crustose morphologies. Ambient illumination on the substratum is low: the muddy sediment surrounding the hardground is easily resuspended by currents and it is slightly deeper (7 m versus 5 m, respectively) than the mean annual vertical Secchi depth in this location (Fonseca, 1999, CARICOMP unpublished report). The bleached scleractinians found here in 1999 probably represented colonies that, due to the chronically low-light levels, had not yet recovered from the mild bleaching event the previous year. The slightly higher values of live coral cover (Table 1) as well as the more robust morphologies of the dominant scleractinians in Eduardo Garden and Chance Mouth may be related to their shallower and more highly illuminated locations. Eduardo Garden and Chance Mouth are surrounded by sandy, autochthonous sediment which presumably also contains fewer pollutants (see Cortés, 1994) than the riverine muds that are more likely to accumulate in Meager Shoal.

The high percentage of colonies with signs of active disease (7-9% at Cahuita in 1999; 15% at Manzanillo in 2000), coupled with the low cover of live stony corals in these Caribbean reefs, is of great concern. Estimates of recent partial mortality for large scleractinians were low in both years (0.5-3%). However, as exposed skeletal surfaces are rapidly covered with sediment and/or overgrown by algae (personal observations), snapshot assessments of recent mortality, as employed in the AGRRA benthos protocol, are likely to underestimate the magnitude of recent soft tissue loss in Costa Rica's coastal scleractinians. Old (and total) partial-mortality values were also fairly low (<20%) in all but Eduardo Garden, which had a higher proportion of large scleractinians (*Acropora palmata* and *Diploria strigosa*) that were mostly or entirely dead. Given its shallow location in the lagoon, some of these corals may have died during the 1983 ENSO event; alternatively, the *A. palmata* may have been infected by white-band disease, as seen elsewhere throughout the Caribbean (Aronson and Precht, 2001).

Densities of *Diadema antillarum* are still much lower than were found in Cahuita during the late 1970's and early 1980's (Cortés, 1994). Nevertheless, individuals were reasonably common in Meager Shoal (Table 4), where several *Diadema* were seen spawning and where they appeared to exert more control on benthic macroalgae than in the two other surveyed reefs. However, it is also likely that benthic algal productivity in this hardground is limited by the dim ambient illumination. Crustose coralline algae and macroalgae were relatively more abundant in Eduardo Garden and Chance Mouth, where *Diadema* was comparatively rare (Table 4). Less fishing occurs in these two reefs, yet their larger populations of herbivorous fishes (Fonseca and Gamboa, this volume) appear less effective at consuming macroalgae (Hughes et al., 1987; Hughes, 1994) than the *Diadema* in Meager Shoal. In Chance Mouth, where the higher surge and wave action may inhibit feeding by herbivorous fishes (Hay, 1981), the macroalgae were taller and the macroalgal index (a proxy for biomass) was correspondingly higher. *Diadema* were also scarce and reef fishes were far less abundant in Manzanillo in 2000 than in Chance Mouth in 1999 (Fonseca and Gamboa, this volume). Macroalgae were more abundant and

larger here than had been found the previous year in Cahuita. Hence coral reefs that are currently in poor condition are not restricted to Cahuita National Park.

Between 1993 and 1999, suspended particulate matter increased from 9 mg/l to about 20 mg/l at monitoring sites in the Cahuita National Park, including Meager Shoal and Eduardo Garden (Cortés, 1994; Fonseca, 1999, CARICOMP unpublished report), which doubtless is contributing to the continued low stony coral cover and low diversity of this reef system. Watershed owners must be encouraged to conscientiously and consistently follow the national regulation which requires a 50 m wide forested buffer along the rivers to ensure good water quality and improve conditions in coastal aquatic ecosystems. Moreover, as Cortés and Murillo (1985) have noted, "Coherent units of reefs and terrestrial environs must be considered when establishing a marine park or reserve. Watershed of rivers near the reef must be included in the protected area."

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Table 1. Site information for AGRRA stony coral and algal surveys in Cahuita and Manzanillo, Costa Rica.

Site name	Site code	Reef type	Latitude (° " N)	Longitude (° " W)	Survey date	Depth (m)	Benthic transects (#)	≥25 cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)	S <sup>1</sup>	N <sup>2</sup>	H <sup>3</sup>	J <sup>4</sup>
<b>Cahuita</b>													
Meager Shoal	A	Carbonate bank	9 43 50	82 48 32	Oct 7 1999	7.0	10	5	2.0 ± 1.0	8	2052	1.9	0.6
Eduardo Garden	B	Lagoonal patch reef	9 44 30	82 48 30	Oct 22 1999	2.0	10	4.5	3.0 ± 1.0	4	3730	1.6	0.8
Chance Mouth	C	Fore-reef spur & groove	9 45 00	82 48 35	Oct 22 1999	5.5	10	4.5	3.0 ± 1.0	4	2823	1.4	0.7
<b>Manzanillo</b>													
	D	Fore-reef platform	9 38 22	82 39 13	Oct 4 2000, Oct 11 2000	4	10	4.7	1.5 ± 0.5	-	-	-	-

<sup>1</sup>S = species number<sup>2</sup>N = sum of maximum diameters<sup>3</sup>H' = linear species diversity index, based on N<sup>4</sup>J = species equity index

Table 2. Size and condition (mean ± standard deviation) of all stony corals (≥25 cm diameter), by sites in Costa Rica.

Site name	Stony corals			Partial-colony mortality (%)			Stony corals (%)			
	#	Diameter (cm)	Height (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased	Other <sup>1</sup>
<b>Cahuita</b>										
Meager Shoal	49	44.5 ± 28.0	17.5 ± 27.0	3.0 ± 9.5	9.5 ± 15.0	12.5 ± 24.5	7.0	0	2.0	8
Eduardo Garden	44	86.5 ± 58.5	53.5 ± 37.0	1.5 ± 6.4	33.0 ± 31.2	34.5 ± 37.6	1.9	7.0	0	7
Chance Mouth	45	62.5 ± 72.5	19.0 ± 19.5	0.5 ± 1.5	20.0 ± 21.0	21.0 ± 22.5	3.8	0	0	9
<b>Manzanillo</b>										
		31.5 ± 30.0	10.1 ± 11.0	3.0 ± 7.5	9.5 ± 15.5	13.0 ± 22.8	6.8	0	0	15

<sup>1</sup>Other = demosponges and other invertebrates

Table 3. Diameter (mean  $\pm$  standard deviation) of coral species by site in Cahuita, Costa Rica.

Site name	<i>Siderastrea</i>	<i>Porites astreoides</i>	<i>Porites porites</i>	<i>Montastraea faveolata</i>	<i>Dichocoenia stokesii</i>	<i>Acropora palmata</i>	<i>Diploria clivosa</i>	<i>Diploria strigosa</i>	<i>Mycetophyllia</i>	<i>Agaricia agaricites</i>
Meager Shoal	54.5 $\pm$ 37.0 n=21	30.0 $\pm$ 6.0 n=16	30.0 n=1	55.5 $\pm$ 20.0 n=6	30.0 n=1	0	60.0 n=1	0	37.0 n=1	35.0 n=2
Eduardo Garden	85.5 $\pm$ 45.0 n=14	27.5 n=2	0	0	0	103.0 $\pm$ 77.5 n=16	0	75.0 $\pm$ 40.0 n=12	0	0
Chance Mouth	70.5 $\pm$ 37.0 n=10	36.0 $\pm$ 16.5 n=4	0	0	0	0	42.5 $\pm$ 15.5 n=4	0	0	69.0 $\pm$ 90.5 n=27

Table 4. Algal characteristics and density of stony coral recruits and *Diadema antillarum* (mean  $\pm$  standard deviation), by sites in Costa Rica.

Site name	Quadrat (#)	Relative abundance (%)		Macroalgal		Recruits (#/0.0625 m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )	
		Macroalgae	Turf algae <sup>1</sup>	Crustose coralline algae	Height			Index <sup>2</sup>
<b>Cahuita</b>								
Meager Shoal	45	14.0 $\pm$ 21.0	67.0 $\pm$ 21.5	18.5 $\pm$ 18.0	3.5 $\pm$ 3.0	48	0.4 $\pm$ 0.6	61 $\pm$ 32
Eduardo Garden	50	25.0 $\pm$ 20.5	43.0 $\pm$ 22.5	32.0 $\pm$ 24.5	3.0 $\pm$ 1.5	78	0.0 $\pm$ 0.1	1 $\pm$ 3
Chance Mouth	50	25.0 $\pm$ 22.0	39.5 $\pm$ 16.5	35.5 $\pm$ 21.5	5.5 $\pm$ 4.5	134	0.1 $\pm$ 0.3	7 $\pm$ 7
<b>Manzanillo</b>	45	56.0 $\pm$ 28.0	23.0 $\pm$ 16.0	21.5 $\pm$ 18.0	7.5 $\pm$ 4.0	416	0.02 $\pm$ 0.16	10 $\pm$ 22

<sup>1</sup>Includes the cyanobacterium *Schizothrix*.<sup>2</sup>Macroalgal index = % relative abundance of macroalgae x canopy height.

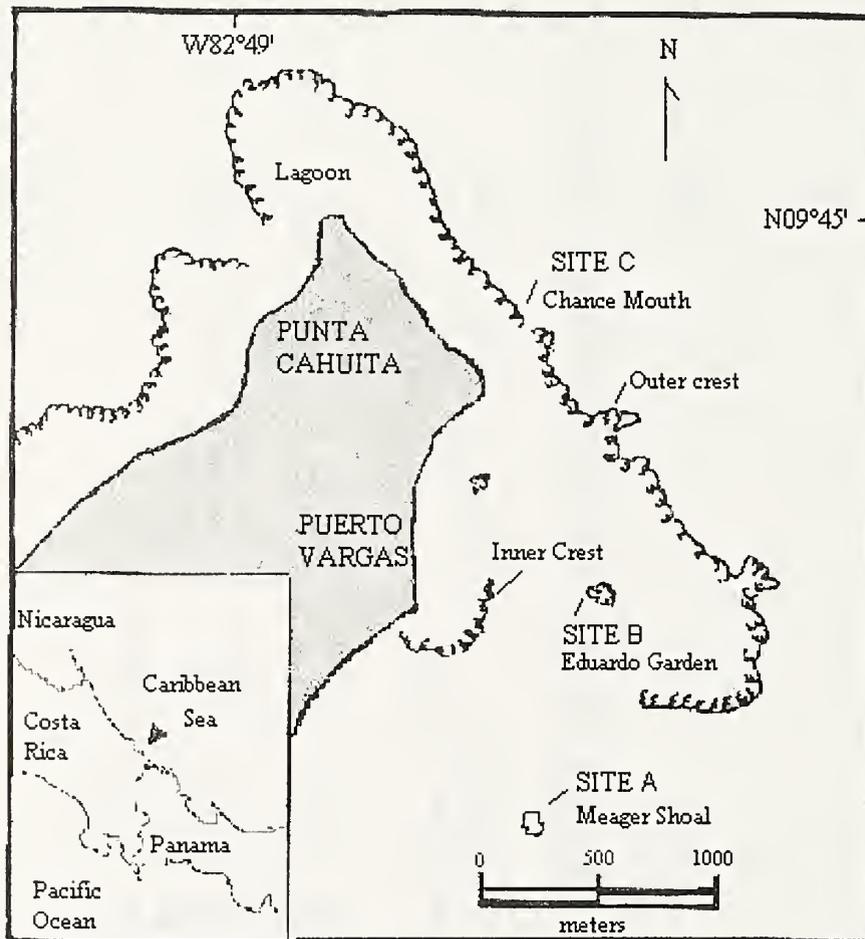


Figure 1. AGRRA survey sites in Cahuita, Costa Rica (modified from Cortés, 1998).

# A RAPID ASSESSMENT AT CAHUITA NATIONAL PARK, COSTA RICA, 1999 (PART 2: REEF FISHES)

BY

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## ABSTRACT

In Costa Rica's Cahuita National Park, fish density, diversity and size were greater in two shallow (<6 m) reefs, where topographic complexity and macroalgal abundance and height were higher and where density of *Diadema antillarum* and fishing pressures were lower than in a hardground at 7 m. Apparently there has been a slight improvement in the status of the lagoonal fish community since the implementation of fishing restrictions in 1978.

## INTRODUCTION

The fringing reef at Cahuita, 35 km south of the city of Limón, is the largest along Costa Rica's Caribbean coast. Nearshore currents in the area flow from northwest to southeast (Cortés, 1998). The outer reef (1 km from the coast) arcs around the northern tip of Punta Cahuita, runs southeast about 5 km, and bends inshore towards Puerto Vargas (Fig. 1). Several small patch reefs occur in the lagoon. There is also a small, 500-m long inner reef (100 m from the coast) in front of Puerto Vargas Beach and some carbonate hardgrounds occur to the south of the main reef.

Cahuita is a relatively protected national park; however, its coral reefs are not exempt from siltation stress arising in unprotected and deforested watersheds nearby. Deforestation, mainly for banana plantations, and consequent damage to reef organisms has greatly increased during the last 30 years (Cortés, 1994). The Caribbean coastal lowlands are very humid (annual rainfall near Cahuita is about 300 cm/year) leading to erosion of poorly covered soils. Currents are strong and wave energy is high (Cortés, 1998). Hence bottom sediments are easily resuspended, and suspended sediment levels are high (Cortés, 1994; Fonseca, 1999, Caribbean Coastal Marine Productivity unpublished report).

There have been few studies of fishes in Costa Rica's Caribbean coral reefs (e.g., Perry and Perry, 1974). In Cahuita, Phillips and Pérez-Cruet (1984) found that reefs with

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similar habitat complexity showed the highest fish composition similarity values, and an increase in fish size was observed as habitat complexity increased. Fish diversity currently is known to be low in Cahuita relative to other Caribbean reef areas, possibly due in part to the poor condition of its coral reefs. Fishing pressure overall is comparatively low because only hook-and-line fishing has been allowed inside the park since 1978.

Our purpose was to characterize the fish communities in Cahuita National Park at the end of the millennium using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocol. Similar data were obtained a year later south of Cahuita in the fringing reef off Manzanillo, close to the Panamanian border. The results of the benthos surveys are described by Fonseca (this volume).

## METHODS

Three reefs (Fig. 1, Table 1) that are each considered representative of a habitat type in Cahuita National Park were strategically chosen for survey by an experienced observer (Gamboa) in October 1999.

- Meager Shoal, a circular, 7 m-deep hardground (10,000 m<sup>2</sup>) south of the southern end of the outer reef, seaward of Puerto Vargas beach, and about 1 km from the coast, was chosen for its higher coral diversity. The sediment that surrounds this bank is muddy and easily resuspended by currents.
- Eduardo Garden, a shallow lagoonal patch reef (6,060 m<sup>2</sup>) between the outer and inner reefs and about 500 m from the coast, was chosen because it is in better condition than other sites in the lagoon. Local guides take tourists snorkeling here, and its fish populations were surveyed in 1982 by Phillips and Pérez-Cruet (1984).
- Chance Mouth, a shallow, spur-and-groove fore reef at an average depth of 5.5 m and about 500 m from the coast, was chosen because it is adjacent to the mouth of the only natural boat channel crossing the outer reef. Off Manzanillo, the shallow fore reef at similar depths (4 m) is a low-relief platform.

The AGRRA Version 2.2 protocol (see Appendix One, this volume) was modified to include all relatively abundant damselfish (*Abudefduf*, *Microspathodon*, *Stegastes*) in the belt transects. The fish counts were all made between 10 a.m. and 4 p.m. Consistency training was undertaken with a graduated T-bar prior to beginning the surveys.

Fish diversity ( $H'$ ), equity ( $J$ ) and Morisita similarity ( $M_i$ ) indices, based on the entire dataset, were also estimated for each reef. Differences in fish diversity indices were tested with a Student t-test.

## RESULTS

The number of species (AGRRA list/total) in the belt transects varied between 15/18 in Meager Shoal and 31/34 in Eduardo Garden (Table 1, Appendix A, this paper). Eduardo Garden also had the highest values for the diversity and equity indices, whereas

the lowest of both were found in Meager Shoal. Diversity indices varied significantly among sites (t-test,  $p < 0.0001$ ). The fish faunas in Eduardo Garden and Chance Mouth were the most similar in composition ( $M_i = 0.90$ ), those in Chance Mouth and Meager Shoal were somewhat less similar ( $M_i = 0.69$ ), while the least similar were the faunas of Eduardo Garden and Meager Shoal ( $M_i = 0.54$ ). Damsel-fishes (mainly *Stegastes fuscus* and, except in Meager Shoal, *Abudefduf saxatilis*) were numerical dominants in all three sites. At Manzanillo in 2000 the number of AGRRA/total species were 13/14, of which the dominant were grunts (haemulids) and surgeonfishes (acanthurids; Tables 1 and 2).

In 1999, AGRRA-listed fishes were over four times more abundant in Eduardo Garden than in Meager Shoal and about a third more plentiful in Meager Shoal than in Chance Mouth (Table 1). The two most common fishes in Meager Shoal were the parrotfish (scarid), *Sparisoma viride*, and the damselfish, *Microspathodon chrysurus* (Appendix A, this paper). In addition, three of the four most abundant AGRRA fishes in Eduardo Garden (two parrotfishes, *Scarus croicensis* and *Sparisoma rubipinne*, and one surgeonfish, *Acanthurus bahianus*) and five of the seven AGRRA fishes that were most common in Chance Mouth (*S. rubipinne*, *M. chrysurus*, and the surgeonfishes, *Acanthurus coeruleus*, *A. bahianus* and *A. chirurgus*) were herbivores.

The snapper (lutjanid), *Lutjanus apodus*, and grunts, particularly *Haemulon aurolineatum*, were also conspicuous in Eduardo Garden, while a grunt (*H. macrostomum*) and a jack (carangid), *Caranx ruber*, were fairly common in Chance Mouth. Three species of small-sized groupers (serranids) were seen in Eduardo Garden, two of which were also present in Meager Shoal (Appendix A, this paper). The overall densities of parrotfishes, snappers and grunts were highest in Eduardo Garden, while surgeonfishes were most abundant in Chance Mouth. Groupers, particularly *Epinephelus cruentatus*, were most common in Meager Shoal. The average density of the AGRRA-listed species off Manzanillo in 2000 was about half of that which had been found in Chance Mouth the previous year (Table 1). The most common species in Manzanillo were the surgeonfish, *A. coeruleus* ( $\sim 2 \pm 3$  individuals/100 m<sup>2</sup>), the grunts, *Haemulon carbonarium* and *H. parra* ( $\sim 1.5 \pm 5$  and  $\sim 2 \pm 6$  individuals/100 m<sup>2</sup>, respectively), and the parrotfish, *S. viride* ( $\sim 1.5 \pm 1.5$  individuals/100 m<sup>2</sup>).

The mean lengths of surgeonfishes in all three of the Cahuita reefs, and of parrotfishes, groupers and snappers in Meager Shoal, were each less than 20 cm (Table 3). Grunts in all reefs, as well as parrotfishes and snappers in Eduardo Garden and Chance Mouth, each had mean lengths in the 20-30 cm interval. Overall, however, many individual fishes were in the 11-20 cm size range. In Manzanillo all the surveyed fishes were less than 20 cm in length, except for snappers which averaged 30 cm, and a single grouper, *Mycteroperca bonaci*, that was 50 cm long.

## DISCUSSION

The density, size and diversity of the AGRRA fishes at Cahuita were higher in the two shallower reefs than in the deeper hardground (Tables 1-3; Appendix A, this paper). Relative to Meager Shoal, these reefs also had larger-sized corals, somewhat more macroalgae and crustose coralline algae (Fonseca, this volume), and fishing pressures

were lower (personal observations). Topographic complexity was also higher in Chance Mouth, due to its spur-and-groove topography, and in Eduardo Garden with its large colonies of mostly-dead *Acropora palmata* than in Meager Shoal. The macroalgal index (macroalgal relative abundance x macroalgal height, a proxy for macroalgal biomass) was highest in Chance Mouth (Table 2), where herbivore feeding may be inhibited by the comparatively high surge and wave action (Hay, 1981). Herbivorous fishes in these two reefs apparently are less capable of removing macroalgae from the substrata than the *Diadema antillarum* that currently are more common in Meager Shoal.

Meager Shoal, with the highest relative abundance of turf algae (Fonseca, this volume), also had the fewest and smallest fishes (excepting groupers). Species targeted by hook-and-line fishers are primarily grunts and snappers (Mug, unpublished report) and one of the most popular, *Haemulon macrostomum*, was extremely rare in this hardground (yet it was the only grunt present in the belt transects; Appendix A, this paper). The fishing pressure in Manzanillo is greater still since it is not a national park. Its fore-reef platform also has lower topographic relief than the spurs and grooves off Cahuita, and macroalgae were about twice as abundant here in 2000 as had been found the previous year at similar depths in Chance Mouth.

Damselfishes and wrasses (labrids) dominated Phillips and Pérez-Cruet's (1984) surveys of Caribbean fishes. Damselfishes are still very abundant in Cahuita (Appendix A, this paper). In Eduardo Garden (their transect 7), Phillips and Pérez-Cruet (1984) found a mean density of 113.3 individuals/100 m<sup>2</sup> and a diversity index of 2.71 ( $J = 0.80$ ;  $S = 30$ ). Our values for fish density and diversity (including all damselfishes but lacking wrasses) were both higher in Eduardo Garden in 1999 (Appendix A, this paper).

Presumably the protection against most forms of fishing in Cahuita National Park is allowing some recovery of its reef fish populations. Nevertheless, fish diversity in the park is low relative to other areas of the Caribbean (Perry and Perry, 1974; Phillips and Pérez-Cruet, 1984) and probably related to the poor condition of its coral reefs (Fonseca, this volume). Hence stronger management actions must be implemented in the park, particularly in the southern hardgrounds.

## ACKNOWLEDGMENTS

We would like to thank the AGRRA Akumal Workshop presenters for training in the AGRRA protocols, the Centro de Investigación en Ciencias del Mar y Limnología (CIMAR) for providing us with field equipment, and Dr. Jorge Cortés for funding (from US-AID-CDR, Project TA MOU-97-C14015). The AGRRA Organizing Committee facilitated the senior author's participation in the Akumal and Miami workshops as described in the Forward to this volume.

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Table 1. Site information for AGRRA fish surveys in Cahuita and Manzanillo, Costa Rica.

Site name	Site code	Reef type	Latitude (° ' " N)	Longitude (° ' " W)	Survey date	Depth (m)	% live stony coral cover (mean ± sd) <sup>1</sup>	30 m fish transects (#)	AGRRA fishes		<i>Diadema</i> (#/10 m <sup>2</sup> ) <sup>1</sup>
									Density (#/100 m <sup>2</sup> ) (mean ± sd)	Species (#)	
<b>Cahuita</b>											
Meager Shoal	A	Carbonate bank	9 43 50	82 48 32	Oct 7 1999	7.0	2.0 ± 1.0	10	20.0 ± 13.08	15	6.1 ± 3.2
Eduardo Garden	B	Lagoonal patch reef	9 44 30	82 48 30	Oct 22 1999	2.0	3.0 ± 1.0	10	92.2 ± 66.13	31	0.1 ± 3
Chance Mouth	C	Fore-reef spur & groove	9 45 00	82 48 35	Oct 22 1999	5.5	3.0 ± 1.0	10	62.0 ± 50.62	17	0.7 ± 7
<b>Manzanillo</b>											
	D	Fore-reef platform	9 38 22	82 39 13	Oct 4 2000, Oct 11 200	4	1.5 ± 0.5	11	10.1 ± 8.37	13	1.0 ± 22

<sup>1</sup> Data from Fonseca (this volume).

Table 2. Density (mean ± standard deviation) of AGRRA fishes and macroalgal index, by sites in Costa Rica.

Site name	Herbivores (#/100 m <sup>2</sup> )		Carnivores (#/100 m <sup>2</sup> )		Chaetodontidae (#/100 m <sup>2</sup> )	Pomacanthidae (#/100 m <sup>2</sup> )	Macroalgal index <sup>2</sup>		
	Acanthuridae (≥5 cm)	Scaridae (≥5 cm)	Haemulidae (≥5 cm)	Lutjanidae (≥5 cm)				Serranidae	
<b>Cahuita</b>									
Meager Shoal	2.5 ± 2.5	6.5 ± 6.5	4.5 ± 5.0	0.2	2.5 ± 4.5	3.5 ± 2.0	0.8 ± 1.2	0	48
Eduardo Garden	20.0 ± 18.5	28.5 ± 17.5	5.0 ± 3.5	23.5 ± 25.0	11.0 ± 16.5	2.0 ± 4.0	2.0 ± 1.3	0	78
Chance Mouth	25.0 ± 27.0	13.5 ± 11.5	7.0 ± 7.0	10.0 ± 8.5	5.0 ± 4.5	0	0.8 ± 1.4	0.2	134
<b>Manzanillo</b>									
	3.2 ± 5.0	1.7 ± 1.5	0.3	3.3 ± 11.1	0.8 ± 2.0	0.2	0	0.4 ± 0.8	416

<sup>1</sup> *Epinephelus* spp. and *Mycteroperca* spp.<sup>2</sup> Macroalgal index = % relative abundance of macroalgae x canopy height; data from Fonseca (this volume).

Table 3. Total length in cm (mean  $\pm$  standard deviation) of AGRRA fishes, by sites in Costa Rica.

Site name	Herbivores			Haemulidae ( $\geq 5$ cm)	Carnivores		Chaetodontidae	Pomacanthidae
	Acanthuridae	Scaridae ( $\geq 5$ cm)	<i>Microspathodon chrysurus</i>		Lutjanidae	Serranidae <sup>2</sup>		
<b>Cahuita</b>								
Meager Shoal	18.5 $\pm$ 6.0 n=6	14.5 $\pm$ 5.0 n=10 <sup>1</sup>	11.0 $\pm$ 2.5 n=10	25.5 n=1	17.5 $\pm$ 4.5 n=5	18.0 $\pm$ 3.5 n=9	8.5 $\pm$ 5.5 n=4	0
Eduardo Garden	18.5 $\pm$ 6.0 n=6	20.5 $\pm$ 5.5 n=10	13.5 $\pm$ 3.0 n=10	24.0 $\pm$ 4.5 n=10	24.0 $\pm$ 7.5 n=9	22.0 $\pm$ 5.0 n=4	14.5 $\pm$ 2.5 n=8	0
Chance Mouth	17.0 $\pm$ 7.0 n=9	23.5 $\pm$ 8.0 n=10	12.0 $\pm$ 2.0 n=10	23.0 $\pm$ 6.0 n=9	20.0 $\pm$ 8.5 n=8	0	13.0 $\pm$ 4.5 n=3	25.5 n=1
<b>Manzanillo</b>	10.6 $\pm$ 12.6 n=21	13.7 $\pm$ 6.7 n=11	11.8 n=2	12.1 $\pm$ 1.41 n=22	30.5 $\pm$ 7.1 n=5	50.0 n=1	0	18.8 $\pm$ 5.8 n=3
<b>All Cahuita sites</b>	17.5 $\pm$ 6.0 n=23	19.5 $\pm$ 7.0 n=30	12.5 $\pm$ 2.5 n=30	23.7 $\pm$ 5.0 n=20	21.0 $\pm$ 7.5 n=22	19.0 $\pm$ 4.0 n=13	12.5 $\pm$ 4.5 n=15	25.5 n=1

<sup>1</sup>n = number of individuals<sup>2</sup>*Epinephelus* spp. and *Mycteroperca* spp.

Appendix A. Density (mean  $\pm$  standard deviation) and diversity indices for AGRRA fishes and other damselfishes, by site in Cahuita, Costa Rica.

Fish species	Density (#/100 m <sup>2</sup> )		
	Meager Shoal	Eduardo Garden	Chance Mouth
<i>Pomacanthus paru</i>	0.0 $\pm$ 0.0	0	0.2 $\pm$ 0.5
<i>Chaetodon capistratus</i>	0.2 $\pm$ 0.5	0.2 $\pm$ 0.5	0
<i>Chaetodon striatus</i>	0.0 $\pm$ 0.0	0.5 $\pm$ 0.8	0
<i>Chaetodon ocellatus</i>	0.7 $\pm$ 1.2	1.3 $\pm$ 1.7	0.8 $\pm$ 1.4
<i>Anisotremus virginicus</i>	0	1.0 $\pm$ 1.4	0.5 $\pm$ 0.8
<i>Anisotremus surinamensis</i>	0	0.2 $\pm$ 0.5	0
<i>Haemulon plumieri</i>	0	0.8 $\pm$ 1.6	0
<i>Haemulon sciurus</i>	0	1.2 $\pm$ 2.1	0
<i>Haemulon flavolineatum</i>	0	4.5 $\pm$ 3.5	0
<i>Haemulon aurolineatum</i>	0	7.5 $\pm$ 15.6	0
<i>Haemulon carbonarium</i>	0	4.8 $\pm$ 15.2	0
<i>Haemulon macrostomum</i> <sup>1</sup>	0.2 $\pm$ 0.5	3.7 $\pm$ 4.4	9.5 $\pm$ 8.4
<i>Sparisoma viride</i>	4.8 $\pm$ 5.6	2.8 $\pm$ 4.0	1.2 $\pm$ 1.4
<i>Sparisoma rubripinne</i>	1.2 $\pm$ 1.8	7.8 $\pm$ 3.7	7.8 $\pm$ 4.4
<i>Sparisoma aurofrenatum</i>	0.3 $\pm$ 0.7	1.7 $\pm$ 3.8	3.3 $\pm$ 10.5
<i>Scarus taeniopterus</i>	0	0.2 $\pm$ 0.5	0
<i>Scarus croicensis</i>	0	14.2 $\pm$ 12.2	1.2 $\pm$ 3.7
<i>Sparisoma chrysopterus</i>	0	1.5 $\pm$ 4.7	0
<i>Scarus coelestinus</i>	0	0.2 $\pm$ 0.5	0.2 $\pm$ 0.5
<i>Epinephelus guttatus</i>	0.3 $\pm$ 0.7	0.3 $\pm$ 0.7	0
<i>Epinephelus cruentatus</i>	3.0 $\pm$ 2.1	1.5 $\pm$ 3.6	0
<i>Epinephelus fulvus</i>	0	0.2 $\pm$ 0.5	0
<i>Lutjanus analis</i> <sup>1</sup>	0	0.2 $\pm$ 0.5	0
<i>Lutjanus apodus</i> <sup>1</sup>	0	7.8 $\pm$ 16.9	0
<i>Lutjanus griseus</i> <sup>1</sup>	0	0	0.2 $\pm$ 0.5
<i>Lutjanus mahogoni</i> <sup>1</sup>	2.2 $\pm$ 4.6	2.2 $\pm$ 3.9	3.7 $\pm$ 3.8
<i>Ocyurus chrysurus</i> <sup>1</sup>	0.2 $\pm$ 0.5	0.7 $\pm$ 0.9	1.3 $\pm$ 3.1
<i>Lutjanus jocu</i> <sup>1</sup>	0	0.3 $\pm$ 0.7	0
<i>Acanthurus bahianus</i>	1.7 $\pm$ 2.1	13.8 $\pm$ 20.0	8.0 $\pm$ 11.1
<i>Acanthurus chirurgus</i>	0.3 $\pm$ 1.1	0.8 $\pm$ 2.1	6.0 $\pm$ 12.5
<i>Acanthurus coeruleus</i>	0	5.2 $\pm$ 5.8	11.2 $\pm$ 19.4
<i>Bodianus rufus</i>	2.8 $\pm$ 4.8	4.0 $\pm$ 3.2	4.5 $\pm$ 8.1
<i>Microspathodon chrysurus</i>	4.3 $\pm$ 5.2	5.2 $\pm$ 3.5	7.0 $\pm$ 7.1
<i>Stegastes fuscus</i> <sup>2</sup>	31.5 $\pm$ 16.0	37.5 $\pm$ 56.4	71.3 $\pm$ 31.6
<i>Stegastes partitus</i> <sup>2</sup>	3.0 $\pm$ 5.1	0	0
<i>Abudefduf saxatilis</i> <sup>2</sup>	0.3 $\pm$ 1.1	47.2 $\pm$ 76.3	69.5 $\pm$ 108.1
<i>Caranx ruber</i>	0	2.0 $\pm$ 3.3	9.3 $\pm$ 12.6
<b>Total fish species (S)</b>	<b>18</b>	<b>34</b>	<b>20</b>
<b>Total fishes (N)</b>	<b>350</b>	<b>1096</b>	<b>1297</b>
<b>Total fish species diversity (H)<sup>1</sup></b>	<b>2.5</b>	<b>3.7</b>	<b>2.8</b>
<b>Total fish species equity (J)</b>	<b>0.6</b>	<b>0.7</b>	<b>0.6</b>

<sup>1</sup>Fishes targeted by line fishers (Mug 2000, unpublished report)<sup>2</sup>Not included in the calculations of AGRRA fish density in Table 1.



**Plate 7A.** “Old mortality,” as in this *Diploria strigosa*, is defined as any non-living parts of a stony coral in which the corallite structures have either been lost or are covered by organisms that are not easily removed (certain algae, invertebrates and colonial foraminifera). (Photo Kenneth W. Marks)



**Plate 7B.** Colonies that are completely, or 100% dead, known as “standing dead,” are also assessed as long as they can be identified to generic level based on colony morphology, as shown for this *Acropora palmata*, or by residual corallite characters (e.g., *Diploria* spp., *Montastraea cavernosa*). (Photo Robert S. Stenenck)

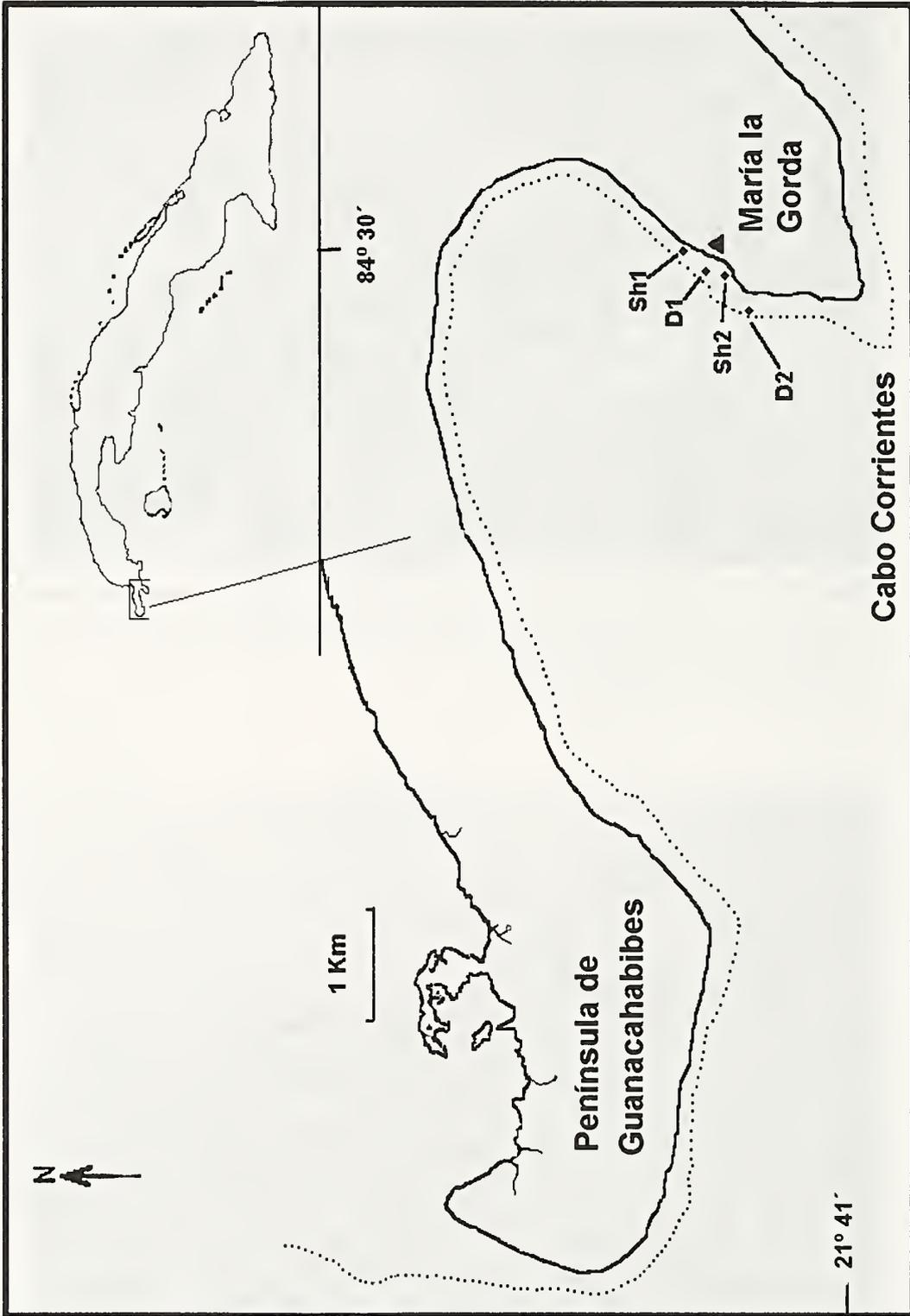


Figure 1. AGRRA survey sites in María la Gorda, Cuba. See Table 1 for site codes.

**RAPID ASSESSMENT OF CORAL COMMUNITIES OF MARÍA LA GORDA,  
SOUTHEAST ENSENADA DE CORRIENTES, CUBA  
(PART 1: STONY CORALS AND ALGAE)**

BY

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**ABSTRACT**

The Atlantic and Gulf Rapid Reef Assessment benthos protocols were utilized in four reefs off María la Gorda, western Cuba, in July 1999. Live stony coral cover ranged from 15.5-23.5%. *Montastraea annularis*, *M. franksi*, *Siderastrea siderea* and *M. faveolata* were the dominant species of stony corals. Large ( $\geq 25$  cm diameter) stony corals were fairly abundant (7.5-10.5/10 m), but in two reefs had incurred moderately high values of recent partial-colony mortality ( $\sim 10\%$ ). The major stressors on these stony corals were damselfish bites, damselfish algal gardens, diseases, and a relatively high abundance of fleshy macroalgae. Damselfish densities were probably elevated because their predators have been overharvested. Enforcement of existing fishing regulations and other management actions are necessary to preserve the ecological and touristic value of this reef system.

**INTRODUCTION**

The María la Gorda coral reef, considered one of the most beautiful, diverse and well-preserved of Cuban reefs, is located in a 1,216 km<sup>2</sup> Biosphere Reserve near the southeastern tip of Bahía (Ensenada) de Corrientes in the Península de Guanahacabibes (Fig. 1). The adjoining land is a karst plain covered by a naturally semideciduous, mesophytic forest that has been little disturbed although it is subjected to a restricted forestry. Annual rainfall averages 100-120 cm and mean air temperatures are 27-28° C in July and 20-22° C in January (IGEO-ICGC-ACC, 1989). Low rocky cliffs with coastal vegetation alternate with small sandy beaches along the coastline. No rivers empty into the bay.

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The only human settlement at María la Gorda is a small (29-room) dive resort with 50 associated dive sites. La Bajada, a tiny village located about 14 kilometers further north, has 22 houses, a frontier guard post, the Reserve's Visitors Center, and a meteorological radar station. Its human population totals less than 150 inhabitants.

The Guanahacabibes peninsula protects María la Gorda's leeward fringing reefs from the prevailing northeastern trade winds and from northerly cold fronts in winter. An irregular series of elongated low-relief (1-2 m) coral lobes (some forming spurs and grooves), each over ~100-200 m in length, about 50-100 m wide and oriented parallel to the shoreline, alternate with tongues of sand at depths of about 5 to 9-12 m on a gently sloping reef terrace. Small patch reefs border the lobes on their shallow and deep margins. However, the diving sites preferred by tourists are located in excess of 15 m on the reef slope where the reefs are aesthetically more beautiful, biologically more diverse, and have higher topographic relief.

Benthic reef condition was first assessed at María la Gorda within the framework of the *General assessment of the ecological state of the Cuban reefs and monitoring of the regional Cuban CARICOMP Station* and the Atlantic and Gulf Rapid Reef Assessment (AGRRA) initiative in July 1999.

## METHODS

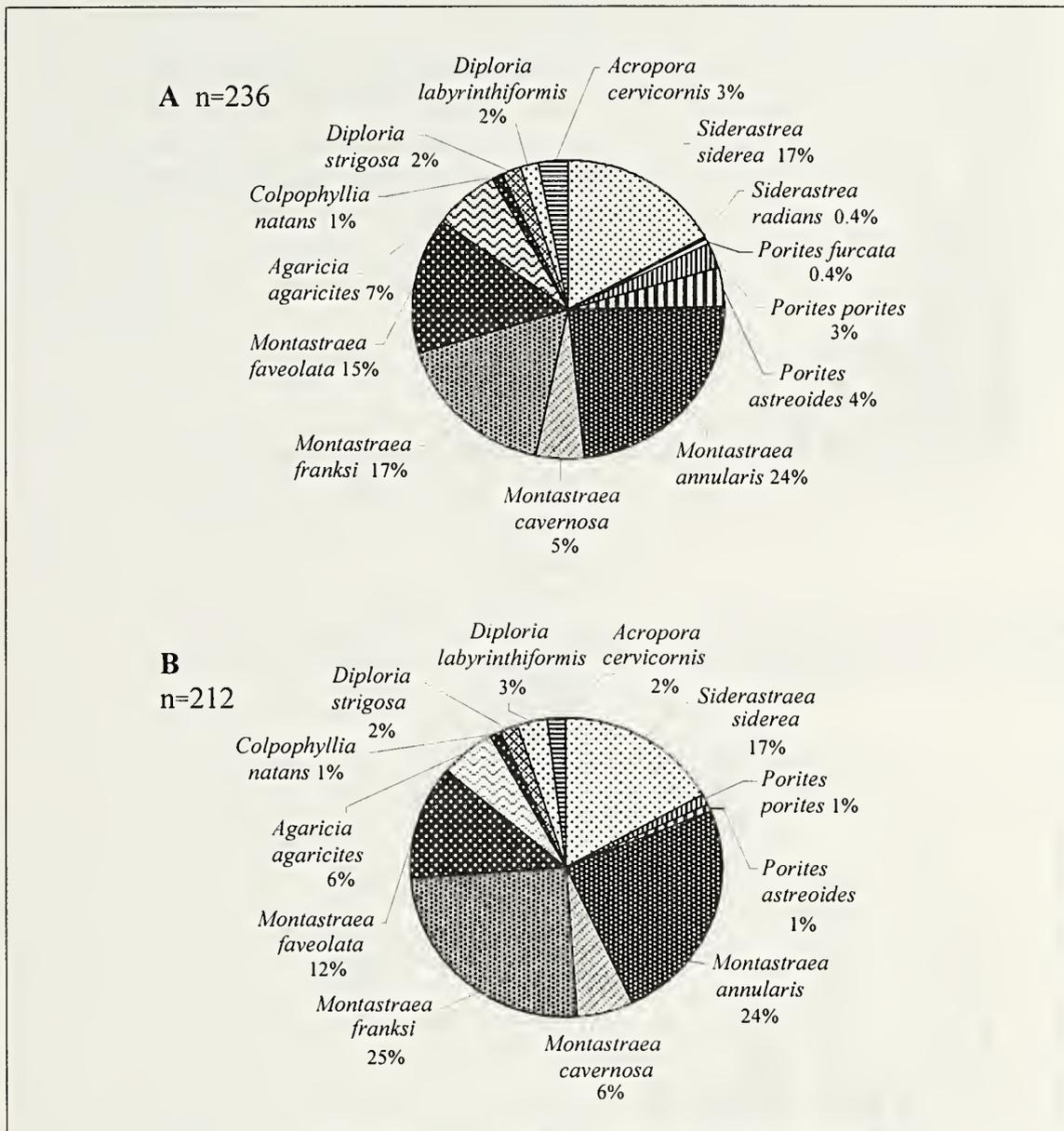
The benthic surveys were made by five divers in four reef lobes (one having well-developed spurs and grooves) that were considered representative of the reef terrace habitats in this strategically chosen tourist location. There were two "shallower" reefs in 5-6 m, and two "deeper" reefs that were located in 8 m and in 11 m, respectively.

Version 2.2 of the AGRRA benthos protocol (see Appendix One, this volume) was followed. We measured the sizes of "large" ( $\geq 25$  cm diameter) stony corals (scleractinians and *Millepora* spp.) to the nearest 10 cm. *Favia fragum* and other species that are small as adults were not included in the counts of coral recruits. Sediment was removed from the algal quadrats before estimating the abundance of crustose coralline algae. Practice and consistency training occurred in reefs near Havana before the surveys were initiated. Our guide for coral identifications in the field was Humann (1993).

## RESULTS

### Stony Corals

Total live stony coral cover was somewhat higher in the two shallower reefs (20.5%, 23.5%) than in the two deeper reefs (15.5%, 17%), as was the density of large ( $\geq 25$  cm diameter) corals (8.5 and 10.5/10 m versus 7.5/10 m, respectively; Table 1). In all four reefs, the large stony corals were dominated by *Montastraea annularis*, *M. franksi*, *Siderastrea siderea* and *M. faveolata* (Fig. 2A, B). The mean diameter of the large corals was about 50% greater in the two more southerly reefs (one each being shallower and deeper) than in the two more northerly reefs (60.5 and 63.5 cm versus 44



**Figure 2.** Species composition and mean relative abundance of the most abundant stony corals ( $\geq 25$  cm diameter) at (A) 5-8 m, (B) 9-13 m in María la Gorda, Cuba.

and 46.5 cm, respectively). Overall, *M. faveolata* formed the largest colonies (to  $>300$  cm in diameter) with no size class predominating, whereas most colonies of *M. franksi* and *M. annularis* were  $<70$  cm in diameter, while few *S. siderea* exceeded 50 cm in diameter (Fig. 3).

Large stony corals in the two southerly reefs also had somewhat higher percentages of each of the following relative to the northerly reefs: recent partial-colony mortality (9 and 10.5% versus 3 and 5%); old partial-colony mortality (28.5 and 34.5% versus 24.5 and 27%); “standing dead” corals having 100% mortality on their upper

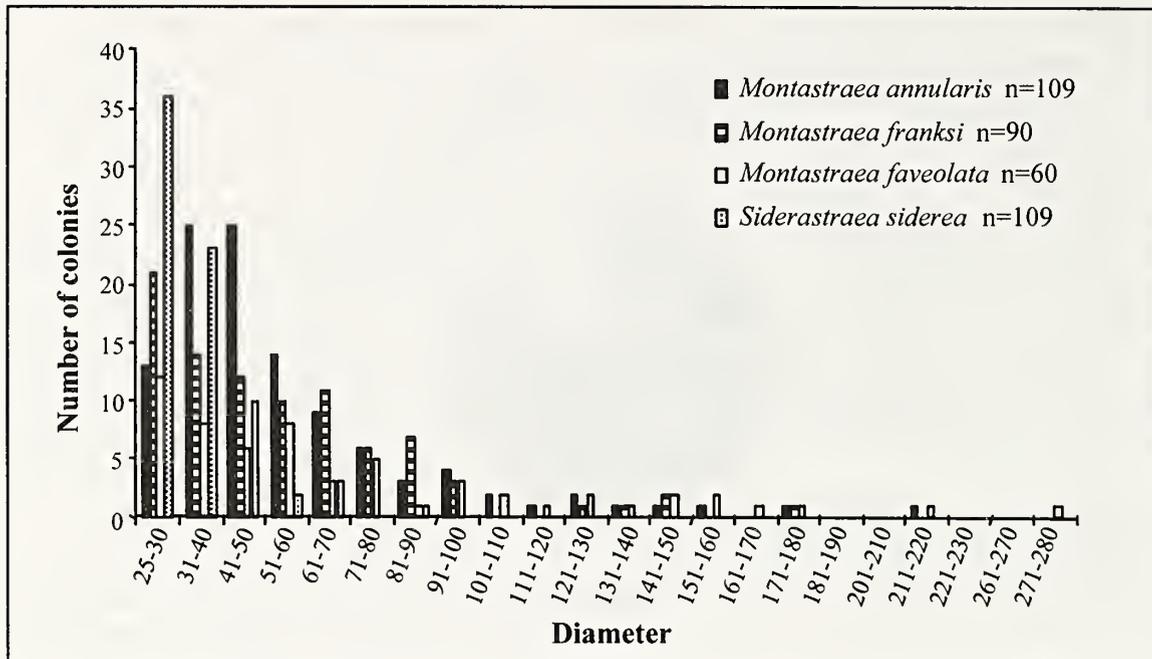


Figure 3. Size-frequency distribution in cm of colonies ( $\geq 25$  cm diameter) of *Montastraea annularis*, *M. franksi*, *M. faveolata* and *Siderastraea siderea* at 5-13 m in María la Gorda, Cuba.

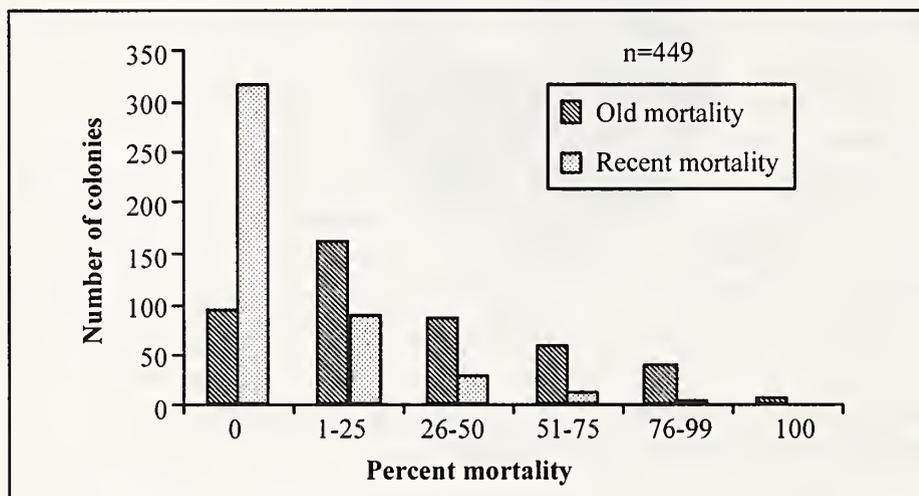
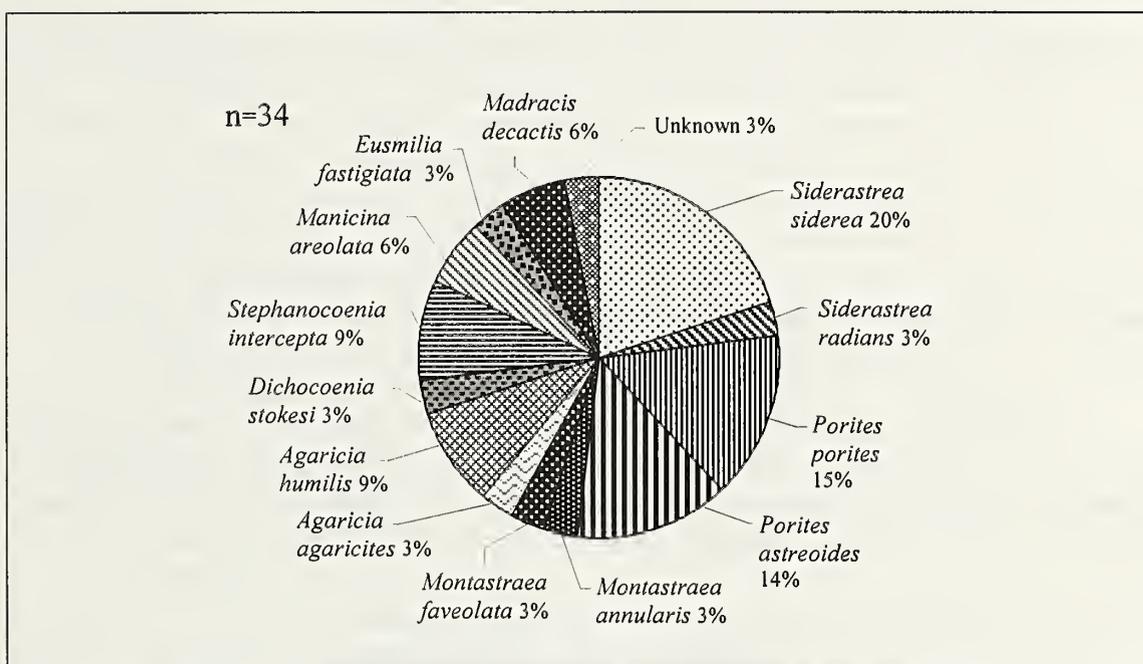


Figure 4. Number of colonies with old partial colony mortality and recent partial colony mortality for all stony corals  $\geq 25$  cm diameter at 5-13 m, in María la Gorda, Cuba.

colony surfaces and still in growth position (2 and 4.5% versus 1 and 1.5%); The percentage of stony corals that were bleached was also higher in the two southerly reefs (3.5 and 5.5% versus 1 and 2%, respectively; Table 2). Nevertheless, the majority of these colonies had no recently dead tissues (Fig. 4).

An average of 3-6.5% of the large stony corals in each reef showed signs of disease. Most commonly noted were white-plague disease in *M. annularis*, *M. faveolata* and *M. franksi*, black-band disease in *Diploria strigosa*, white-band disease in *Acropora cervicornis*, and dark spots disease in *S. siderea*. In addition, some unusually darkly pigmented tissues were seen in *Agaricia agaricites* (two colonies) and *Colpophyllia natans* (one colony). The species that were either pale or partially bleached were *S. siderea*, *M. annularis*, *M. franksi*, *M. faveolata* and *P. astreoides*. Colonies of *S. siderea*, *M. faveolata*, *M. franksi* and *M. annularis* were found with deep parrotfish bites, while damselfish and their algal gardens were particularly common in *D. strigosa* and *D. labyrinthiformis*.

The density of stony coral recruits varied from 0.1-0.2/0.625m<sup>2</sup> (Table 3) which is equivalent to 1.6-3.2 /m<sup>2</sup>. *Siderastrea siderea*, *Porites porites* and *P. astreoides* dominated this assemblage (Fig. 5).



**Figure 5.** Species composition and mean relative abundance of all stony coral recruits ( $\leq 2$  cm diameter) at 5-13 m, in María la Gorda, Cuba.

#### Algae and *Diadema antillarum*

In terms of relative abundance, macroalgae were the predominant algal functional group (46-60%) in all four reefs, with crustose coralline algae being the least abundant (13-18%; Table 3). Macroalgae were dominated by *Dictyota divaricata*, *Lobophora variegata* and *Sargassum hystrix* whereas green calcareous algae were scarce. Mean macroalgal heights were 2-2.5 cm, and macroalgal indices (relative abundance macroalgae x mean macroalgal height) ranged from 97 to 150. Densities of *Diadema antillarum* varied from zero to 4.3/100 m<sup>2</sup> (Table 3).

## DISCUSSION

María La Gorda is the first Cuban coral reef to have been assessed by the AGRRA protocols. The prevalence of the *Montastraea annularis* species complex in all four reefs is an indication that this reef is generally in good condition. The relatively limited abundance of the sediment-tolerant *Siderastrea siderea* (Fig. 2) is suggestive that sedimentation is a less severe stress than in many other Cuban reefs e.g., at Archipélagos Sabana-Camaguey, Cayo Diego Pérez, Cayo Cantiles and Cayo Juan García (Alcolado in prep.), Cayo Largo (Alcolado et al., 2001), and Ciudad Habana (Herrera-Morena and Alcolado, 1983; Alcolado and Herrera-Moreno, 1987). *Siderastrea siderea* was, however, more common in the deeper reefs where sediment deposition rates may be somewhat higher due to reduced wave energy.

Although relatively few stony corals were still affected at the time of our survey in July 1999, the populations at María la Gorda had undergone severe bleaching during the 1997-98 El Niño warming event (C. Carrodegua and N. Capetillo, personal communication). The relatively high estimates of recent partial-colony mortality found in the two more southerly reefs (Table 2) may, in part, represent post bleaching-related mortality and, in part, the effects of diseases that were on-going in summer 1999. Overall estimates of recent partial-colony mortality (3-10.5%) at María la Gorda varied from low to moderately high by comparison with results obtained in earlier AGRRA surveys (Kramer and Kramer, 1999; Lang, 1999; Leão et al., 1999; Steneck and Lang, 1999). Similarly, the mean percentages of old mortality (<35%) can be considered moderate.

Damselfish bites (primarily due to *Stegastes planifrons*) and their algal gardens, a suite of coral diseases, and fleshy macroalgae were the most common stressors of the large stony corals at María La Gorda. Injuries inflicted by the damselfish could also have facilitated the spread of coral diseases. The positive relationship between colony diameter and impacts (partial-colony mortality, percent of stony corals with disease and damselfish algal gardens) at the two southerly reefs suggests a gradual process of accumulation of partial-colony death as size (and, to some extent, age) increases.

The somewhat elevated densities of *S. planifrons* presumably resulted from a local deficit of carnivorous fishes (Claro and Cantelar Ramos, this volume). The high relative abundance of fleshy macroalgae (and perhaps the scarcity of calcareous macroalgae, Hay, 1997) is probably a response to the continued scarcity of the key herbivore, *Diadema antillarum* (Table 3), and to low herbivory by fishes (Claro and Cantelar Ramos, this volume). Hence, to sustain the María la Gorda coral reefs, the following management actions will be necessary:

- Avoid the construction of hotels and other major tourism infrastructure within the coastal zone of the reserve.
- Strictly prohibit all illegal fishing in the Península de Guanahacabibes Biosphere Reserve, allowing stocks of reef fishes to recover. (Partial enforcement of fishery regulations is ineffective, Watson et al., 1997.)

- Avoid nutrification (Alcolado et al., Instituto de Oceanología unpublished report) of the coastal waters. Currently the resort's wastewaters are poured into excavations in the karst and probably leak into the sea via groundwater. Thus, an alternate disposal system will be required.
- Deploy permanent mooring bouys at all dive sites.
- Ensure the resort divers are educated in safe reef-diving etiquette, and that the carrying capacity of the reefs (Hawkins and Roberts, 1997) is not exceeded.

### ACKNOWLEDGMENTS

The AGRRA organizing Committee, which facilitated financial support as described in the Forward to this volume, is also thanked for the invitation to collaborate in this regional initiative. We acknowledge the invaluable collaboration with, and support from, Reef Relief.

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Table 1. Site information for AGRRA stony coral and algae surveys in María la Gorda, Cuba.

Site name	Site code*	Reef type	Latitude (°')	Longitude (°')	Survey dates	Depth (m)	Benthic transects (#)	≥25 cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)
Yemayá	MG-Sh-1	Fringing lobe	21 50.059	84 29.390	July 8-9 1999	5-6	12	10.5	23.5 ± 4.5
Acuario	MG-Sh-2	Fringing lobe	21 47.644	84 30.733	July 6-7 1999	5-6	13	8.5	20.5 ± 5.5
Jardín de las Gorgonias	MG-D-1	Fringing spur	21 48.566	84 30.823	July 8-10 1999	8	14	7.5	17.0 ± 4.0
La Cadena Misteriosa	MG-D-2	Fringing spur	21 47.270	84 31.094	July 6-8 1999	11	15	7.5	15.5 ± 3.0

\* MG = María la Gorda; Sh = shallower; D = deeper

Table 2. Size and condition (mean ± standard deviation) of all stony corals (&gt;25 cm diameter), by site in María la Gorda, Cuba.

Site name	Stony corals		Partial-colony mortality (%)		Stony corals (%)			
	(#)	Diameter (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased
Yemayá	128	46.5 ± 23.5	3 ± 9	24.5 ± 27	27.5 ± 27.5	1.5	1	3
Acuario	108	60.5 ± 46	9 ± 18	28.5 ± 28.5	37.5 ± 29.5	2	5.5	6.5
Jardín de las Gorgonias	106	44 ± 19	5 ± 12	27 ± 28	33.5 ± 27.4	1	2	4
La Cadena Misteriosa	110	63.5 ± 37	10.5 ± 19	34.5 ± 32	45.0 ± 31.1	4.5	3.5	4.5

Table 3. Algal characteristics (mean ± standard deviation), abundance of stony coral recruits and *Diadema antillarum*, by site in María la Gorda, Cuba.

Site name	Quadrats (#)	Relative abundance (%)		Crustose coralline algae		Macroalgal		Recruits (#/0.0625m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )
		Macroalgae	Turf algae	Height (cm)	Index*	Height (cm)	Index*		
Yemayá	60	46 ± 20	37 ± 20	16 ± 12	97	2.1 ± 0.8	97	0.12	0
Acuario	60	55 ± 20	32 ± 21	13 ± 11	110	2.0 ± 0.8	110	0.10	0.8
Jardín de las Gorgonias	60	60 ± 18	25 ± 15	15 ± 11	150	2.5 ± 1.0	150	0.20	4.3
La Misteriosa	50	47 ± 24	35 ± 22	18 ± 17	113	2.4 ± 3.4	113	0.16	2

\*Macroalgal index = relative macroalgal abundance x macroalgal height

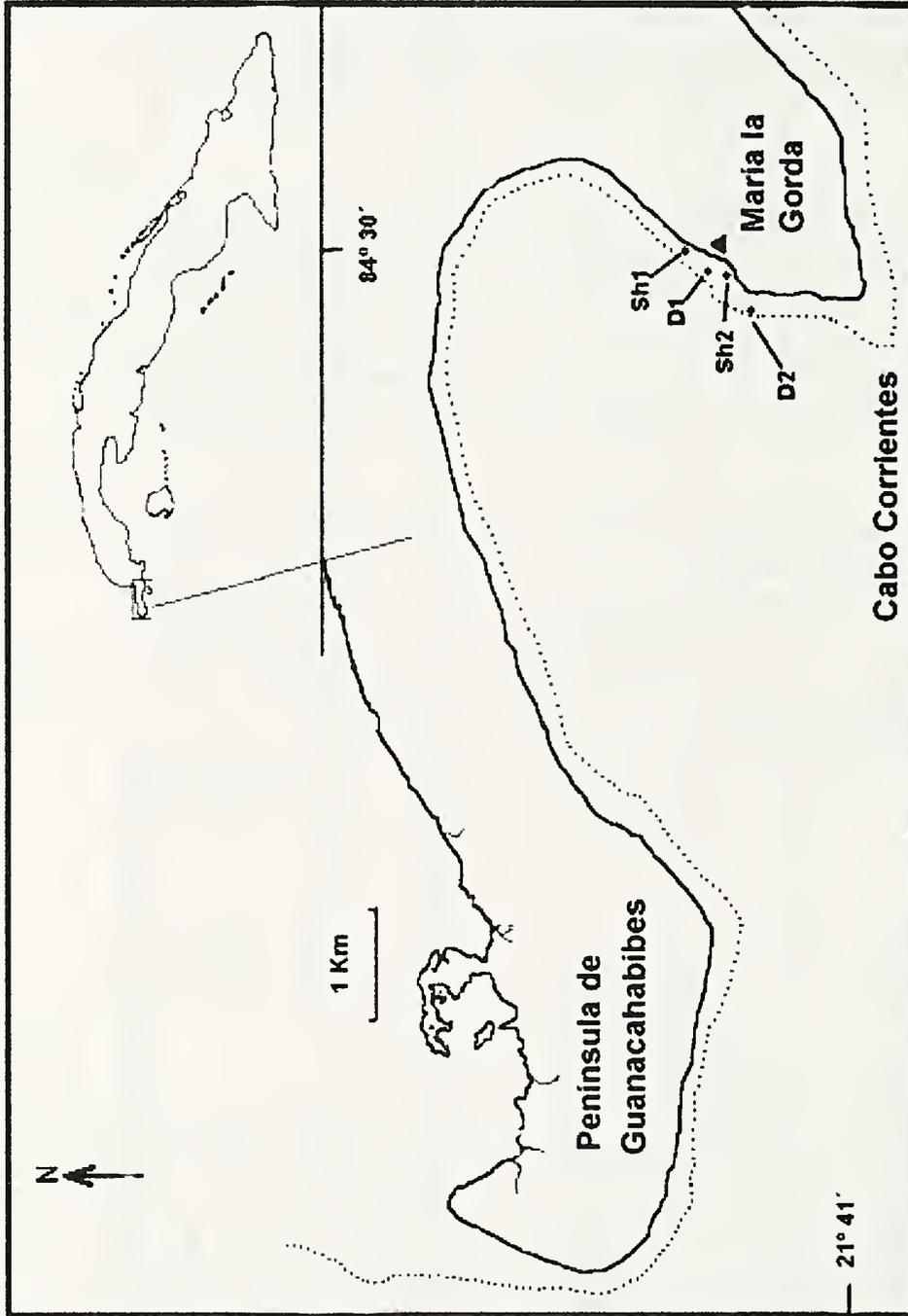


Figure 1. AGRRA survey sites in María la Gorda, Cuba. See Table 1 for site codes.

# RAPID ASSESSMENT OF CORAL COMMUNITIES OF MARÍA LA GORDA, SOUTHEAST ENSENADA DE CORRIENTES, CUBA (PART 2: REEF FISHES)

BY

RODOLFO CLARO<sup>1</sup> and KAREL CANTELAR RAMOS<sup>1</sup>

## ABSTRACT

An assessment of fish community structure was carried out in the fringing reef at María la Gorda in the Guanacahabibes Biosphere Reserve, following the Atlantic and Gulf Rapid Reef Assessment protocols. Quantitative “all species” surveys were also taken for comparison with existing information elsewhere. Unexpectedly for a no-take reserve, fish density and biomass were very low in comparison with other Cuban reefs. In the two shallower (5-8 m) reefs, parrotfishes and grunts were numerically dominant, and grunts were dominant in terms of biomass, but most were small-sized species. A balistid (*Melichthys niger*) dominated one of the deeper (9-10 m) reefs. Groupers and most snappers were uncommon at all depths. The scarcity of medium- and large-sized fishes is a consequence of illegal fishing in the reserve. Reduced predation by piscivores may also have caused the proliferation of damselfishes, some of which are killing corals and may be facilitating the spread of coral diseases.

## INTRODUCTION

The leeward fringing coral reef at María la Gorda, a small resort in the Península de Guanahacabibes Biosphere Reserve in western Cuba, is considered one of the nation's most beautiful and best conserved. The only tourist facility, which has 29 rooms and belongs to a scuba diving resort, is located near the southeastern tip of Ensenada de Corrientes (Fig.1). Not far from this resort is a small settlement with 22 houses, a frontier guard-post, a visitor's center for the reserve, and a meteorological radar station. There are fewer than 200 inhabitants (including tourists) during the high tourism season (about six months of the year). The Reserve has been a no-take area for fishing since 1996 by a Resolution of the Ministry of Fishing Industry. Hence its coral reefs would be expected to have a relatively intact fish population. However, there is no existing quantitative information regarding the fish communities in this area.

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Reef lobes of irregular shapes, but elongated parallel to the bay's coastline, rise 1-2 m above a gently sloping sandy terrace between depths of 5 and 15 m. Live stony coral cover and the density of "large" ( $\geq 25$  cm diameter) colonies were both considered moderate in July 1999, slightly higher at 5-6 m in two shallow lobes (5-6 m) than in two deeper lobes (8-11 m), and dominated by *Montastraea annularis*, *M. franksi*, *Siderastrea siderea* and *M. faveolata* (Alcolado et al., this volume). Nevertheless, the preferred dive sites for tourists are on the higher relief, structurally more complex, and biologically more diverse reefs that are located at depths of 20-30 m along the outer margin of the terrace.

The topography on the reef lobes is very irregular, potentially providing plenty of shelter for fishes and mobile invertebrates. Fishermen who have worked for many years in the area and local villagers emphasized that, until about 20 years ago, the fringing reef supported numerous, large- and medium- sized predatory fishes such as snappers (lutjanids), groupers (serranids), and large-sized parrotfishes (scarids).

The María la Gorda area was traditionally fished by commercial fishermen using such artisanal fishing gear as fish traps and hooks and lines. Eventually, non-commercial fishers began to use spearguns. Illegal speargun and hook-and-line fishing increased during the early 1990s due to the economic crisis in the country. Small-scale commercial fishing currently occurs outside the reserve, where a few tourists are probably also allowed to fish. The latter is not significant, but due to the small size of the reserve, resident fishes can be harvested close to its boundaries during their local migrations. Illegal subsistence fishing, practiced by local residents and outsiders, continues within and outside the reserve, due to a low level of enforcement of the regulations.

Located approximately eight km southwest of the reserve's boundary, at a depth of 25-40 m in Cabo Corrientes, is a well-known spawning site for mutton snapper (*Lutjanus analis*) in June, cubera (*L. cyanopterus*) and dog snapper (*L. jocu*) in July-August, Nassau grouper (*Epinephelus striatus*) in December-January, plus black grouper (*Mycteroperca bonaci*) and yellowfin grouper (*M. venenosa*) in January-March. These aggregations are fished mainly with hooks and lines and fish traps (Claro and Lindeman, in press).

The objective of this paper is to contribute to the assessment of reef condition at María la Gorda by examining the status of the fish community and its relationship to impacts.

## METHODS

The reef at María la Gorda was strategically chosen on the basis of its reputation among divers. Four fish surveys were carried out at two depths (5-8 m and 9-13 m) on representative fore-reef lobes in July 1999 (Table 1). Fishes were visually censused using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) Version 2.2 protocols (see Appendix One, this volume), modified as described below. Quantitative data for the AGRRA fishes in each reef were collected along six belt transects, each 50-m long x 2 m wide, provided quantitative data for the AGRRA fishes in each reef. The transect tape was

set on the substratum by the diver, and the counts started five minutes later. Counts of serranids were restricted to species of *Epinephelus* and *Mycteroperca*; scarids and haemulids (grunts) less than 5 cm in length were not tallied. All surveys were made between 10:00 and 17:00 hours by two divers. For identifications, we used Humann's (1994) guide to reef fishes.

Leatherjackets were classified according to Eschmeyer's (1998) revision, in which *Balistes* and *Melichthys* are in the Balistidae, while *Aluterus* and *Cantherhines* are assigned to the Monacanthidae. Biomass estimates of the AGRRA fishes were made using the length-weight relationships given by Garcia-Arteaga et al. (1997) for Cuban fishes. A ranking index (% sighting frequency x % abundance) was calculated for the 25 most abundant of these fishes (data summed over all sites). Species diversity ( $H'$ ), richness ( $R_1$ ), and evenness ( $J$ ) indices for all fish species were estimated on the basis of two qualitative, 30-minute roving diver surveys in each reef.

We made six additional "all species" belt transects (each 50 m long x 2 m wide) to count and estimate sizes of all fishes in three of the reefs, to compare with similar surveys made at 15 m depth in 16 sites of the Archipelago los Canarreos (SW Cuba) in 1988-1989. Even though the data are not fully comparable statistically, due to the small sample size at María la Gorda, the comparison may serve as a primary reference for assessing current conditions in María la Gorda's population of reef fishes.

## RESULTS

### Fish Diversity

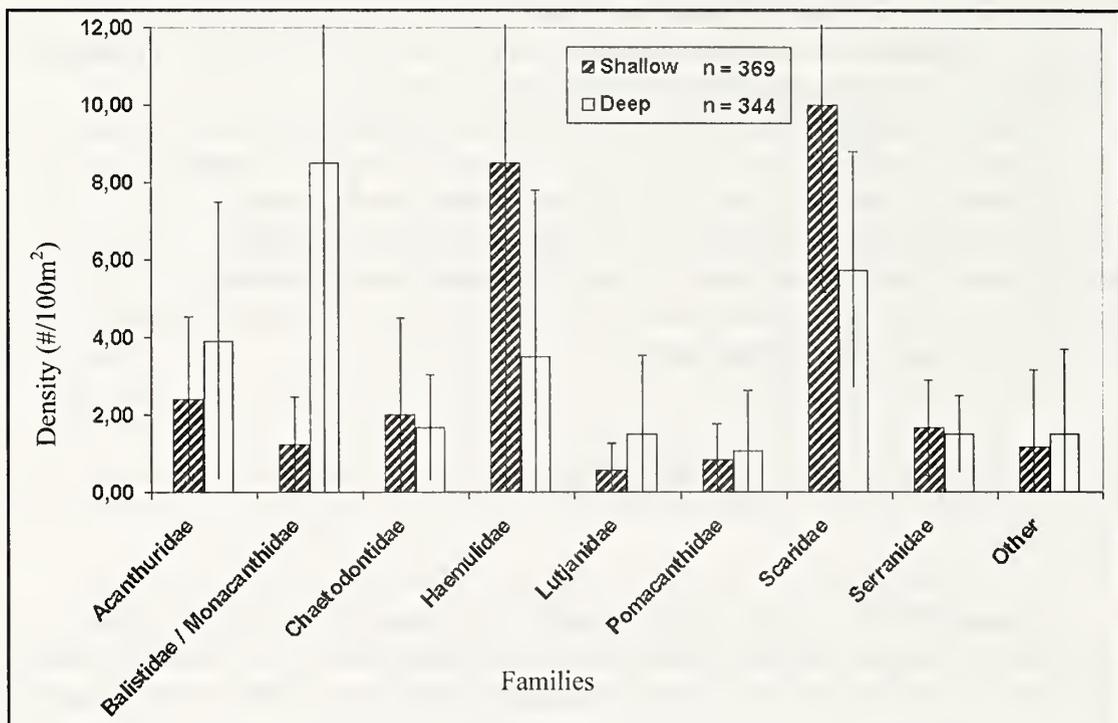
A total of 88 fish species (Table 2) were found during the eight roving diver censuses (mean = 58/reef, sd= 6). Relative to the two shallow reefs, estimates of species diversity and evenness in the two deeper reefs were slightly lower, while species richness estimates were slightly higher. The most frequently sighted and abundant of these species were the blue chromis (*Chromis cyanea*), bluehead wrasse (*Thalassoma bifasciatum*), and bicolor damselfish (*Stegastes partitus*) (Table 3).

In the "all species" belt transects, there were 42-54 species per reef (for a total of 71 species). Between 25 and 29 of the AGRRA species were seen in the AGRRA belt transects in each reef (total =45). Highest in the AGRRA hierarchy index were French grunt (*Haemulon flavolineatum*), striped parrotfish (*Scarus iserti*, =*S. croicensis*, see Eschmeyer, 1998) and blue tang (*Acanthurus coeruleus*) (Table 4).

### Fish Density

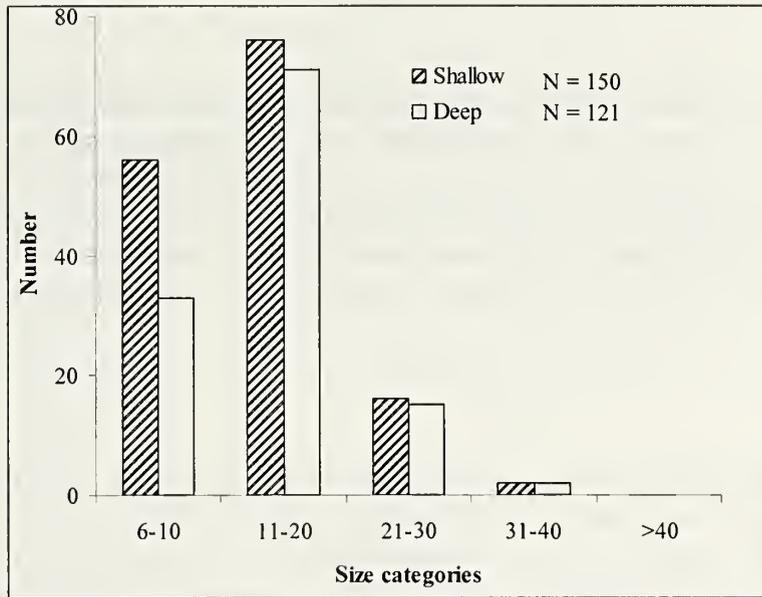
The total density of the AGRRA fishes (Table 5) was nearly identical in the two shallower reefs (~30.3 and 31.2 individuals/100 m<sup>2</sup>) but more variable in the two deeper reefs (~20.3 and 37.5 individuals/100 m<sup>2</sup>). Herbivorous fishes, primarily small-sized parrotfishes, comprised over half (~52%) of the total numerical count while grunts constituted the second most abundant family overall (Fig. 2). Striped parrotfish dominated in one of the shallow reefs (Sh-1), whereas French grunts were predominant in

the other (Sh-2). A large group of black durgelon (*Melichthys niger*) and blue tang (*Acanthurus coeruleus*) predominated in one of the deeper reefs (D-1), where yellowtail snapper (*Ocyurus chrysurus*) were also moderately common. The second deeper reef (D-2) had the fewest fishes overall and lacked dominance by any species. Surgeonfishes (acanthurids) were slightly more abundant in the two deeper reefs than in the shallower reefs. Densities of snappers, angelfishes (pomacanthids), butterflyfishes (chaetodontids), and select groupers (*Epinephelus* and *Mycteroperca*) each totaled <6 individuals/100 m<sup>2</sup> for the four reefs. However, a larger number of snappers (*O. chrysurus* and *Lutjanus apodus*) were observed close to the edge of the reef (and outside the area covered by the belt transects) during the roving diver censuses at one of the deeper reefs (D-1).

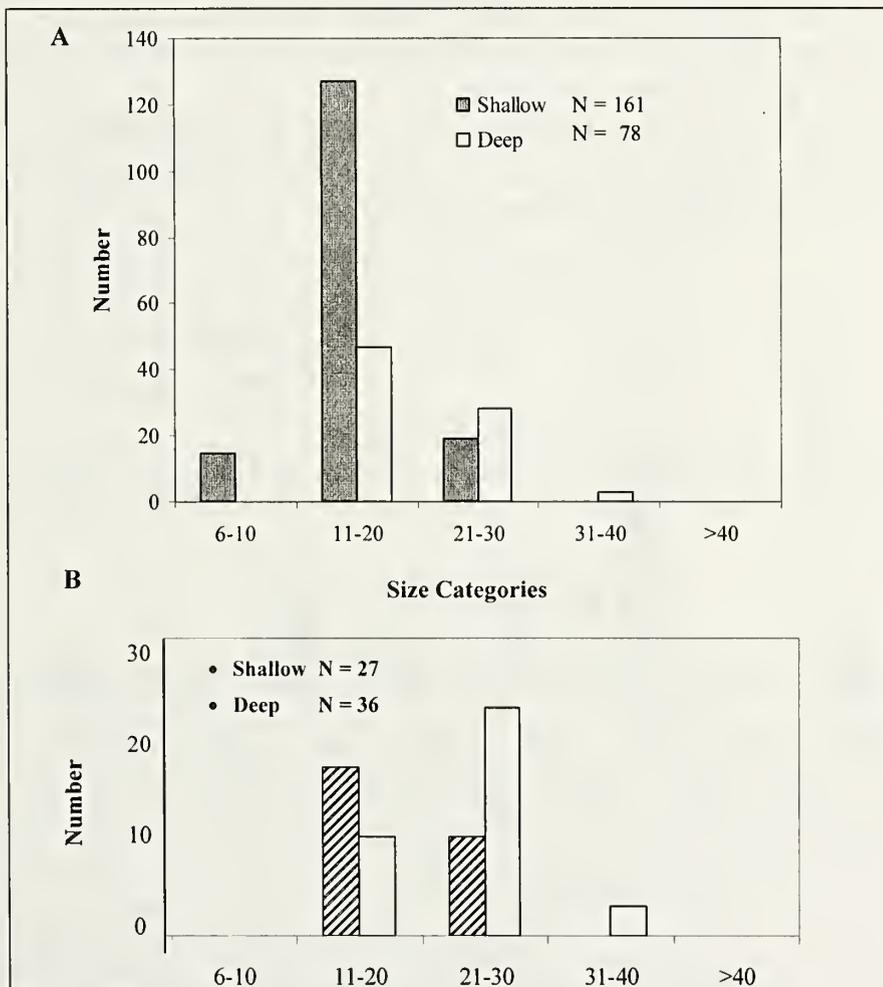


**Figure 2.** Mean fish density (no. individuals/100 m<sup>2</sup> ± sd) for AGRRR fishes in María la Gorda, Cuba. Other = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Microspathodon chrysurus*, *Sphyræna barracuda*.

The mean density of fishes in the “all species” belt transects was nearly 20 times greater than that of the selected species in the AGRRR fish transects (541 versus ~30 individuals/100 m<sup>2</sup>). Damsel fishes (pomacentrids) were the most abundant family with a mean density of 126 individuals/100 m<sup>2</sup>, and most (>93%) of these were bicolor damselfish (Table 6). At approximately 36 individuals/100 m<sup>2</sup>, the average density of herbivores (all parrotfishes and surgeonfishes, *Microspathodon chrysurus*) was almost 20 times greater than the average density of carnivores (all snappers, *Epinephelus*, *Mycteroperca*), of which there were only 2.6 individuals/100 m<sup>2</sup>.



**Figure 3.** Size frequency distribution (in cm) of herbivores (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) in María la Gorda, Cuba.



**Figure 4.** Size frequency distribution in cm of carnivores, as (A) all lutjanids and haemulids  $\geq 5$  cm, select serranids and (B) all lutjanids and select serranids, in María la Gorda, Cuba.

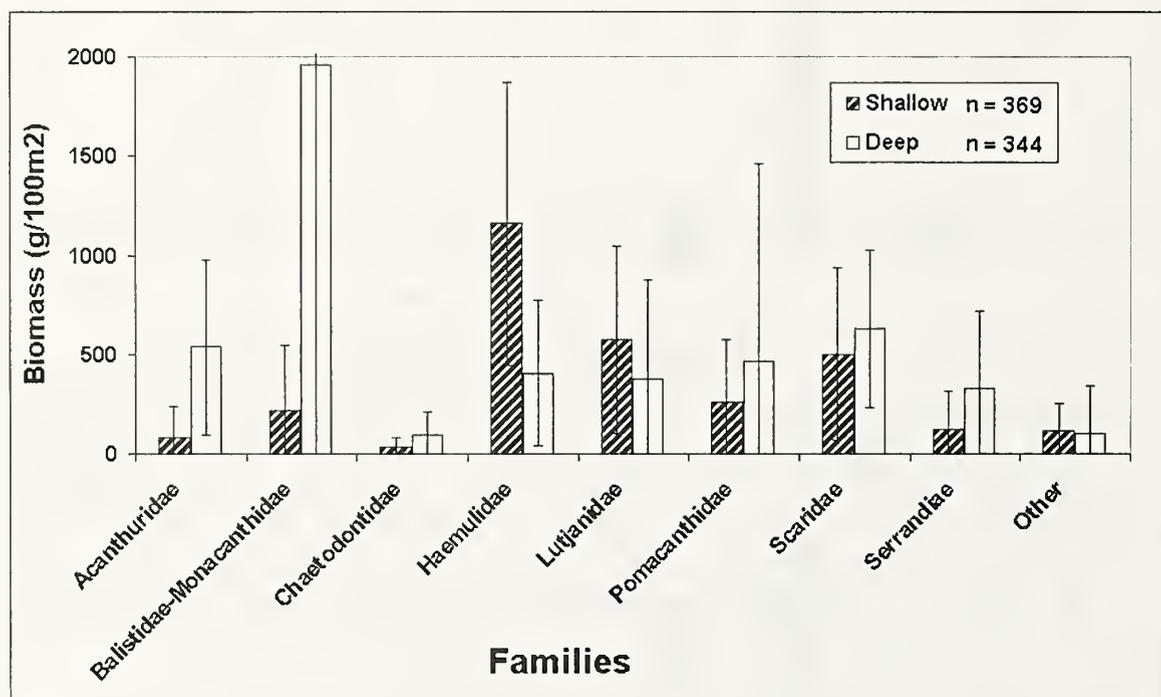
## Fish Size

Most of the key herbivores (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) were less than 20 cm in total length (Fig. 3). Large-sized parrotfishes (such as *Scarus guacamaia* and *S. coelestinus*) were absent from the AGGRA belt transects, and most of the surgeonfishes were also small-sized. Regardless of whether or not grunts are included with the key AGGRA carnivores, most predatory fishes were in the 11-20 cm size class in the two shallower reefs and somewhat larger (21-30 cm) in the two deeper reefs (Fig. 4A,B).

## Fish Biomass

The total biomass of the AGGRA fishes (Table 7) ranged from 2,312 g/100 m<sup>2</sup> (in D-2, which also had the fewest fishes) to 7,502 g/100 m<sup>2</sup> (in D-1, where it was elevated by the presence of the yellowtail snapper and black durgéon). Despite their similar densities, the two shallower reefs differed somewhat in biomass due to their differing species composition, the French grunts in Sh-2 being slightly larger than the striped parrotfish in Sh-1. The key herbivores overall constituted 29% of the total fish biomass. The biomass of grunts was greater in shallow water, whereas the biomass of black durgéon, surgeonfishes and, to a lesser extent, the AGGRA-listed groupers (*Epinephelus*, *Mycteroperca*, were higher in the deeper reefs (Fig. 5).

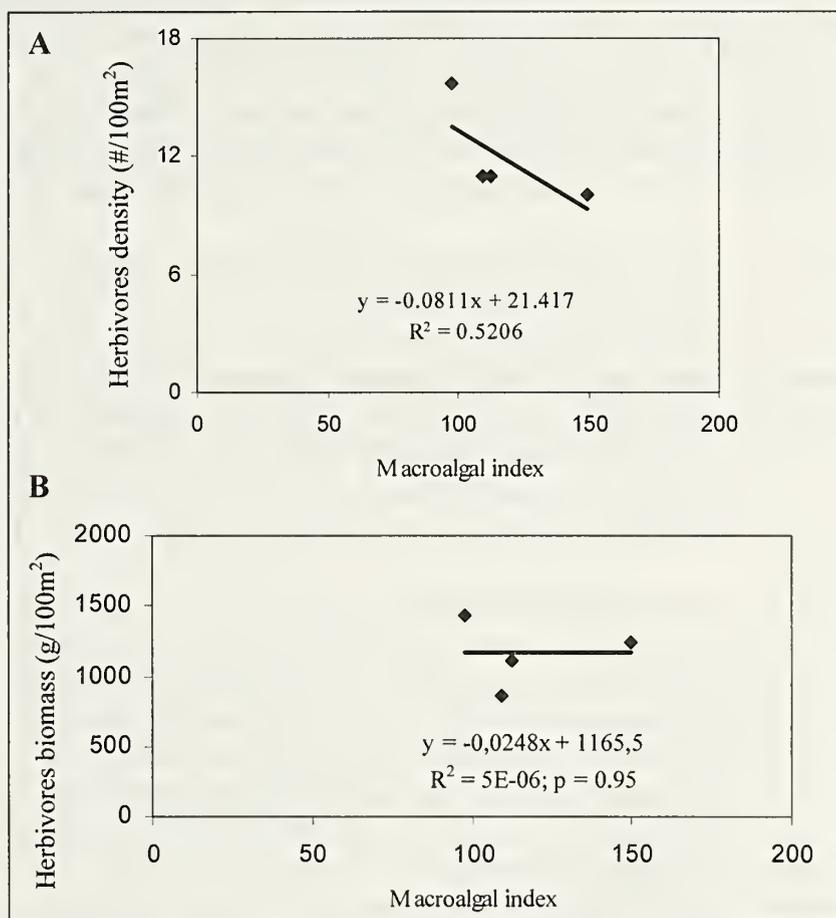
In the "all-species" surveys, the total fish biomass equaled 8,620g/100 m<sup>2</sup>, of which 3,256 and 499 g/100 m<sup>2</sup> respectively, were comprised of the herbivores (many of which were small-sized scarids) and carnivores (Table 6).



**Figure 5.** Mean fish biomass (grams/100m<sup>2</sup>  $\pm$  sd) for AGGRA fishes in María la Gorda, Cuba. Other = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Microspathodon chrysurus*, *Sphyraena barracuda*.

## Macroalgae

The macroalgal index (data from Alcolado et al., this volume) had a significant ( $r^2 = 0.52$ ,  $p = 0.95$ ) inverse relationship with herbivore density, but showed no relationship with total herbivore biomass (Fig. 6).



**Figure 6.** Regression plots between mean macroalgal index and (A) mean herbivore density (no. individuals/100m<sup>2</sup>), (B) mean herbivore biomass (grams/100m<sup>2</sup>), by site in María la Gorda, Cuba.

## DISCUSSION

Relatively high values for the species diversity indices were found during the qualitative fish surveys at María la Gorda (Table 2). Moreover, the total number of species recorded in the “all species” transects (42 to 54,  $n = 6$  sites) was slightly higher than the value in the Archipiélago los Canarreos (mean = 42,  $sd = 5$ ,  $n = 21$  sites) recorded in 1988-1989 (Instituto de Oceanología database). The most frequently sighted and abundant of these species (blue chromis, bicolor damselfish, and bluehead wrasse) are common dominants in many other Cuban coral reefs (Claro and García-Arteaga, 1994; Claro et al., 1998; Claro and Parenti, 2001).

Small-sized species of parrotfishes and grunts dominated the AGRRA fishes in belt transects at the two shallower reefs. Total fish biomass in María la Gorda was low

compared to other Cuban reefs. For example, corresponding biomass values for the "all species" census were approximately 30% lower than those at comparable depths in the Archipiélago los Canarreos and Archipiélago Sabana-Camagüey in 1988-1989, and less than 50% of that recorded for the Archipiélago Jardines de la Reina in 1997 (Claro and Parenti, 2001).

Herbivore biomass was 37% lower in María la Gorda than that found in 1996 in the marine reserve at the Archipiélago Jardines de la Reina, where larger-sized species are more abundant (Sierra et al., 2001). Nevertheless, the densities and biomass of herbivorous fishes in María la Gorda were higher than those recorded at the Archipiélago los Canarreos (Table 6) and at the Archipiélago Sabana-Camagüey in 1989-1990, and even exceeded corresponding values for Martinique, Guadeloupe, and Key West, Florida (Claro and Parenti, 2001).

The mean biomass of carnivorous fishes (selected groupers and snappers) was very low; values were about eight times higher in the 1988-1989 "all species" surveys at Archipiélago los Canarreos (Table 6). Particularly notable was the scarcity of snappers, which are usually common in Cuban reefs (Claro and García-Arteaga, 1994; Claro et al., 1998). For example, average snapper biomass at the Archipiélago los Canarreos in 1988-1989 was thirteen times higher than that found in 1999 off María la Gorda. The slightly higher abundance and size of carnivores in one of its deeper reefs (D-1) may result in part from the higher relief and topographic complexity at its edge, where snappers and black durgon tend to aggregate.

Medium- to large-sized species of fishes were uncommonly scarce off María la Gorda. Habitat conditions (live stony coral cover, topographic relief and complexity, water quality, etc.) in 1999 seemed excellent and capable of supporting much higher fish densities. Anecdotal information and qualitative information (personal observations of the senior author) during the 1970s and 1980s indicate that María La Gorda's fish populations, especially the snappers and groupers, are drastically reduced. Indeed, this area, along with Cabo San Antonio at the westernmost edge of the Biosphere Reserve, had been well known for its abundance of large fishes.

Management objectives for the Biosphere Reserve are intended to protect its natural resources, to attract tourists, and to serve as reproductive replenishment reserves for downcurrent reefs; yet it is clear that fishing regulations here are poorly enforced. Large-sized fishes are vulnerable even to very low harvesting levels (Watson et al., 1997). Furthermore, fish that are subjected to spearfishing are likely to swim away from divers, which further diminishes the reserve's touristic value.

Removing large predators alters a reef's trophic balance, potentially allowing small territorial fishes and invertebrate predators of scleractinians, such as the snail, *Coralliophila abbreviata*, and the bristle worm, *Hermodice carunculata*, to proliferate. Indeed, along with blue chromis, loreto (*Gramma loreto*), and bluehead wrasse, the density of damselfishes at María la Gorda was eightfold greater than had been found in the Archipelago los Canarreos in 1988-1989, and the bicolor damselfish was 13 times more abundant (Claro, unpublished data; Table 5 for damselfishes). Individuals of the three-spot damselfish (*Stegastes planifrons*) were conspicuously destroying live corals in order to construct algal gardens with which to defend their territory and attract females. Fishing the larger-sized herbivorous parrotfishes can also have unintended consequences

when macroalgae and turf algae increase in abundance, as may have happened at María la Gorda (Alcolado et al., this volume).

For all of these reasons, it is very important that the regulation prohibiting fishing at María la Gorda be strictly and effectively enforced. We also recommend increasing the no-take area protected from commercial fishing, educating the professional dive guides about the ecological and economic value of intact residential fish populations, and periodic monitoring of the local fish communities. Furthermore, the commercial and sport fishing that is carried out nearby at Cabo Corrientes during the fish spawning aggregations should be completely suspended. Establishing a carefully self-regulated "catch and release" fishery for certain non-reef fishes (tarpon, bonefish) should be considered as an alternative source of revenue for displaced fishers.

### ACKNOWLEDGMENTS

The AGRRA organizing Committee, which facilitated financial support as described in the Forward to this volume is thanked for the invitation to collaborate in this regional initiative. We acknowledge the invaluable collaborations with, and support from, Reef Relief.

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Table 1. Site information for AGRRA fish surveys in María la Gorda, Cuba.

Site name	Site code	Reef type	Lat.itude (° ')	Longitude (° ')	Survey date	Depth (m)	(#/10 m) <sup>1</sup>	% live stony coral cover (mean ± sd) <sup>1</sup>	Macroalgal index <sup>1</sup>	AGRRA 50 m fish transects (#)
Yemayá	Sh-1	Fringing lobe	21 50.059	84 29.390	July 8, 1999	5-6	10.5	23.5 ± 4.5	97	6
Acuario	Sh-2	Fringing lobe	21 47.644	84 30.733	July 7 1999	5-8	8.5	20.5 ± 5.5	110	6
Jardín de las Gorgonias	D-1	Fringing spur	21 48.566	84 30.823	July 9 1999	9-10	7.5	17.0 ± 4.0	150	6
La Cadena Misteriosa	-D-2	Fringing spur	21 47.270	84 31.094	July 6 1999	12-13	7.5	15.5 ± 3.0	113	6

<sup>1</sup> from Alcolado et al. (this volume).

Table 2. Species numbers in the surveys in María la Gorda, Cuba; species indices based on roving diver surveys.

Site code	Species number		Roving diver surveys		Species indices	
	Belt transects	"All species"	Diversity (H')	Richness (R <sub>i</sub> )	Evenness (J)	
Sh-1	26	54	3.4	4.8	0.6	
Sh-2	26	49	3.3	4.3	0.6	
D-1	29	42	3.2	4.9	0.5	
D-2	25	-	2.7	5.1	0.4	
<b>Σ All sites</b>	<b>45</b>	<b>71</b>	<b>3.5</b>	<b>6.2</b>	<b>0.5</b>	

Table 3. Twenty-five most frequently sighted fish species during rover diver surveys in María la Gorda, Cuba, with mean density for species in AGRRA belt transects

Species name	Ranking index <sup>2</sup>	Sighting frequency (%) <sup>3</sup>	Density (#/100 m <sup>2</sup> )
<i>Chromis cyaneus</i>	1	100	
<i>Thalassoma bifasciatum</i>	2	100	
<i>Stegastes partitus</i>	3	100	
<i>Chromis multilineatus</i>	4	100	
<i>Gramma loreto</i>	5	100	
<i>Haemulon flavolineatum</i>	6	100	5.46
<i>Scarus iserti</i> (= <i>S. croicensis</i> ) <sup>1</sup>	10	100	3.29
<i>Lutjanus apodus</i>	13	100	0.29
<i>Melichthys niger</i>	14	10	4.58
<i>Sparisoma viride</i>	15	100	1.25
<i>Sparisoma aurofrenatum</i>	20	100	0.63
<i>Stegastes planifrons</i>	18	100	
<i>Stegastes adustus</i> (= <i>S. dorsopunicans</i> ) <sup>1</sup>	22	100	
<i>Halichoeres bivittatus</i>	23	100	
<i>Serranus tigrinus</i>	24	100	0
<i>Haemulon plumieri</i>	25	100	0.42
<i>Acanthurus coeruleus</i>	16	87	2.46
<i>Sparisoma chrisoptera</i>	28	85	
<i>Halichoeres garnoti</i>	17	85	
<i>Caranx ruber</i>	8	75	
<i>Ocyurus chrysurus</i>	12	75	0.63
<i>Haemulon sciurus</i>	19	75	1.38
<i>Mulloidichthys martinicus</i>	21	75	
<i>Scarus taeniopterus</i>	30	75	0.88
<i>Chaetodon capistratus</i>	27	75	1.42

<sup>1</sup>Species names according to Eschmeyer's (1998) revision.

<sup>2</sup>Ranking index = (% sighting frequency x % abundance).

<sup>3</sup>Sighting frequency = percent of all roving diver surveys in July, 1999 in which the species was recorded.

Table 4. Sighting frequency and density for the twenty-five species highest in the hierarchy index in AGRRA belt transects in María la Gorda, Cuba.

Species name	Hierarchy index <sup>2</sup>	Sighting frequency	Density (#/100 m <sup>2</sup> )
<i>Haemulon flavolineatum</i>	1	0.75	5.46
<i>Scarus iserti</i> (= <i>S. croicensis</i> ) <sup>1</sup>	2	0.83	3.29
<i>Acanthurus coeruleus</i>	3	0.79	2.46
<i>Chaetodon capistratus</i>	4	0.58	1.42
<i>Sparisoma viride</i>	5	0.75	1.25
<i>Haemulon sciurus</i>	6	0.50	1.38
<i>Melichthys niger</i>	7	0.67	4.58
<i>Scarus taeniopterus</i>	8	0.42	0.88
<i>Cephalopholis cruentata</i>	9	0.67	0.92
<i>Sparisoma chrysopterygum</i>	10	0.42	0.88
<i>Sparisoma aurofrenatum</i>	11	0.42	0.63
<i>Microspathodon chrysurus</i>	12	0.33	0.88
<i>Scarus vetula</i>	13	0.38	0.46
<i>Ocyurus chrysurus</i>	14	0.25	0.63
<i>Acanthurus chirurgus</i>	15	0.33	0.50
<i>Cephalopholis fulva</i> (= <i>Epinephelus fulvus</i> ) <sup>1</sup>	16	0.25	0.38
<i>Haemulon plumieri</i>	17	0.25	0.42
<i>Sparisoma atomarium</i>	18	0.25	0.42
<i>Balistes vetula</i>	19	0.29	0.29
<i>Holacanthus ciliaris</i>	20	0.25	0.29
<i>Lutjanus apodus</i>	21	0.21	0.29
<i>Acanthurus bahianus</i>	22	0.17	0.21
<i>Mycteroperca tigris</i>	23	0.17	0.17
<i>Pomacanthus paru</i>	24	0.25	0.56
<i>Bodianus rufus</i>	25	0.17	0.29

<sup>1</sup>Species names according to Eschmeyer's (1998) revision.

<sup>2</sup>Hierarchy index = % sighting frequency x % density.

Table 5. Density of AGRRA fishes, by site in María la Gorda, Cuba.

Site code	Density (#/100 m <sup>2</sup> )							Total
	Herbivores			Carnivores		Others	Total	
	Acanthuridae	Scaridae (≥5 cm)	<i>Microspathodon chrysurus</i>	Haemulidae (≥5 cm)	Lutjanidae			
Sh-1	2.67	12.33	0.17	9.00	0.17	1.50	5.34	31.83
Sh-2	2.17	7.17	1.67	13.33	0.83	1.83	3.33	30.33
D-1	4.50	5.17	0.33	2.33	2.67	1.33	20.67	37.50
D-2	3.33	6.33	1.33	4.67	0.33	1.67	2.66	20.33
<b>All sites</b>	<b>3.17</b>	<b>7.75</b>	<b>0.88</b>	<b>7.33</b>	<b>1.00</b>	<b>1.58</b>	<b>7.99</b>	<b>29.7 ± 6.9</b>

<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.

Table 6. Mean density and biomass of "all species" in María la Gorda (this study) and Los Canarreos (in 1988-1989).

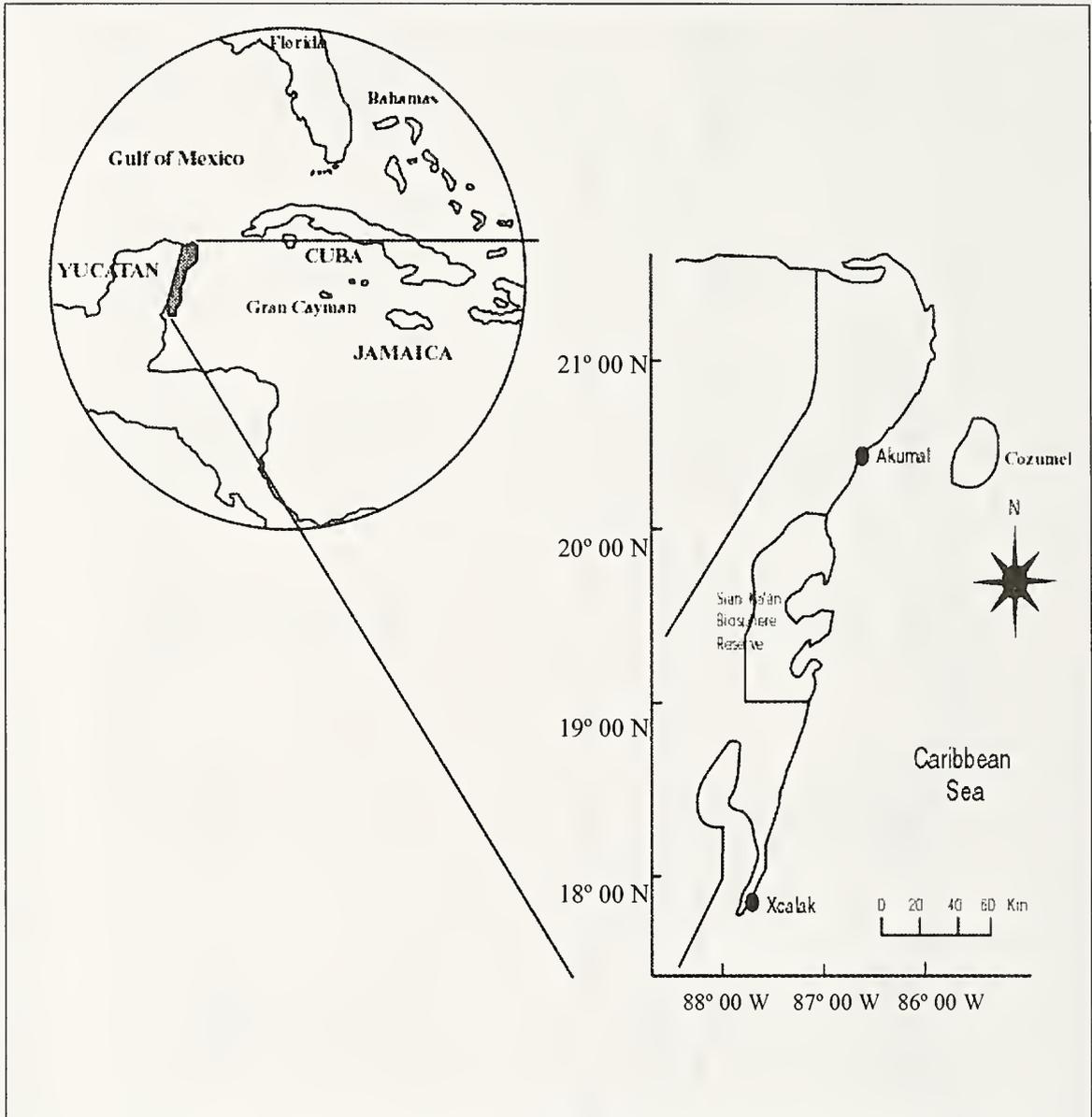
Taxon and parameter	Canarreos (1988-1989, n=16 reefs, depth=15 m)	María La Gorda (1999, n=4 reefs, depth=5-13 m)
<sup>1</sup> Herbivore density (#/100 m <sup>2</sup> )	22	36
Herbivore biomass (g/100 m <sup>2</sup> )	1,800	3,256
<sup>2</sup> Snapper and grouper density (#/100 m <sup>2</sup> )	8	2
Snapper and grouper biomass (g/1100 m <sup>2</sup> )	3,979	499
Damselfish density (#/100 m <sup>2</sup> )	15	126
Bicolor damselfish density (#/100 m <sup>2</sup> )	9	118
<b>Total fish density (#/100 m<sup>2</sup>)</b>	<b>197</b>	<b>541</b>
<b>Total fish biomass (g/100 m<sup>2</sup>)</b>	<b>12,432</b>	<b>8,620</b>

<sup>1</sup>Herbivore = all herbivores; <sup>2</sup>Snapper and grouper = all species of lutjanids and serranids.

Table 7. Biomass of AGRRA fishes, by site in María la Gorda, Cuba.

Site code	Biomass (g/100 m <sup>2</sup> )							Total
	Herbivores		Carnivores			Others		
	Acanthuridae	Scaridae (≥5 cm)	<i>Microspathodon chrysurus</i>	Haemulidae (≥5 cm)	Lutjanidae	Serranidae <sup>1</sup>		
Sh-1	324.4	1105.7	1.9	949.6	46.1	148.9	217.6	2,794
Sh-2	165	677.5	18.6	1,213.1	193.6	202.9	869	3,340
D-1	623.6	693.4	16.5	364.7	691.2	177.4	5025	7502
D-2	450.5	648.5	14.9	447.5	56.8	478.9	215.3	2312
<b>All sites</b>	<b>413.0</b>	<b>643.1</b>	<b>16.7</b>	<b>675.1</b>	<b>313.9</b>	<b>386.4</b>	<b>2,036.5</b>	<b>3,987 ± 2,380</b>

<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.



**Figure 1.** AGRRA survey sites (Akumal, Xcalak) in Quintana Roo, México. Modified from Núñez-Lara et al. (this volume).

# RAPID ASSESSMENT OF MÉXICO'S YUCATÁN REEF IN 1997 AND 1999: PRE- AND POST-1998 MASS BLEACHING AND HURRICANE MITCH (STONY CORALS, ALGAE AND FISHES)

BY

ROBERT S. STENECK<sup>1</sup> and JUDITH C. LANG<sup>2</sup>

## ABSTRACT

Rapid assessments were made along México's Yucatán coast in six fore reefs each off Akumal (northern area) and Xcalak (southern area) and in two Xcalak patch reefs. Live stony coral cover averaged <20% overall in March 1999. At this time, both recent and old partial-colony mortality of the  $\geq 25$  cm stony corals were significantly higher in the Xcalak patch reefs than in the fore reefs, and more corals were moderately bleached at all depths off Xcalak than off Akumal. Anomalously high sea surface temperatures, having caused widespread bleaching in September 1998, possibly persisted longer at Xcalak because different exposure to hurricanes caused the southern Yucatán to cool later than the northern area. Bleaching may have rendered the *Montastraea annularis* species complex more susceptible to white-band disease, particularly in the Xcalak patch reefs. Partial mortality of stony corals and macroalgal abundance increased significantly in the Akumal fore reefs between 1997 and 1999, when macroalgal abundance and biomass were significantly higher than in Xcalak. Macroalgal increases off Akumal could have resulted from anthropogenic nutrient enrichment from groundwater and/or reduced grazing by *Diadema antillarum* and herbivorous fishes.

## INTRODUCTION

Are coral reefs "dying" throughout the wider Caribbean? If they are, at what scale and at what rate? The Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocols are tools for rapidly quantifying spatial and temporal patterns in the condition of coral reefs. Their value is in providing species-explicit and commensurable ways of comparing reefs within regions, larger scale bioregions, or the entire, tropical western Atlantic (Ginsburg et al., 2000; Steneck et al., 2000; Kramer, this volume).

The AGRRA protocols were also designed to record indicators of stress and potential agents of disturbance. We use stress to mean any loss of *productivity* and disturbance to mean any loss of *biomass* in the organism or assemblage in question (after

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Grime, 1977, 1989; Steneck, 1988; Steneck and Dethier, 1994). Specifically for stony corals, transient, nonlethal bleaching indicates stress whereas disturbances are indicated by signs of disease and patterns of mortality.

Early versions of the AGRRA protocols were prototyped off two villages of the Mexican Caribbean (Fig. 1). Narrow fringing and fringing-barrier reefs skirt the eastern flank of the Yucatán Peninsula—where many coastal ecosystems are under assault from the rapid growth of the tourist industry (Jordán Dahlgren, 1993; Lang et al., 1998). Akumal, a resort community with a resident population of >600 near the southern end of the northerly Cancún-Tulum corridor, is surrounded by numerous large-scale developments. Xcalak, near the southern border with Belize, has a population of about 300 and is being transformed from a reliance on offshore fishing to ecotourism.

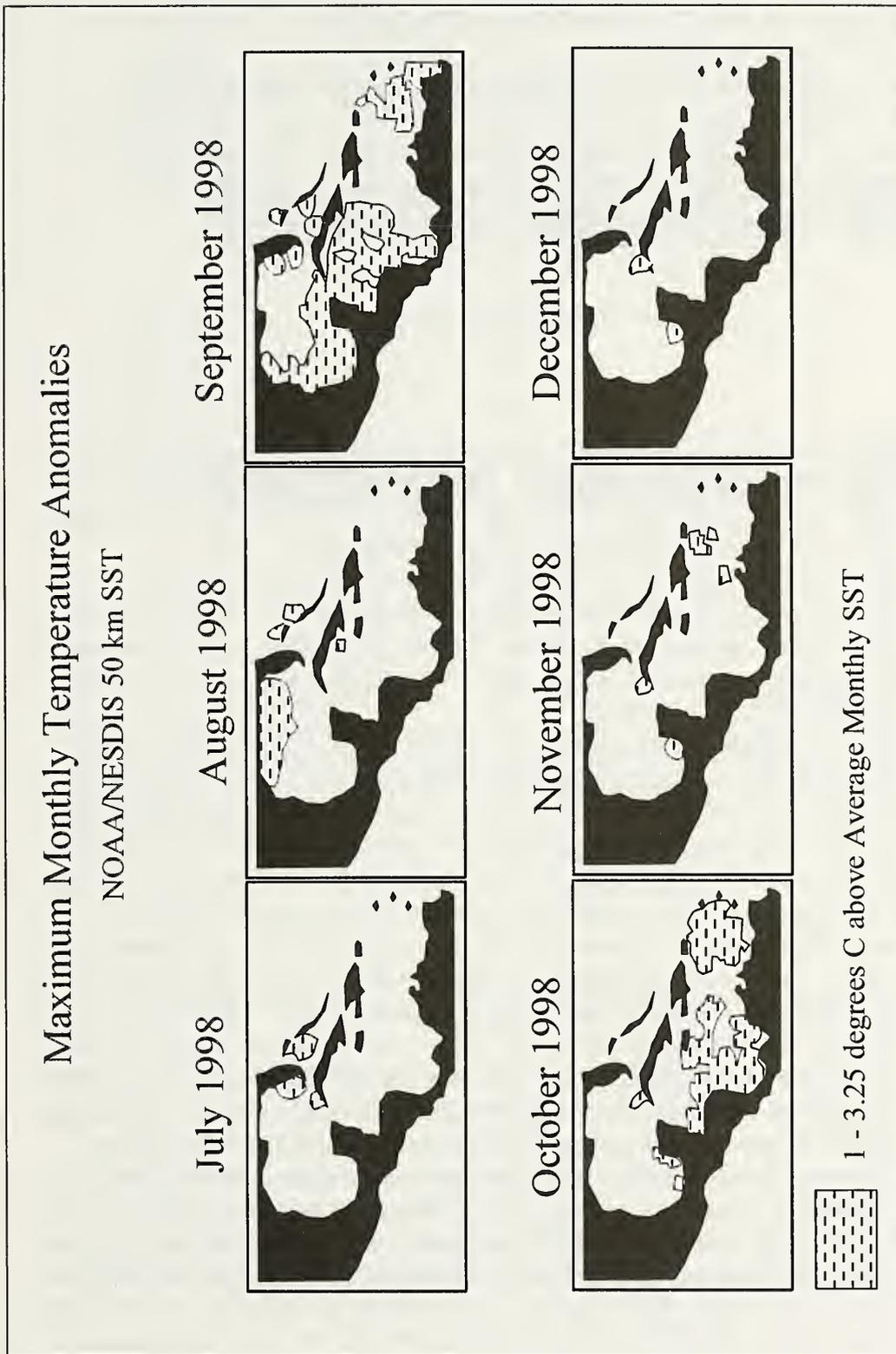
Our initial assessments of these reefs were made in March and August 1997. Resurveys in March 1999 were conducted six-seven months after the first reports of thermally induced (Fig. 2) bleaching in the Yucatán (Kramer et al., 2000), five months after Hurricane Georges passed near the northern Yucatán, and four months after Hurricane Mitch affected the southern area. We sought to determine patterns in coral mortality, whether particular species were suffering higher than average mortality, and whether mortality patterns corresponded with known stressors and disturbance agents (bleaching, hurricanes, macroalgae) or with inferred changes in the reefs' trophic structure (i.e., reductions in herbivory).

## METHODS

Chosen for assessment were six, moderate-relief “middle-lobe” fore reefs at depths of 11–18 m near Akumal (Fig. 1; Table 1). Comparable habitats were selected at 8.5–16.5 m in six fore reefs off Xcalak along with two shallow (1–3 m) patch reefs in its wider lagoon. All sites were considered to be representative of local reefs based on the knowledge and advice of residents who had dived there for years.

The draft AGRRA benthos protocol was utilized by six divers in early March 1997. The following August two divers applied the revised AGRRA Version 1 procedures at two reefs off Akumal (Dick's, Las Redes). In early March 1999, when three-six divers used the AGRRA Version 2.0 protocols (see Appendix One, this volume) at all 14 reef sites, live coral cover was measured to the nearest 5 cm. The maximum diameter of stony corals (i.e., scleractinians and *Millepora complanata*) was approximated to the closest 5 cm with one side of the algal quadrat for scale. Bleaching was scored as “light” or “moderate,” completely bleached stony corals being absent at this time. Two species of *Colpophyllia* (*C. breviserialis*, *C. natans*) were recognized. Humann (1993) served as the primary field guide for other scleractinians. Stony coral recruits were not quantified. Surficial sediment was removed from the algal quadrats by gentle hand sweeping before estimating crustose coralline algal abundance.

A limited number of fish surveys were made by one diver (Steneck) in 1999. The AGRRA version 2.0 fish belt-transect protocol (see Appendix One) was modified by swimming three transects, each 30 long x 2 m wide, in six of the fore reefs (Akumal–Média Luna, Dick's, Xcalak–Coral Gardens, Chimney, El Quebrado, Dos Cocos). Scarids (parrotfish) and haemulids (grunts) <5 cm in length were not counted. Although



**Figure 2.** Sea surface thermal anomalies in 1998 (for July 4, Aug. 1, Sept. 1, Oct. 6, Nov. 1 and Dec. 1) show the development of a thermal anomaly around the entire Yucatán by early September, its contraction to the south by early October and disappearance by early November. (Images from NOAA/NESDIS).

dives were made from 9 a.m. to 5:30 p.m., most of the fish surveys were conducted during mid-day hours (10 a.m.-2 p.m.). The Fish Bite Method (see Appendix One) was used to assess herbivory in Dick's and Las Redes reefs by one diver (J. Reichman) in August 1997.

Data for all species were pooled for certain sites in some analyses; in other analyses the data for certain species were pooled across sites. Variance is given either as standard deviation (sd) or as standard error (se). Changes in the AGRRA benthos protocol between March 1997 and 1999 render some comparisons (e.g., of mortality patterns and macroalgal biomass) between those two dates impossible.

## RESULTS

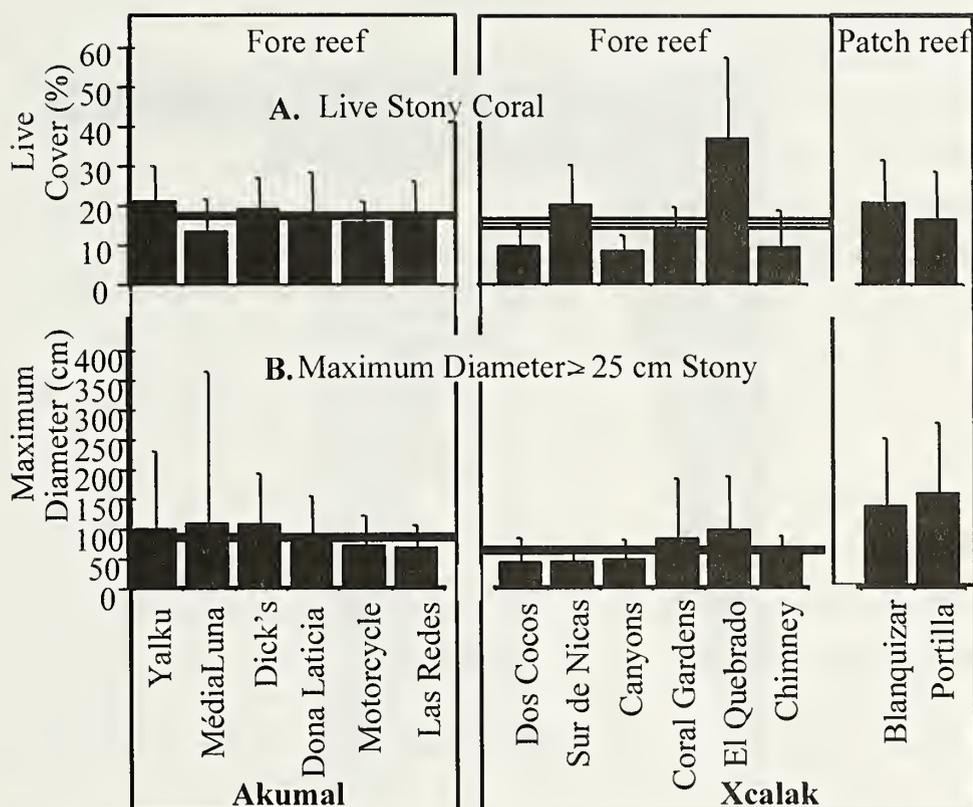
### Stony Corals

Live stony coral cover, first measured in March 1999, averaged less than 20% for most (10/14) of the surveyed reefs (Fig. 3A; Table 1). Variance among the fore reefs was lower off Akumal than Xcalak, where the live cover in one reef was about four times higher than at three others (37% versus ~9%, respectively).

Nineteen species of "large" ( $\geq 25$  cm maximum diameter) stony corals were quantified in our surveys. Eighteen were found in the fore reefs where the *Montastraea annularis* species complex predominated, numerically comprising >50% of the surveyed colonies in 1999 (with *M. annularis* > *M. faveolata* >> *M. franksi*) and each remaining species constituting <7% (Fig. 4; Table 2).

The maximum diameter of the  $\geq 25$  cm fore-reef corals averaged 76 cm (se=24, n=12 reefs). Colonies were somewhat smaller off Xcalak (mean=62, se=22, n=6 reefs) than in Akumal (mean=89, se=20, n=6 reefs) and there was considerable variation within some reefs, especially Akumal's Média Luna (Fig. 3B; Table 3). *A. palmata* was clearly the largest of the fore-reef corals, averaging 205 cm in maximum diameter (Fig. 5; Table 2), and three (*A. palmata*, *M. franksi*, *M. faveolata*) of the four largest species were among the six most abundant of the fore-reef corals. Nevertheless, when abundance is expressed as percent relative coverage (mean number of colonies/m x mean maximum diameter in cm), co-dominants were *M. annularis* and *M. faveolata*, each with 10.5% of the total.

Only nine species of  $\geq 25$  cm stony corals were recorded in the Xcalak patch reefs. *M. annularis* was the numerical dominant with 75% of all colonies. *Acropora cervicornis* and *Agaricia tenuifolia* each had 8.5%, *Porites porites* comprised 6.5%, while all the remainder (*A. palmata*, *Agaricia agaricites*, *Porites astreoides*, *Colpophyllia natans*, *Millepora complanata*) were each <3%. Stony corals were substantially larger in the two patch reefs than in the fore reefs (Table 3), with an overall mean of 150 cm (se=14), primarily due to the higher abundance of *M. annularis* which here averaged ~165 cm in maximum diameter.

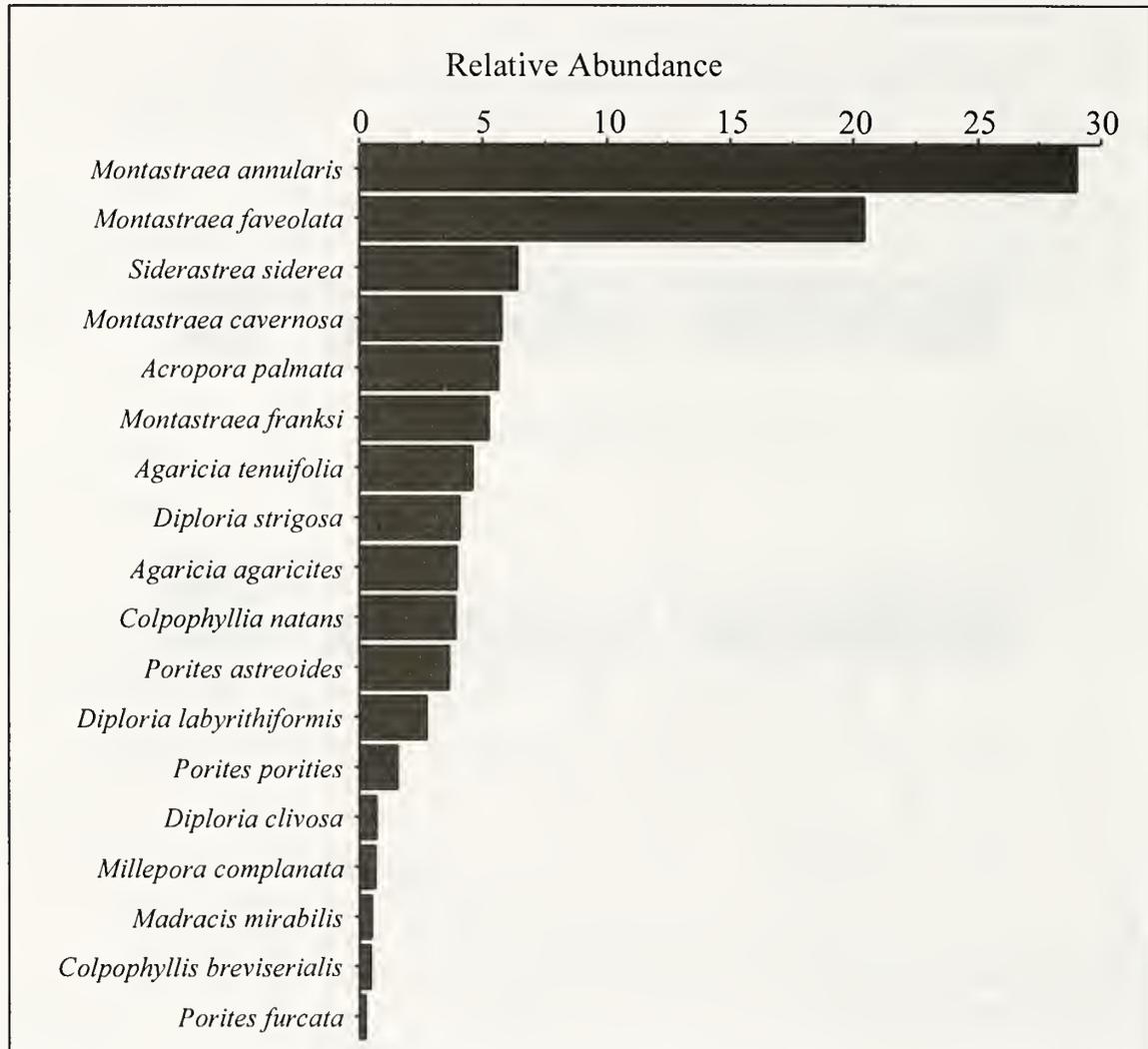


**Figure 3.** Mean ( $\pm$  standard deviation) for (A) percent live stony coral cover and (B) maximum (max.) diameter of all stony corals ( $\geq 25$  cm max. diameter) in March 1999 by site for Akumal ( $n=336$ ) and Xcalak ( $n=333$  in fore reefs and 196 in patch reefs). Horizontal lines = pooled mean  $\pm$  standard error for each habitat.

### Stony Coral Condition

We saw no evidence of bleaching during the 1997 surveys. In March 1999, moderate bleaching affected  $<3\%$  of the  $\geq 25$  cm stony corals in the Akumal fore reefs and about 10% in each habitat off Xcalak (Fig. 6A; Table 3). A further 4-18% of the colonies in each reef were considered to be lightly bleached at this time.

None of the  $\geq 25$  cm stony corals that were censused at Dick's and Las Redes reefs in August 1997 appeared diseased, although signs of white-band disease (WBD) and yellow-blotch disease were noticed in a few colonies of *A. palmata* and *M. faveolata*, respectively (Lang, personal observations). White plague was widespread in March 1999, however, with surveyed corals in 5/6 of the Akumal fore reefs, 3/6 of the Xcalak fore reefs and both patch reefs showing signs of infection (Fig. 6B; Table 3). The percentage of diseased colonies varied between areas and habitats, being  $\sim 1\%$  in the Xcalak fore reefs,  $\sim 5\%$  in Akumal and  $\sim 10\%$  in the Xcalak patches. Most affected were colonies of *Montastraea annularis* (fore reefs—60% at Akumal and 57% at Xcalak; patch reefs—95%),

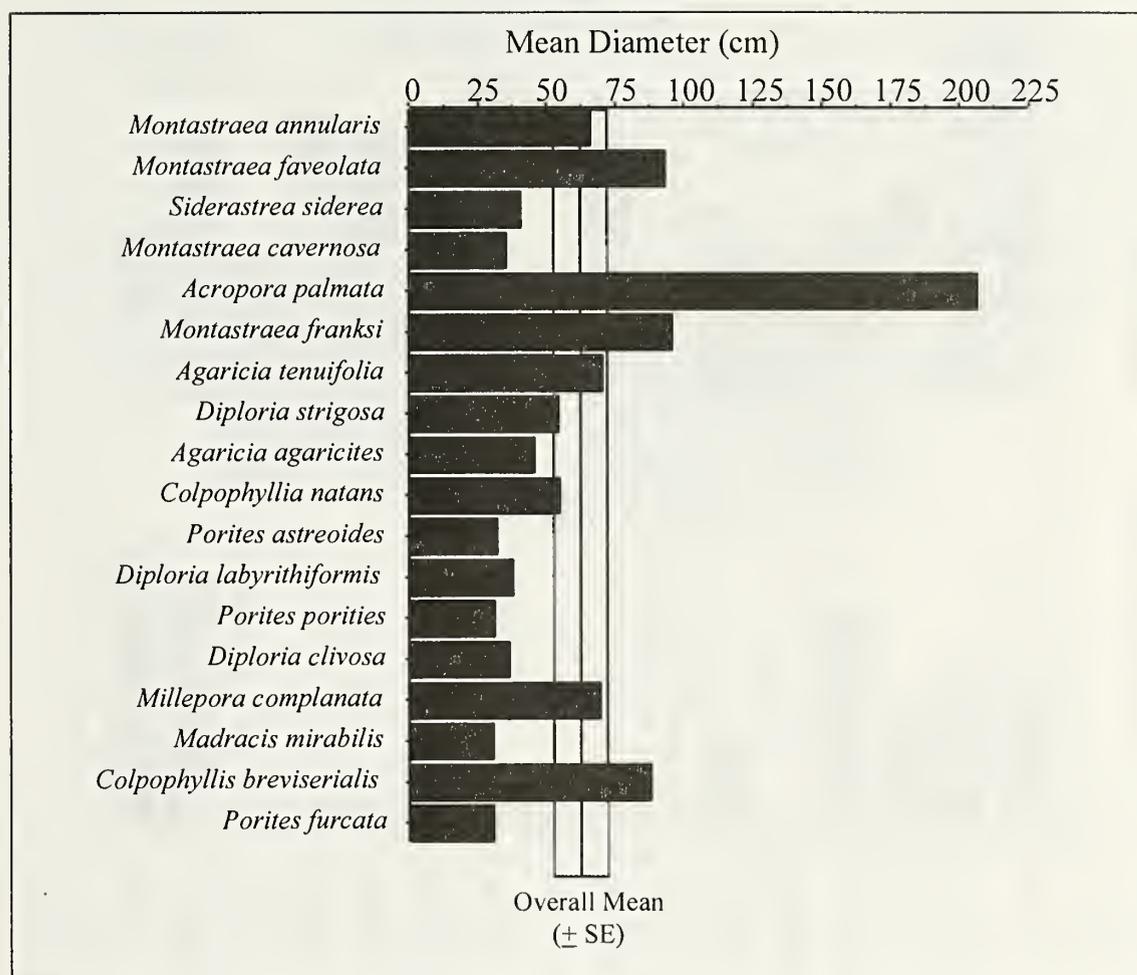


**Figure 4.** Relative abundance as % of the Akumal and Xcalak fore-reef stony corals (all  $\geq 25$  cm max.diameter;  $\Sigma=669$ ) in March 1999, ordered from most abundant to least abundant species.

followed by *M. faveolata* and *M. franksi*. Black-band disease was seen in one colony of *Agaricia tenuifolia* off Akumal (Table 3, footnote).

Recent partial-colony mortality (hereafter recent mortality) of large stony corals averaged 1.5% (sd =2.5) in the two Akumal fore reefs that were surveyed in August 1997. Overall estimates of recent mortality in the fore-reef were higher (Akumal–4.5%, Xcalak–5.0%) in March 1999, but revealed no large-scale differences between the two areas (Fig. 7, Table 3). Old partial-colony mortality (hereafter old mortality) in the two Akumal fore reefs averaged 31.5% (se=2) in August 1997. Old mortality was somewhat elevated in fore reefs with above-average values for recent mortality, but overall estimates in 1999 were virtually identical in the two areas, averaging 27.5% (se=7.0, n=12 reefs).

Recent partial mortality averaged 28% in the two Xcalak patch reefs in 1999, when mean values of old partial mortality were ~40%. Hence total (recent + old) partial mortality in the patch reefs was approximately twice as high as in the fore reefs (Fig. 7; Table 3). Regardless of location, recent mortality declined steadily between 7 and 15 m in

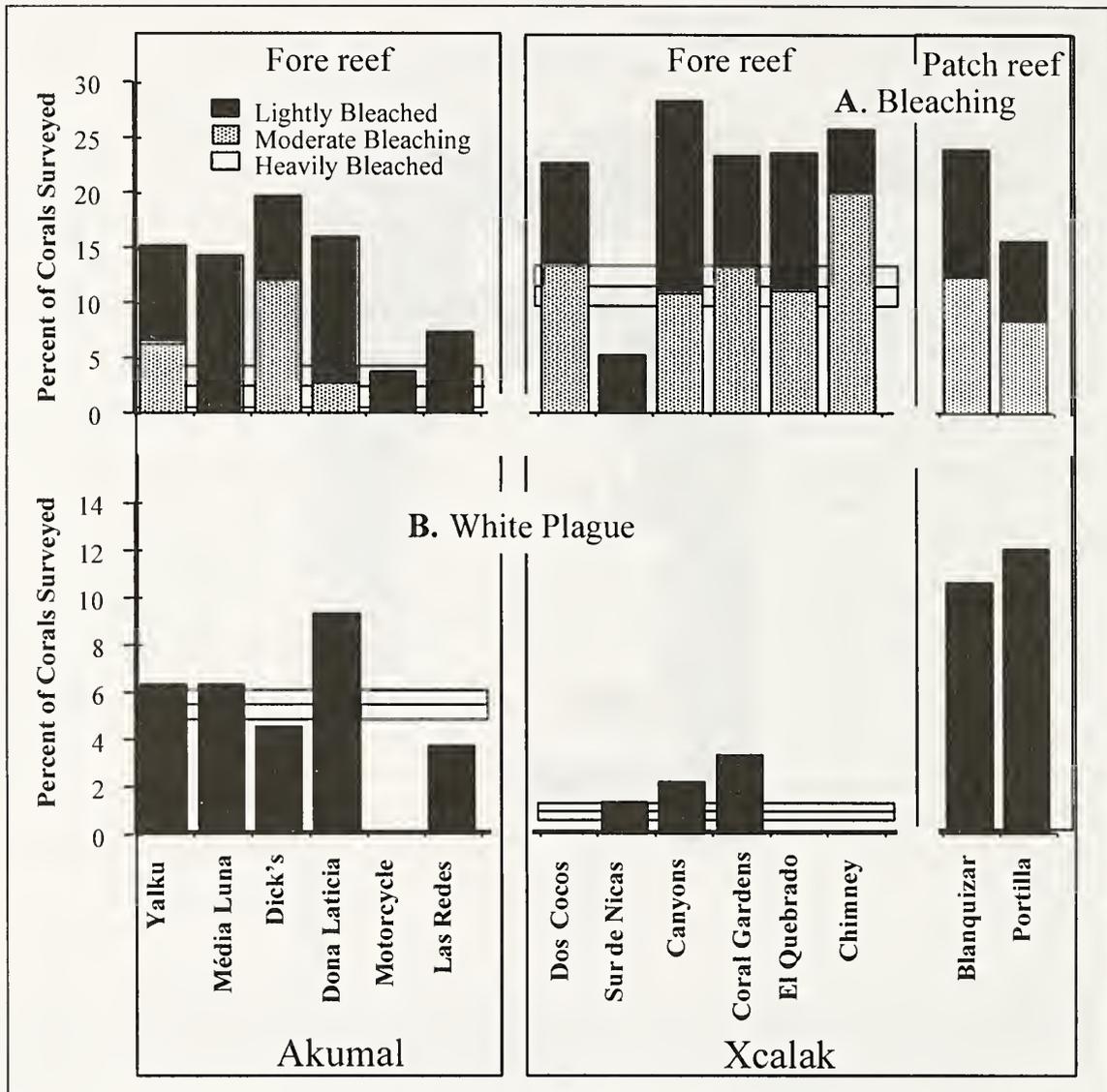


**Figure 5.** Mean maximum diameter of the Akumal and Xcalak fore-reef stony corals (all  $\geq 25$  cm max. diameter,  $n=669$ ) in March 1999, ordered from most abundant to least abundant species (as in Figure 4). Vertical lines = mean  $\pm$  standard error for all species pooled.

the fore reefs (Fig. 8) and a similar, but less, dramatic depth-related decline was evident in old mortality as well.

The percentage of large stony corals that were “standing dead” (entire upper surfaces dead, colony still in growth position) in March 1999 was very variable among sites (Table 3), averaging 4-5% overall in the fore reefs and about 9% in the patch reefs. However, a much higher proportion of the *A. palmata* were standing dead ( $n=16/41$  colonies in the fore reefs,  $n=2/6$  colonies in the patch reefs). Detached or fallen stony corals were rare in all the surveyed reefs.

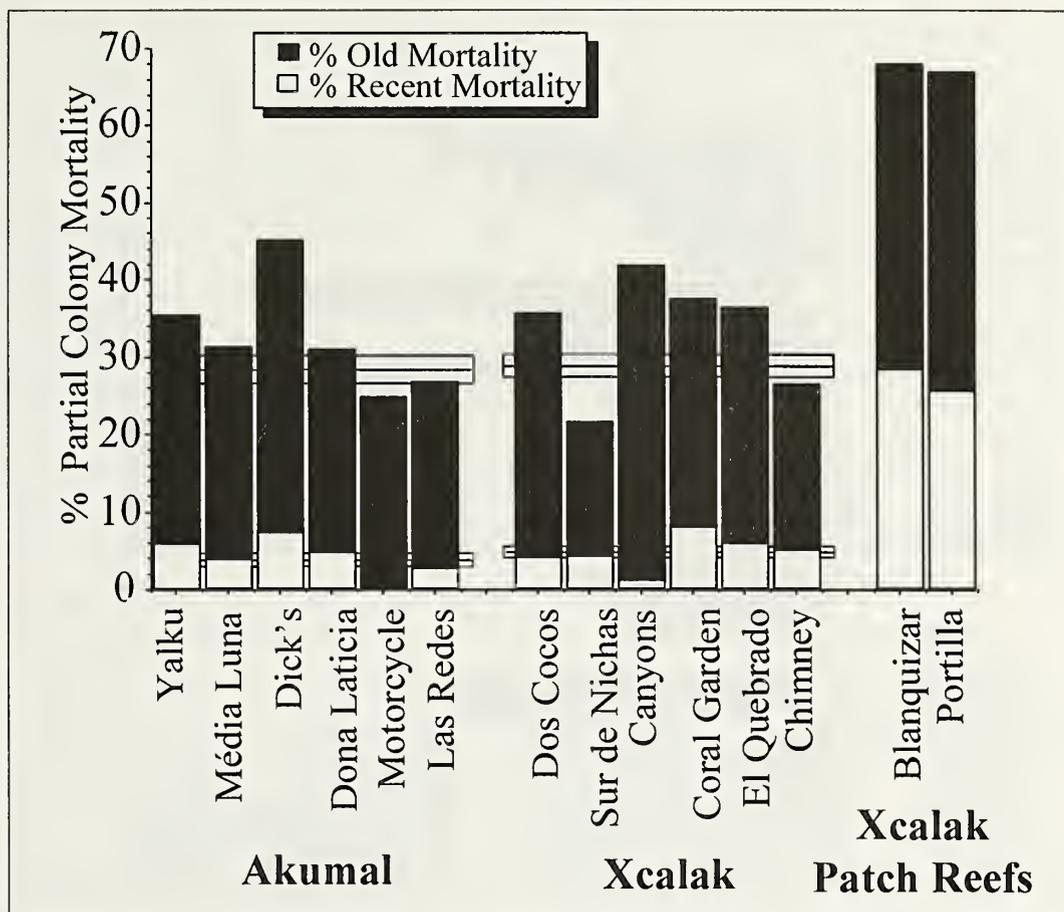
Recent mortality in *Millepora* and *Agaricia* is likely to be underreported. Their corallite structures are relatively indistinct, and soon after death the skeletons are probably scored as “old dead” by the AGRRA protocol. Bearing this caveat in mind, recent and old mortality patterns varied by species in 1999, with mean values of 3% and 22%, respectively, for the 18 fore-reef species (Fig. 9; Table 2). Values for recent mortality that were well above the average were found in *M. annularis* and *M. faveolata*, whereas lower than average values were recorded for *Montastraea cavernosa*, *M. complanata* and *Colpophyllia breviserialis* (Fig. 9, Table 2). Percent recent mortality of



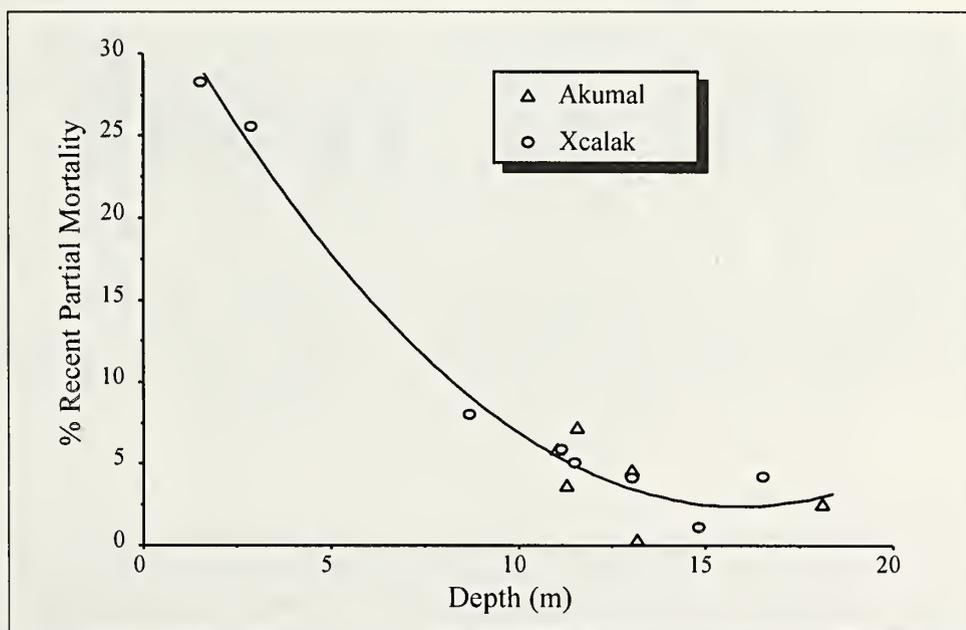
**Figure 6.** Percent of (A) bleaching with averages among reefs only determined for moderate bleaching and (B) white plague in stony corals (all  $\geq 25$  cm max. diameter) in March 1999 by site for Akumal ( $n=336$ ) and Xcalak ( $n=333$  in fore reefs and 196 in patch reefs). Mean and standard error notation as in Figure 3.

colony surfaces of *A. palmata* was somewhat greater in Akumal (mean=6.5, se=3,  $n=31$  colonies) than off Xcalak (mean=2, se=2,  $n=10$  colonies). Fore-reef species with higher than average values for old (and total) partial-colony mortality were *A. palmata*  $\gg$  *Agaricia tenuifolia*  $>$  *M. mirabilis*  $\sim$  *M. annularis*.

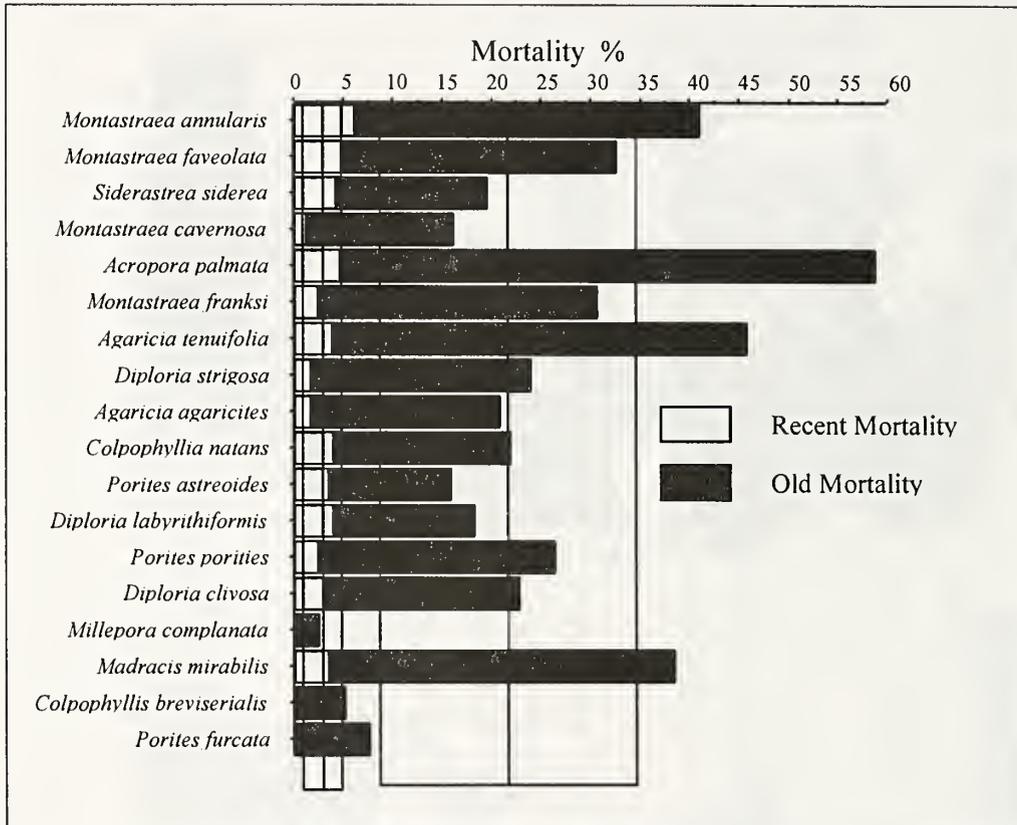
Mean values of recent mortality in the patch reefs were especially high for *M. annularis* (31.5%), *A. agaricites* (24%), *Acropora cervicornis* and *A. tenuifolia* (each 15%), while old mortality was greatest in *A. palmata* (87%), *P. porites* (64%), *A. agaricites* (55%) and *M. annularis* (37%). The only colony of *Millepora complanata* censused here was scored as having 100% old mortality.



**Figure 7.** Total (recent and old) partial colony mortality of all stony corals ( $\geq 25$  cm max. diameter) in March 1999 by site for Akumal ( $n=336$ ) and Xcalak ( $n=333$  in fore reefs and 196 in patch reefs). Mean and standard error notation as in Figure 3.



**Figure 8.** Plot of mean recent partial mortality of all stony corals ( $\geq 25$  cm max. diameter) versus water depth in March 1999 at each site off Akumal and Xcalak. Curve fit is 3rd order polynomial.



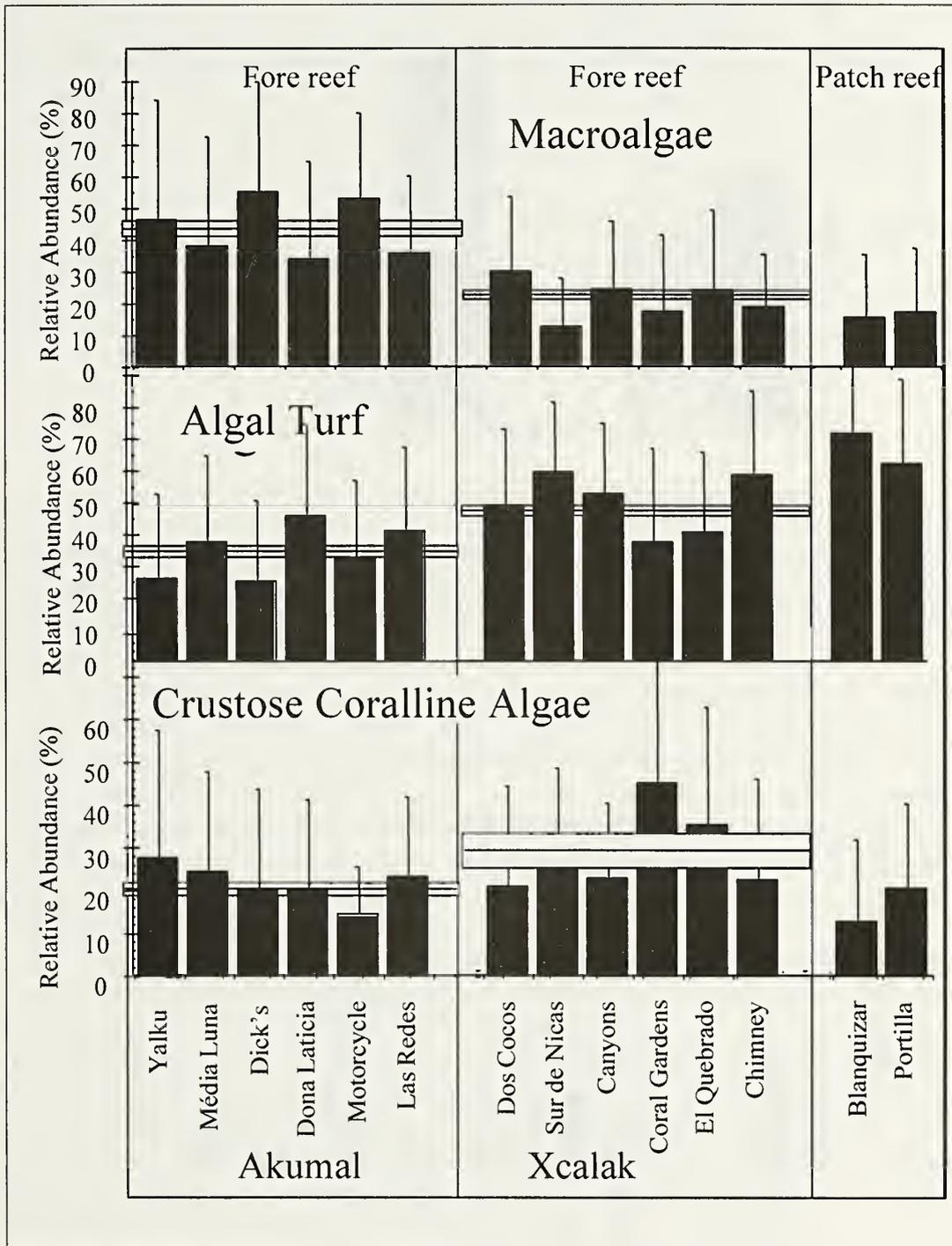
**Figure 9.** Total (recent and old) partial colony mortality as % of the Akumal and Xcalak fore-reef stony corals ( $\geq 25$  cm max. diameter,  $n=669$ ) in March 1999 ordered from most abundant to least abundant species (as in Figure 4). Mean and standard error notation as in Figure 5.

## Algae

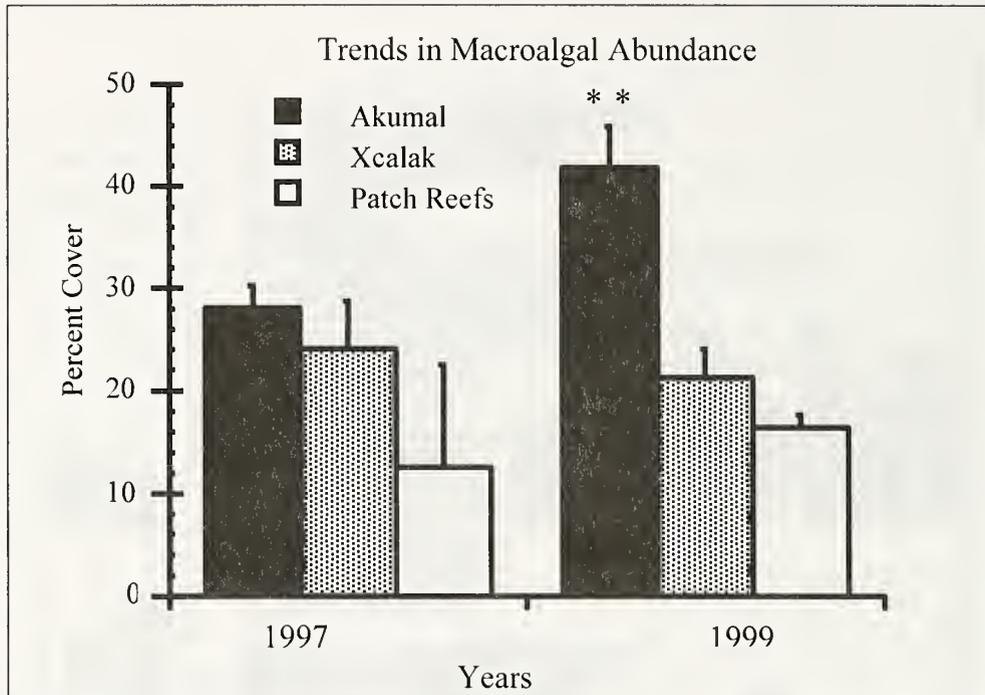
In March 1997, the relative abundance of macroalgae averaged  $\sim 25\%$  in the fore reefs of both regions and  $\sim 12\%$  in the Xcalak patch reefs. Two years later, their relative abundance had significantly increased in the Akumal fore reefs (averaging  $44\%$ ), whereas the reefs at Xcalak were essentially unchanged (Figs. 10, 11; Table 4). Macroalgal heights in 1999 were also slightly higher in Akumal, thus macroalgal indices (an approximation of their biomass, see Table 4) were three to four times greater here than in Xcalak. Turf algae off Akumal showed a pattern opposite that of the macroalgae, having declined from  $62\%$  in 1997 to nearly  $35\%$  in 1999. However, turfs were still the predominant algal group in both habitats off Xcalak. The relative abundance of crustose coralline algae in 1999 varied from  $>15\%$  in the patches to nearly  $30\%$  in the Xcalak fore reefs.

## Fish and *Diadema antillarum*

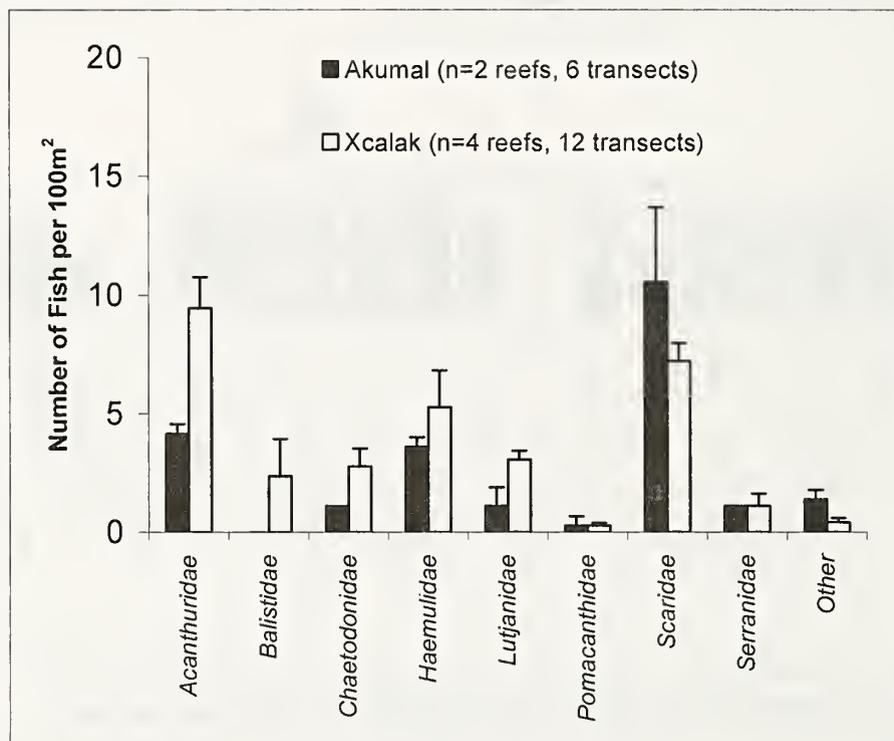
The total abundance of fore-reef fishes surveyed in the belt transects in numbers/100  $m^2$  was slightly higher off Xcalak (mean=34.6, se=3.1) than Akumal (mean=26.1, se=0.8). Most common on the fore reefs in 1999 were herbivores; acanthurids (mean=7.7, se=0.8) were the most abundant in Xcalak whereas scarids



**Figure 10.** Relative abundance (mean  $\pm$  standard deviation) of (A) macroalgae, (B) turf algae, and (C) crustose coralline algae in March 1999 by site for Akumal ( $n=301$  quadrats) and Xcalak ( $n=361$  quadrats in fore reefs and 136 in patch reefs). Mean and standard error notation as in Figure 3.



**Figure 11.** Temporal trends (mean  $\pm$  standard error) in relative abundance of macroalgae in the Akumal and Xcalak fore reefs and the Xcalak patch reefs for March 1997 and March 1999. The increase in abundance in the Akumal fore reefs was the only significant difference (ANOVA,  $p < .0001$ ) between the surveys.



**Figure 12.** Density (mean no. fish/100m<sup>2</sup>  $\pm$  standard error) of AGRRA fishes in the Akumal and Xcalak fore reefs in March 1999. Other = *Bodianus rufus*, *Caranx ruber*, *Microspathodon chrysurus*.

(mean=8.3, se=0.6) were predominant off Akumal (Fig. 12). Parrotfish (scarids) were slightly larger in the southern fore reefs: average lengths of the five species present in belt transects were 16.33 cm (se=1.15) off Xcalak versus 14.64 cm (se=0.99) off Akumal. Large-sized groupers (serranids) were absent, but smaller species such as coneys (*Epinephelus fulvus*), red hinds (*E. guttatus*) and graysbys (*E. cruentatus*) collectively averaged between 1.4 and 3.3/100 m<sup>2</sup> off Akumal and Xcalak, respectively. Bite rates for grazing fishes (acanthurids, scarids, *Microspathodon chrysurus*) averaged 175/m<sup>2</sup>/hour (se=160) at Dick's and Las Redes reefs in Akumal in August 1997; similar results were obtained at two other fore-reef sites in this area.

The long-spined sea urchin, *Diadema antillarum*, was rarely seen in fore-reef habitats and none were counted in any of the transects in March or August 1997. However, a few (3.5/100m<sup>2</sup>) were found in one of the Xcalak patch reefs in March 1999 (Table 4).

## DISCUSSION

*Are coral reefs "dying" throughout the Caribbean?* To address this question, we need baseline information against which comparisons can be made. Our data for reefs in two areas of the Yucatán can contribute to a baseline, but no single study can address this question because each is too limited in its spatial and temporal coverage. Comparisons require combining our results with other AGRRA studies throughout the Caribbean as discussed by Kramer (this volume).

It must also be noted that rapid assessments are little more than "snapshots" of reef condition. We have no way of determining variability in time outside of our survey periods. Although assessments were conducted in the same reefs and during the same weeks of March 1997 and 1999, we cannot say if a month earlier or later the patterns would have remained the same. However, populations of key fish groups at least are unlikely to exhibit short-term increases, particularly given ongoing spearfishing around Akumal (personal observations) and the as-yet unregulated poaching that occurs in Xcalak (S. Redman, personal communication).

Nevertheless, many of the AGRRA indices for 1999 are suggestive that the Yucatán reefs we studied may be in decline. For example, average live stony coral cover overall was <20% (Fig. 3A; Table 1) and, for the Akumal-area fore reefs, was somewhat lower than Munoz-Chagin and de la Cruz-Aguera's (1993) previous *area*-based estimate (18% versus 23%, respectively). Population densities of the AGRRA fishes were low in the seven examined fore reefs (Fig. 12), and the abundance of macroalgae had increased significantly over a period of two years off Akumal (Fig. 11). Mean values of recent and old partial mortality for surveyed corals were extremely high in the two Xcalak patch reefs (Fig. 7).

Partial (recent and old) mortality values for the ≥25 cm stony corals were similar in the fore reefs off Akumal and Xcalak (Fig. 7) in March 1999. Nonetheless, the underlying causes for their mortality may differ since numerous potential stressors and disturbance agents, such as hurricanes, fishing, groundwater input and outbreaks of disease, complicate simple explanations. Below, we discuss how the sudden "pulse"

events of 1998 may have interacted with “chronic” stresses and disturbances in the region from 1997 to 1999 to produce the patterns of mortality we recorded.

#### Recent Pulse Events: Mass Bleaching and Hurricanes of 1998

High temperatures can induce stony corals to bleach, and they will die if the thermal condition persists (e.g., Fitt et al., 2001). Over the past century tropical oceans have been warming which, together with El Niño-related periods of elevated temperatures, have contributed to an increased frequency of coral bleaching (Glynn, 1993; Brown, 1997). The strong El Niño of 1997-1998 resulted in unusually warm sea surface temperatures and widespread lethal bleaching of stony corals worldwide (Wilkinson, 2000). By September 1, an enormous area that was nearly +1.5° C above the average monthly high temperature had developed throughout the western Caribbean (Fig. 2; see also Kramer et al., 2000; Kramer and Kramer, in press). As September is the warmest month in the Caribbean, a warm anomaly at this time is most likely to exceed the thermal thresholds (*sensu* Fitt et al., 2001) of many stony corals. Indeed local divers observed widespread bleaching at Akumal (D. Brewer and S. Slingsby, personal communication) and Xcalak (T. Biller, personal communication).

Subsequent spatial and temporal patterns of sea-surface cooling are consistent with hurricane activity in the area. Hurricanes propagate large-amplitude, low-frequency storm swells that can mix warm surface water layers with deep cooler water. Two hurricanes traversed the western Caribbean in 1998. On September 25, Georges passed over the northwest coast of Cuba, adjacent to the northeast Yucatán coast, as a Category 2 hurricane with sustained winds of 169 km/hour and continued northwest toward the Mississippi delta. The satellite image for October 6 shows an entirely different thermal anomaly pattern from that of the previous month, with most of the western Caribbean north of 19°N latitude, including Akumal, being only about +0.5° C above the monthly average (Fig. 2). This rapid cooling resulted in only transient rather than lethal bleaching over much of the Mexican Caribbean.

Nevertheless, a warm region of nearly +1.5° C persisted throughout the southwestern Caribbean from southern Mexico (including Xcalak) through Belize and Honduras to Colombia. Then Hurricane Mitch developed off the Colombian coast on October 24. Over the next four days it slowly built to a Category 5 hurricane with sustained winds of 290 km/hour as it stalled in the Gulf of Honduras adjacent to Belize. Mitch generated 10 m-high storm swells that crested and broke on the Xcalak reefs (T. Biller, personal communication). Three days later, the November 1 satellite image recorded no thermal anomalies anywhere in the western Caribbean. In fact, the only remaining warm spots were localized off northwestern Cuba, in the southwestern Gulf of Mexico and areas in the ordinarily hurricane-free region from south of Puerto Rico to Venezuela.

That moderately-bleached  $\geq 25$  cm stony corals were more common in March 1999 off Xcalak (both habitats) than in the Akumal fore reefs (Fig. 6) is consistent with the pattern of a longer-duration thermal event in the south having created greater stress and allowed less time for recovery. Reef organisms would have experienced higher temperatures (and presumably higher solar irradiance) in the shallow (1-2 m), calm waters of the Xcalak lagoon than in its offshore fore reefs. Lethal bleaching probably

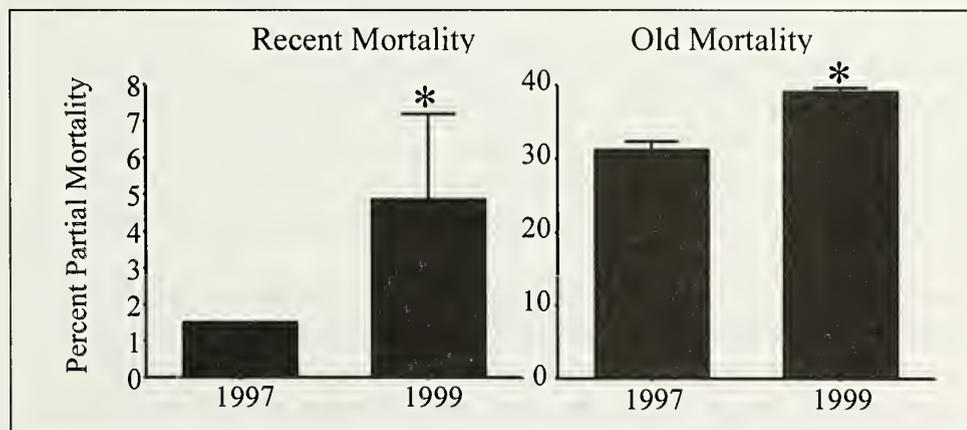
contributed to the extremely high values of recent partial mortality (~27% of colony surfaces) found in the patch reefs, and thereby reduced the percentage of colonies still showing residual levels of light or moderate bleaching in March 1999 to approximate those in the fore reefs nearby (Table 3).

Bleaching-induced mortality of *A. tenuifolia* was first reported in the Yucatán in October 1998 (Kramer et al., 2000). As the transition from what appears to be “recent” to what is scored as “old” probably occurs relatively quickly in species of *Agaricia*, we also suspect that at least some of the comparatively high values for old mortality in the  $\geq 25$  cm fore-reef colonies of this species (Table 2; Fig. 9) had occurred as a result of the 1998 bleaching event.

Mexican reefs sustained relatively little physical damage from hurricanes in 1998, as evidenced by the scarcity of detached and fallen colonies in the March 1999 transects (see also Kramer et al., 2000). That macroalgal blooms were conspicuous off Akumal immediately following Hurricane Mitch (S. Slingsby, personal communication) is consistent with the significant increase in their abundance recorded in the Akumal fore reefs (but not at Xcalak) between 1997 and 1999 (Fig. 11).

#### Chronic Events: Trends and Agents of Mortality

Recent and old partial-colony mortality of the  $\geq 25$  cm stony corals increased significantly between 1997 and 1999 in the two Akumal fore reefs for which commensurable protocols were applied (Fig. 13; T-test,  $p < 0.05$ ). We do not know how long “recently dead” coral skeletons generally remain in that category before being recorded as “old dead” according to criteria in the AGRRA protocol (Appendix One). However, colonies of the abundant *Montastraea annularis* species complex and of *Acropora palmata* frequently retain surficial corallite structures for months after death, particularly when grazing pressures are low (personal observations). Thus we suspect that a majority of the increase in recent mortality recorded in March 1999 was the result of events occurring since the previous summer whereas, apart from *A. tenuifolia*, the higher values of old mortality reflected cumulative impacts of disturbances which had occurred since the 1997 surveys.



**Figure 13.** Temporal trends (mean  $\pm$  standard error) in recent and old partial colony mortality of stony corals (all  $\geq 25$  cm max. diameter;  $n=52$  in August 1997 and 93 in March 1999) in Las Redes and Dick's reefs off Akumal. Comparable data do not exist for March 1997 because of changes in the AGRRA protocol.

Outbreaks of disease (and potential disease states, sensu Richardson, 1998) appear to have increased in diversity and severity throughout the Caribbean in recent decades, and had probably contributed to the above-average levels of partial mortality observed in some species in March 1999 (Fig. 9; Table 2). In particular, the stress of the 1998 bleaching event might have increased the susceptibility of *Montastraea annularis* and related species to white plague. A high percentage of diseased colonies were also reported throughout the Mesoamerican reef system after the 1998 bleaching event, although the incidences (at least in Belize) had decreased by 2000 (Kramer et al., 2000; Kramer and Kramer, in press).

WBD is apparently specific to species of *Acropora* and it spreads relatively slowly (Gladfelter, 1982). A few live *A. palmata* can still be found in fore-reef habitats to depths of about 15 m and we have seen signs of WPD in some of the colonies off Akumal. Thus disease might have caused the above-average levels of both old and recent partial mortality observed in this species in 1999 (Table 2; Fig. 9).

High macroalgal biomass often correlates with mortality of stony corals (Lewis, 1986; Hughes, 1994; Steneck, 1994), and may result in reduced growth (Lewis, 1986; Lirman, 2001) as well as decreased recruitment of juvenile scleractinians (Birkeland, 1977). By having provided substrata for algal colonization and growth, the skeletons of *A. palmata* (Aronson and Precht, 2000) and other corals killed by disease or lethal bleaching (such as the *M. annularis* species complex and *A. tenuifolia*, respectively) may have facilitated the expansion of macroalgae in the Yucatán's fore reefs. In turn, overgrowth of live coral polyps by macroalgae may have contributed to the relatively high values of recent mortality found in the Akumal population of *A. palmata*.

Proliferations of macroalgae can also occur as a result of increases in nutrients (e.g., Lapointe, 1997) and/or declines in herbivory (e.g., Hughes, 1994; Steneck 1994). Hasty development of coastal infrastructures, locally high human populations, inadequate sewage treatment (especially in the pueblos), and the hydrography of the Caribbean coast of México make it particularly vulnerable to nutrient enrichment. Groundwater discharged from a ramified underground river complex of cenotes permeates much of the northern Yucatán coast. At Akumal, where discharges are commonly observed in the deep, shelf-edge reefs (D. Brewer, personal communication), nutrients in groundwater potentially pose an elevated risk because of the close proximity of the reefs to the shore.

Although herbivorous acanthurids and scarids were among the more common fishes quantified in the fore-reefs off Akumal and Xcalak (Fig. 12), their population densities were low (combined means of 16/100m<sup>2</sup>) compared to many other areas in the wider Caribbean (Kramer, this volume). The 1997 fish herbivory rates off Akumal resembled those found at 10 m depths in 1988 on the severely overfished reefs of Jamaica, but were less than 10% the rates which had been obtained that year in St. Croix (Steneck, 1994).

The key herbivore, *Diadema antillarum*, was very rare, particularly in the fore reefs (Table 4), and other grazing echinoids, such as *Echinometra viridis* and *Tripneustes ventricosus* (e.g., Aronson and Precht, 2001; Edmunds and Carpenter, 2001), were notably scarce (personal observations). Hence, low levels of herbivory may well have contributed to the high relative abundance and biomass of macroalgae, particularly off Akumal.

As populations of *Diadema* have begun to recover in some Caribbean reefs, macroalgae are becoming less abundant (personal observations). Edmunds and Carpenter (2001) have recently reported an upsurge in coral recruitment at shallow (<9m) fore reefs near Discovery Bay, Jamaica in which densities of *Diadema* had increased. Since 1999, populations of *Diadema* have similarly rebounded in lagoonal patch reefs off both Akumal (R. Roy, personal communication) and Xcalak (S. Redman, personal communication). Should similar increases occur in the Akumal fore reefs, the build-up seen in its macroalgal populations might be reversed at least partially.

### ACKNOWLEDGMENTS

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Table 1. Site and transect information for AGRRA surveys in March 1999 off Akumal and Xcalak, México.

Site name	Reef type	Latitude (°' N) <sup>1</sup>	Longitude (°' W) <sup>1</sup>	Survey date	Depth (m)	Benthic transects (#)	≥25 cm stony corals (#/10m)	% live stony coral cover (mean ± sd)
<b>Akumal</b>								
Yalkul	Fringing-barrier fore reef	20 24.5	87 18.0	Mar 12 99	11	12	6.5	21.0 ± 9.0
Mérida Luna	Fringing-barrier fore reef	20 21.1	87 18.2	Mar 12 99	11.5	14	4.5	13.5 ± 8.0
Dick's	Fringing-barrier fore reef	20 24.0	87 18.4	Mar 12 99	11.5	10	6.5	19.0 ± 8.0
Doña Laticia	Fringing-barrier fore reef	20 23.8	87 18.4	Mar 12 99	13	12	6.5	18.0 ± 10.5
Motorcyle	Fringing-barrier fore reef	20 23.5	87 18.5	Mar 8 99	13	7	3.5	15.5 ± 5.5
Las Redes	Fringing-barrier fore reef	20 23.5	87 18.6	Mar 8 99	18	7	4	17.0 ± 9.0
<b>Xcalak</b>								
Dos Cocos	Fringing-barrier fore reef	18 19.7		Mar 10 99	13	10	4.5	9.5 ± 5.5
Sur de Nicas	Fringing-barrier fore reef	18 16.9	87 49.0	Mar 9 99	16.5	12	6.5	20.0 ± 10.0
Canyons	Fringing-barrier fore reef	18 15.5	87 49.1	Mar 9 99	15	17	2.5	8.5 ± 4.0
Coral Gardens	Fringing-barrier fore reef	18 14.4	87 49.6	Mar 10 99	8.5	10	6	14.0 ± 6.0
El Quebrado	Fringing-barrier fore reef			Mar 10 99	11	12	6	37.0 ± 20.5
Chimney	Fringing-barrier fore reef	18 13.3	87 52.3	Mar 10 99	11.5	16	2	9.5 ± 9.5
Blanquezar	Lagoonal patch reef	18 19.5	87 48.9	Mar 11 99	1.5	15	7.5	20.5 ± 11.0
Portillas	Lagoonal patch reef	18 14.1	87 49.9	Mar 9 99	3	16	5	16.0 ± 12.5

<sup>1</sup>Missing locations resulted from GPS failures.

Table 2. Abundance, size and mortality of large ( $\geq 25$  cm diameter) reef-building corals in March 1999, by species in fore reefs off Akumal and Xcalak.

Species	Colonies		Density (#/m)	Maximum diameter (cm)	% Live cover	Partial-colony mortality (%)		Total
	(#)	%				Recent	Old	
<i>Acropora palmata</i>	43	5.55	0.031	205	6.36	4.5	54.0	58.5
<i>Agaricia agaricites</i>	28	3.90	0.019	45	0.86	1.5	19.0	20.5
<i>Agaricia tenuifolia</i>	35	4.53	0.025	70	1.75	3.5	42.0	45.5
<i>Colpophyllia natans</i>	27	3.82	0.019	55	1.03	4.0	18.0	22.0
<i>Colpophyllia breviserialis</i>	2	0.36	0.001	85	0.09	0	5.0	5.0
<i>Diploria clivosa</i>	4	0.65	0.003	35	0.11	3.0	19.5	22.5
<i>Diploria labyrinthiformis</i>	16	2.63	0.013	35	0.48	4.0	14.0	18.0
<i>Diploria strigosa</i>	26	4.04	0.020	55	1.08	1.5	22.0	23.5
<i>Madracis mirabilis</i>	3	0.44	0.003	30	0.09	3.5	35.0	38.5
<i>Millepora complanata</i>	4	0.59	0.003	70	0.21	0	2.5	2.5
<i>Montastraea annularis</i>	188	28.94	0.139	65	9.04	6.0	34.5	40.5
<i>Montastraea cavernosa</i>	36	5.68	0.026	35	0.91	1.0	15.0	16.0
<i>Montastraea faveolata</i>	136	20.34	0.104	95	9.67	5.0	27.5	32.5
<i>Montastraea franksi</i>	33	5.17	0.026	95	2.47	2.5	28.0	30.5
<i>Porites astreoides</i>	25	3.58	0.019	30	0.61	3.5	12.5	16.0
<i>Porites furcata</i>	2	0.22	0.001	30	0.03	0	7.5	7.5
<i>Porites porities</i>	11	1.48	0.009	30	0.28	2.5	24.0	26.5
<i>Siderastrea siderea</i>	32	6.36	0.026	40	1.04	4.0	15.0	19.0
<b>Mean <math>\pm</math> standard error</b>	<b>36.0 <math>\pm</math> 48.5</b>	<b>5.5 <math>\pm</math> 7.5</b>	<b>0.025 <math>\pm</math> 0.035</b>	<b>60 <math>\pm</math> 40</b>	<b>2.0 <math>\pm</math> 3.0</b>	<b>3.0 <math>\pm</math> 2.0</b>	<b>22.0 <math>\pm</math> 13.0</b>	<b>24.5 <math>\pm</math> 14.5</b>

Table 3. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 25$  cm diameter) in March 1999, by site off Akumal and Xcalak.

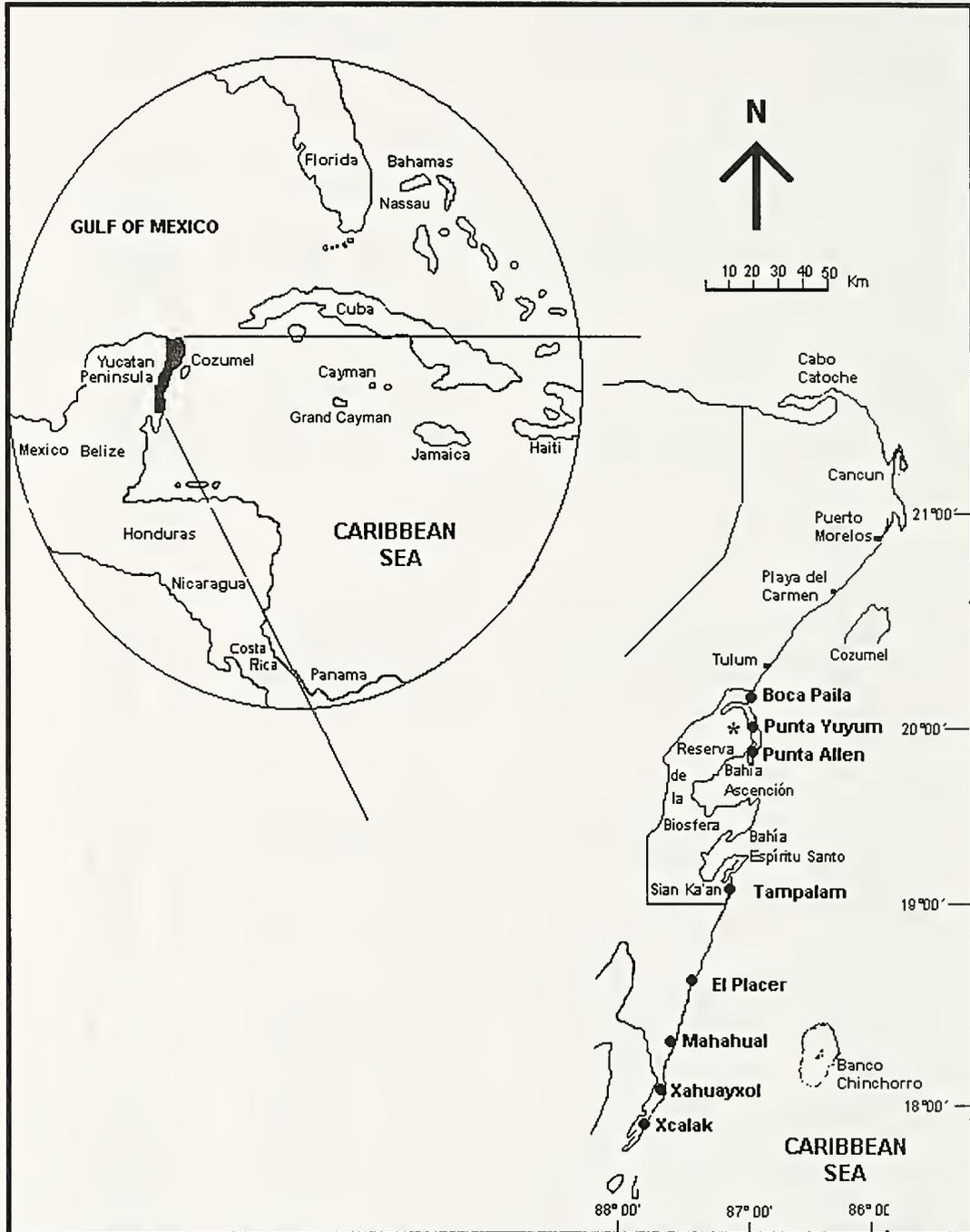
Site name	Stony corals		Partial-colony mortality (%)			Standing dead	Stony corals (%)		Diseased <sup>1</sup>
	(#)	Diameter (cm)	Recent	Old	Total		Light	Moderate	
<b>Akumal</b>									
<i>Fore reefs</i>									
Yalku	79	100 $\pm$ 130	6.5 $\pm$ 14.5	29.5 $\pm$ 30.5	35.0 $\pm$ 33.0	6	9	6	6
Mé dia Luna	63	110 $\pm$ 255	4.0 $\pm$ 7.5	27.5 $\pm$ 30.5	31.0 $\pm$ 32.0	8	14	0	6
Dick's	66	110 $\pm$ 85	8.5 $\pm$ 21.5	37.5 $\pm$ 33.5	43.5 $\pm$ 35.5	15	8	12	4
Doña Laticia	75	75 $\pm$ 80	4.5 $\pm$ 10.5	26.0 $\pm$ 26.0	31.0 $\pm$ 31.0	2	14	2	11
Motorcyle	26	70 $\pm$ 50	0.5 $\pm$ 1.0	24.5 $\pm$ 19.0	24.5 $\pm$ 19.0	0	4	0	0
Los Redes	27	70 $\pm$ 40	2.5 $\pm$ 5.0	23.5 $\pm$ 19.0	25.5 $\pm$ 20.0	0	8	0	4
<b>Xcalak</b>									
<i>Fore reefs</i>									
Sur de Nichas	76	45 $\pm$ 25	4.0 $\pm$ 8.5	17.5 $\pm$ 17.0	21.5 $\pm$ 20.0	0	6	0	2
Canyons	46	50 $\pm$ 35	1.0 $\pm$ 3.0	40.5 $\pm$ 37.5	41.5 $\pm$ 37.5	6	18	11	2
Coral Gardens	60	85 $\pm$ 100	8.5 $\pm$ 15.5	29.0 $\pm$ 33.5	37.5 $\pm$ 37.0	8	10	14	4
Chimney	35	55 $\pm$ 30	5.5 $\pm$ 11.5	21.5 $\pm$ 26.0	26.5 $\pm$ 27.5	6	6	20	0
El Quebrado	72	95 $\pm$ 90	6.0 $\pm$ 10.5	30.5 $\pm$ 27.5	36.5 $\pm$ 29.5	4	12	11	0
Dos Cocos	44	45 $\pm$ 40	4.0 $\pm$ 7.5	31.5 $\pm$ 28.0	35.5 $\pm$ 29.0	0	9.0	14	0
<b>Pauch Reefs</b>									
Blanquezar	113	140 $\pm$ 115	29.0 $\pm$ 22.0	39.5 $\pm$ 27.0	67.5 $\pm$ 25.5	12	12	12	10
Portillas	83	160 $\pm$ 120	26.5 $\pm$ 25.0	41.5 $\pm$ 26.5	67.0 $\pm$ 25.0	6	7	8	12

<sup>1</sup>All diseased corals had white plague, except for one colony at Doña Laticia which had black-band disease.

Table 4. Algal characteristics, and density of *Diadema antillarum* (mean  $\pm$  standard deviation) in March 1999, by site off Akumal and Xcalak.

Site name	Quadrats (#)	Relative abundance (%)			Macroalgal		<i>Diadema</i> (#/100m <sup>2</sup> )
		Macroalgae	Turf algae	Crustose coralline algae	Height	Index <sup>1</sup>	
<b>Akumal</b>							
<b>Fore reefs</b>							
Yalku	63	46.5 $\pm$ 38.0	26.0 $\pm$ 27.0	27.5 $\pm$ 30.0	2.0 $\pm$ 1.5	93	0
Média Luna	68	38.0 $\pm$ 34.5	37.5 $\pm$ 27.0	24.5 $\pm$ 23.5	2.0 $\pm$ 2.0	76	0
Dick's	45	55.0 $\pm$ 35.0	25.0 $\pm$ 25.5	20.0 $\pm$ 24.0	3.0 $\pm$ 1.5	165	0
Doña Laticia	59	34.0 $\pm$ 31.0	45.5 $\pm$ 29.0	20.5 $\pm$ 21.0	2.0 $\pm$ 1.5	68	0
Motoreyle	33	53.0 $\pm$ 27.0	32.5 $\pm$ 24.5	14.5 $\pm$ 11.5	2.5 $\pm$ 1.5	132.5	0
Los Redes	33	36.0 $\pm$ 24.5	41.0 $\pm$ 26.5	23.0 $\pm$ 18.5	2.5 $\pm$ 1.5	90	0
<b>Xcalak</b>							
<b>Fore reefs</b>							
Sur de Nichas	55	13.0 $\pm$ 15.0	59.5 $\pm$ 22.0	28.0 $\pm$ 21.0	1.5 $\pm$ 1.5	19.5	0
Canyons	85	24.5 $\pm$ 21.5	52.5 $\pm$ 22.5	23.0 $\pm$ 17.5	2.0 $\pm$ 1.5	49	0
Coral Gardens	40	17.5 $\pm$ 24.5	37.5 $\pm$ 29.5	45.0 $\pm$ 33.0	0.5 $\pm$ 1.0	9.0	0
Chimney	79	19.0 $\pm$ 16.5	58.5 $\pm$ 26.5	22.5 $\pm$ 23.5	1.5 $\pm$ 1.5	28.5	0
El Quebrado	57	24.0 $\pm$ 25.5	40.5 $\pm$ 25.5	35.5 $\pm$ 27.5	2.0 $\pm$ 2.0	48	0
Dos Cocos	45	30.0 $\pm$ 24.0	49.0 $\pm$ 24.0	21.0 $\pm$ 23.5	1.5 $\pm$ 1.0	45	0
<b>Patch reefs</b>							
Blanquezar	62	15.5 $\pm$ 20.0	71.5 $\pm$ 26.0	12.5 $\pm$ 19.0	1.0 $\pm$ 1.5	15.5	3.5
Portillas	74	17.5 $\pm$ 20.5	62.0 $\pm$ 26.5	20.5 $\pm$ 20.0	2.0 $\pm$ 2.0	35	0

<sup>1</sup>Macroalgal index = % relative macroalgal abundance x macroalgal height



**Figure 1.** AGRRA survey reefs in central-southern Quintana Roo, México.  
 \* = Location of the Campechen, Boca Paila and San Miguel Lagoons. Modified from Núñez-Lara and Arias-González (1998).

# CONDITION OF CORAL REEF ECOSYSTEMS IN CENTRAL-SOUTHERN QUINTANA ROO, MEXICO (PART 1: STONY CORALS AND ALGAE)

BY

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## ABSTRACT

In 1999 *Montastraea faveolata* and *M. annularis* were the most numerous “large” ( $\geq 25$  cm diameter) stony corals at  $\sim 10$  m on fore reefs in the central and southern areas, respectively, of Quintana Roo, México. Reductions in live stony coral cover (from  $\sim 25\%$  to  $\sim 12\%$  in  $< 10$  years) and high recent partial-colony mortality (7-27.5%) are indications of declining reef conditions. Diseases in the five more northerly reefs, as well as tissue loss from the 1998 El Niño Southern Oscillation (ENSO) bleaching event and/or from on-going bleaching during June-July 1999 in the three southernmost reefs, appeared responsible for much of the recent mortality. Although turf algae predominated everywhere, macroalgae were relatively more abundant in the five more northerly reefs (four of which are in a reserve where herbivorous fishes currently are less numerous than further south). Additional perturbations associated with tourism development in the southern area could result in a loss of resilience of these coastal reefs.

## INTRODUCTION

The greatest reef development in México is found in its Caribbean reefs which are principally of the fringing type. Extending for over 350 km along the coast of the state of Quintana Roo (Gutiérrez et al., 1995), they comprise the northernmost part of the Mesoamerican Reef System which continues through the neighboring countries of Belize, Guatemala and Honduras.

Tourism has become the principal economic activity along the Quintana Roo coast representing  $\sim 75\%$  of the state's 1999 Gross Internal Product (INEGI, 2000). At

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present this activity is concentrated throughout the northern coastal area, such as at Isla Mujeres, Cozumel, and along the Cancún-Tulum Corridor. However, tourism infrastructure is either planned or developing in some southern areas of the state, particularly between Pulticub and Xcalak (Daltabuit Godás, 1999). Specific threats associated with this development include ecologically inappropriate forms of coastal development, diving, pollution from solid and liquid wastes, vessel traffic, and fishing (Daltabuit Godás, 1999).

Another important aspect of the Mexican Caribbean is that a number of protected areas have been created with a view towards direct or indirect protection of its coral-reef ecosystems. The largest among these is the Sian Ka'an Biosphere Reserve established in 1986 between 19°05' N and 20°06' N in the central area of the coast. This reserve is protected by regulations restricting commercial fishing activities, aquatic sports, and development.

Hence coral reefs are severely threatened by intense tourism activity in the northern area, protected in the central area, and potentially at risk in the south. Consequently, reef "health" within each area of the Mexican Caribbean may vary as a function of these differing levels of anthropogenic input. The aim of this study was to evaluate benthic reef condition in the spur-and-groove habitat of the fore-reef slope zone in the central and southern areas using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) methodology.

## METHODS

The study area extends along approximately 230 km of the Quintana Roo coast (Fig. 1) and includes eight fringing reefs, four of which are located in the central area within the Sian Ka'an Biosphere Reserve. On the basis of morphology and degree of development, the Quintana Roo reef system has been divided into 16 subregions (ICRI, 1998). Three reefs (Boca Paila, Punta Yuyum, Punta Allen) are in the Sian Ka'an North subregion which is characterized as having a narrow continental shelf with steep slopes, "intermediate amounts" of freshwater input, narrow to "moderately wide" lagoons and, generally, highly developed fore-reef spurs and grooves. Coral cover averaged nearly 25% in the early 1990's when 42 species of scleractinian corals were found in this subregion (Gutierrez et al., 1993; ICRI 1998). These three reefs are distinguished from the remainder in having submerged crests and lagoons that are open to the ocean, and by their close proximity to coastal bays (Ascensión and Espíritu Santo) and lagoons (Campeche, Boca Paila and San Miguel).

Reef structures are more highly developed and complex overall on the five southernmost reefs (one of which, Tampalam, is located within the Biosphere Reserve in the central geographic area). The extreme northern portion of their reef crests runs into the shore, thereby enclosing and protecting large seagrass meadows landward of the reef flats. Tampalam and El Placer in the Sian Ka'an South subregion have narrow reef crests and slightly lower values for coral cover and scleractinian species number (22% and 41, respectively). Two reefs (Mahahual and Xahuayxol) in the Mahahual subregion, which are in the "shadow" of an offshore bank (Banco Chinchorro), have particularly well-developed fore-reef terraces with high-relief spurs and grooves, and back reefs that

extend almost to the beach. The highest values for coral cover (nearly 28%) and species richness (46) occurred in this subregion. Xcalak, in the Xcalak Trench subregion, has a well-developed lagoon and breaker zone and well-developed spurs and grooves on the fore reef and is somewhat under the shadow of Banco Chinchorro. Coral cover here was nearly 25% but there were only 33 species of corals in the early 1990s.

A characteristic of all the studied reefs is that the fore reef develops as a series of “strips” (spurs and grooves) that run parallel to the coast and are usually separated from each other by intervening sand channels. Hence it is possible to differentiate between an inner fore reef (at depths ranging from 6 m to as much as 25 m), an outer fore reef (from ~15 m to as deep as 40 m), and a deep fore reef (from ~35 to ~50 m) terminating in a shelf-edge reef at the continental margin (ICRI, 1998).

It has been suggested that the Mexican Caribbean reef system has been molded by the high frequency of tropical storms and hurricanes crossing this area, particularly near the Belizean border in the southern geographic area and between Cozumel and the Yucatán Channel in the northern area (ICRI, 1998). The last significant storm that passed near our study areas was Hurricane Mitch in October 1998; nevertheless, there are just three descriptions of its effects at Mahahual (Garza-Pérez, 1999; Garza-Pérez et al., 2000; Bastida-Zavala, 2000). Similarly, there is only one formal report of the 1998 mass bleaching event, which is an important recent disturbance. Its effects were documented at Xcalak, Mahahual, Tampalam, and further north at Akumal by Garza-Pérez (1999), who continued to assess the condition of the Mahahual reef through 2000 (Garza-Pérez, in preparation).

This study was conducted in the fore-reef spur-and-groove habitat at an average depth of 10 m where stony coral cover in central-southern Quintana Roo is highest (Gutiérrez et al., 1993; Gutiérrez et al., 1995). We did not survey the outer-reef crest, another important habitat, for logistical reasons and because it is not well-developed on all the study reefs.

We established a hierarchical survey design based on three spatial scales: geographic area (central and southern), reefs (four in each area), and subreefs (three per reef). Reefs that were known to exhibit maximum spur-and-groove development were chosen on the basis of previous field work and from published descriptions (Gutiérrez et al., 1993; Jordán, 1993) and were thus selected on the basis of both strategic and representative criteria (see Appendix One). The distance between reefs, which were named after the nearest town or coastal feature (Fig. 1), was approximately 20-30 km, except for Tampalam which is ~75 km south of Punta Allen. To avoid pseudoreplication (i.e., Oxley 1997), the north (N), center (C) and south (S) subreef for each reef were positioned 0.9-1 km apart. This separation distance was chosen to prevent large geomorphological changes between the subreefs of any given reef. Their geographical coordinates were established by GPS (Table 1). Borders delimiting each subreef were temporarily established within which surveys were performed on haphazardly positioned transects over the spur structures in the inner spur-and-groove zones.

Fieldwork was carried out by two observers (Ruiz-Zárate on the transects and Hernández-Landa on the quadrats) between June and October 1999 (Table 1), using the AGRRA Version 2.2 benthos protocols (see Appendix One, this volume). The sizes of the individually surveyed stony corals (scleractinians and *Millepora* spp.) were measured to the nearest cm. Crustose coralline algae were exposed by vigorous removal of

sediment and pushing apart any macroalgae and species of scleractinians that are small as adults were omitted from the counts of tiny (<2 cm diameter) corals. Training exercises were conducted in the field prior to performing the surveys. Humman's (1993a, 1998b) field guides aided in the identification of organisms.

Averages were computed for each spatial scale (subreef, reef, and area). Correlation and regression analyses were then applied to find statistical tendencies.

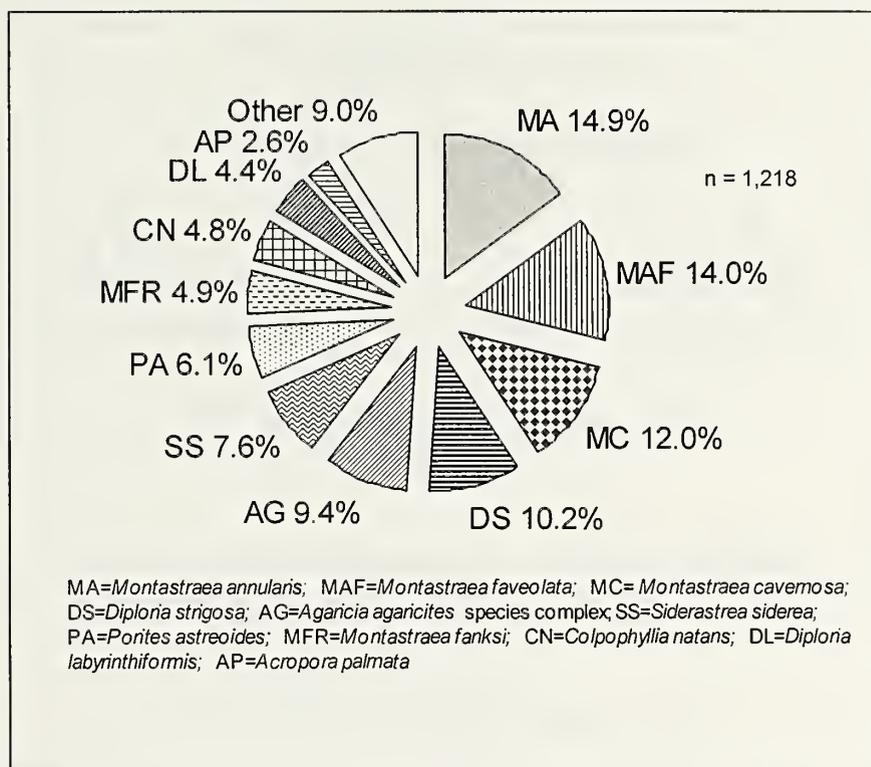
## RESULTS

### Stony Corals

Four morphological forms of *Agaricia agaricites* (*A. agaricites agaricites*, *A. agaricites carinata*, *A. agaricites danai*, *A. agaricites purpurea*) that were recognized in this study have not previously been discriminated in the Mexican Caribbean and, for matters of comparison, they were considered as a single species (*Agaricia agaricites*). A total of 24 species of "large" stony corals (colonies  $\geq 25$  cm maximum diameter) were recorded in our transects, representing nearly 60% of the 47 species listed in central-southern Quintana Roo by Beltrán-Torres and Carricart-Ganivet (1999). Of these 47 species, 37 can be expected to occur in depths of about 10 m (Cuadro 2; Beltrán-Torres and Carricart-Ganivet, 1999; personal observations). Twenty-six are reported to grow larger than 25 cm in diameter (after Humann, 1993), seven to be smaller than 25 cm, and three of the four species for which no sizes were given (*Porites furcata*, *Montastraea faveolata*, *M. franksi*) are well known to exceed 25 cm in diameter while the fourth, *Porites divaricata*, is usually small. Included in our surveys were 23 (79 %) of the 29 large species known to occur in this habitat and one species (*Madracis decactis*) that Humann (1993) had characterized as being less than 25 cm in diameter.

Four species numerically constituted 50% of the total: *Montastraea annularis* > *M. faveolata* > *M. cavernosa* > *Diploria strigosa* (Fig. 2). Species with relative frequencies of 70-100% (i.e., recorded at 17-24 of the subreefs), in order of decreasing frequency, were *D. strigosa*, *M. faveolata*, *M. cavernosa*, *M. annularis*, *Siderastrea siderea*, *Colpophyllia natans*, *D. labyrinthiformis*, *Porites astreoides* and *M. franksi*.

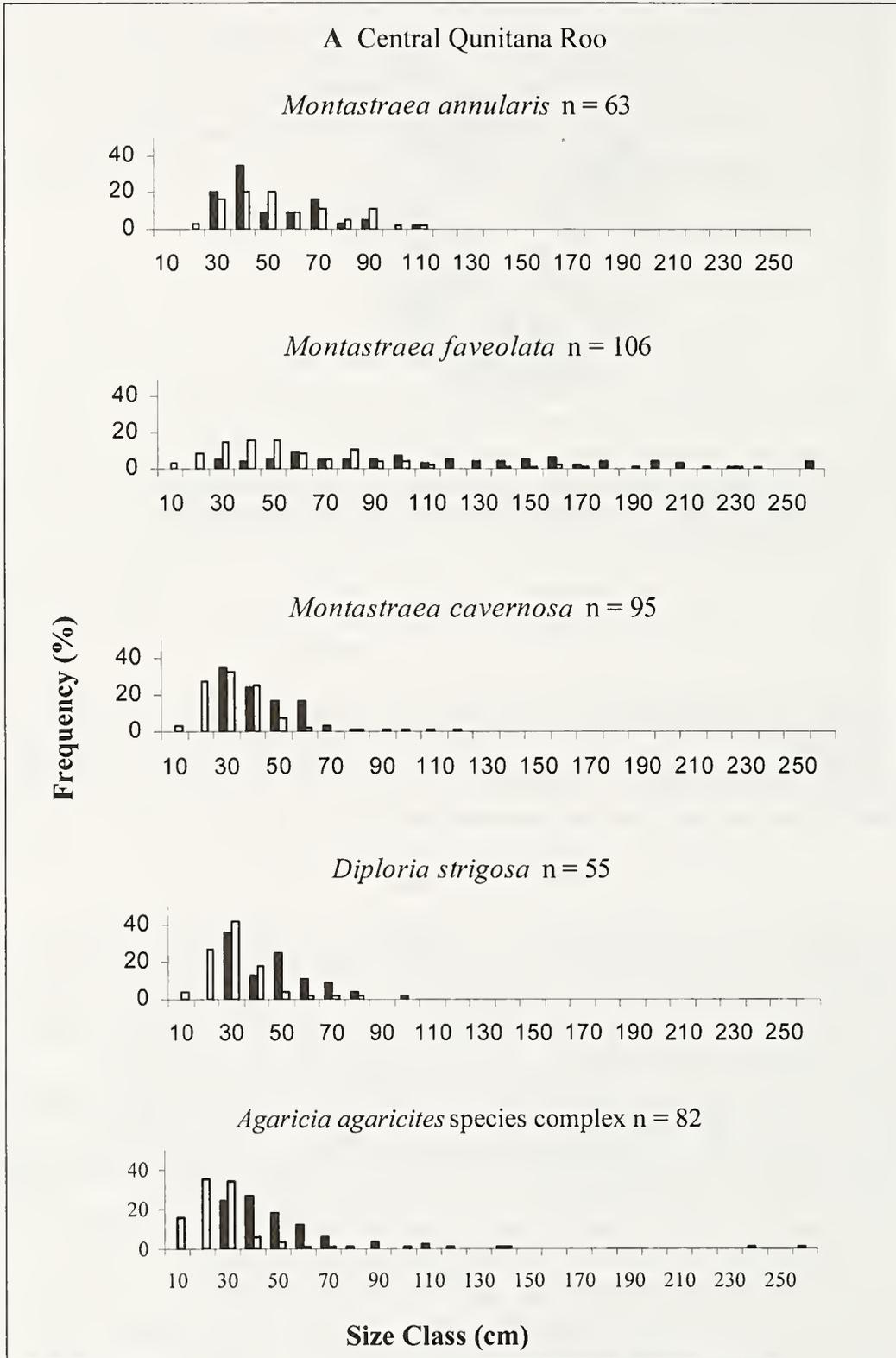
The two reefs (Xcalak, El Placer) with the lowest mean live stony coral cover (~9%), and Mahahual, with the highest mean cover (~17%), were all in the southern area (Table 1). Mean live stony coral cover for the four reefs in the central area was virtually identical to that of the four southern reefs and averaged 12.0% overall (sd = 6.2, n = 311 transects). The mean density of large ( $\geq 25$  cm diameter) stony corals varied from 1.5-6.0/10 m transect.



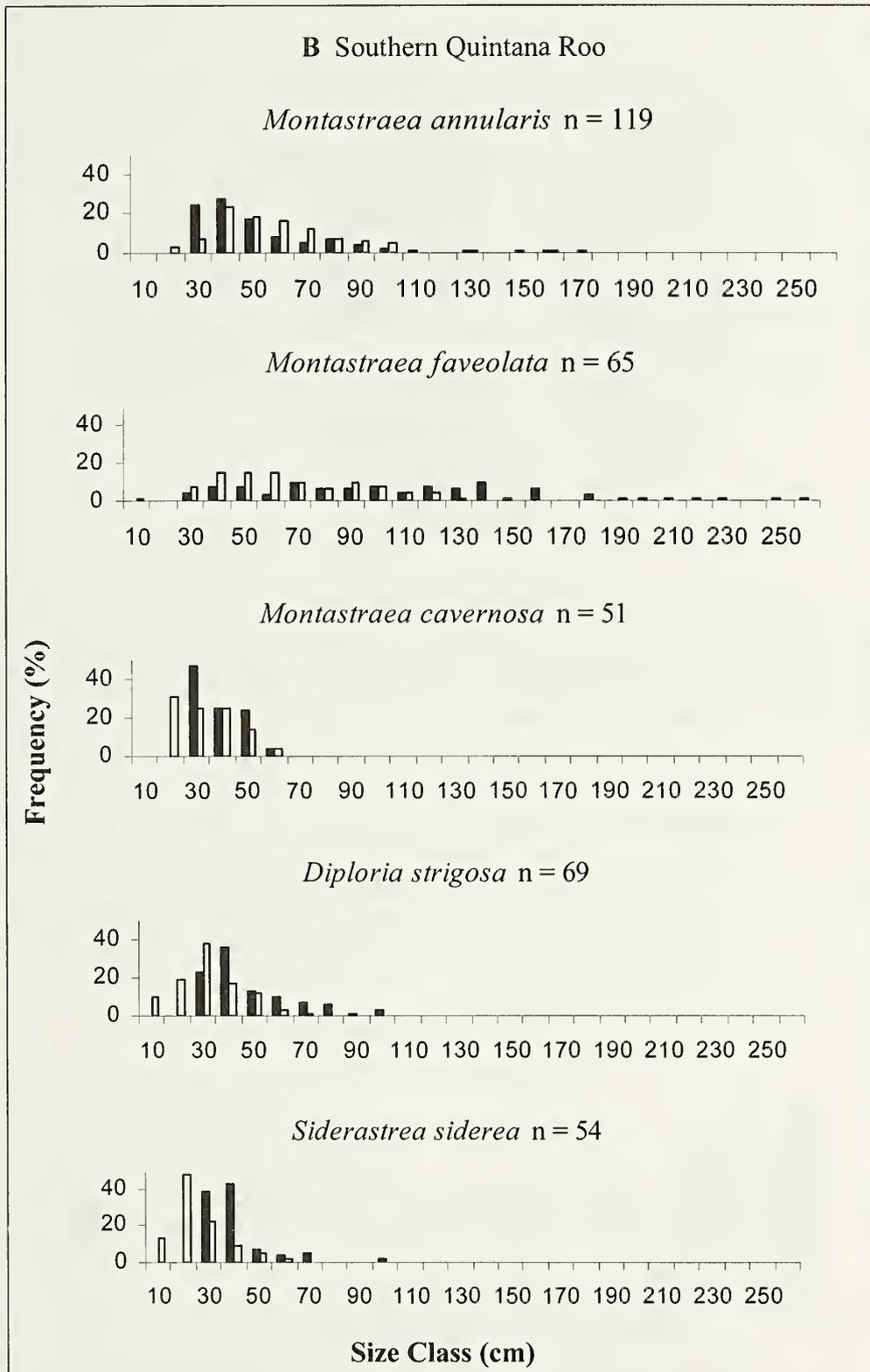
**Figure 2.** Species composition and mean relative abundance of the most abundant stony corals ( $\geq 25$  cm diameter) at 8-12 m on fore reefs in central-southern Quintana Roo, México. Other = *Agaricia tenuifolia*, *Meandrina meandrites*, *Porites furcata*, *P. porites*, *Acropora cervicornis*, *Madracis mirabilis*, *M. decactis*, *Millepora complanata*, *M. alcicornis*, *Mycetophyllia ferox*, *M. danaana*, *Dendrogyra cylindrus*, *Eusmilia fastigiata*, and standing dead colonies of *Diploria*, *Montastraea* and *Agaricia*.

The diameters of large stony corals were smallest in El Placer (~ 48 cm) and largest in Boca Paila (~ 65 cm) (Table 2). Stony corals were slightly larger overall in the central area than in the southern area; the mean diameter for both areas combined was 56.3 cm (sd = 40.6, n = 1,218 corals). The size-frequency distributions, both as maximum diameter and as maximum height, of the five most abundant large species in each geographic area are summarized in Figures 3A, B. Colonies of *M. faveolata*, which were more abundant in reefs of the central area, constituted the largest corals in both areas. Apart from *M. faveolata*, most of the surveyed colonies were in small to intermediate size classes (30-50 cm for maximum diameter, 10-60 cm for maximum height).

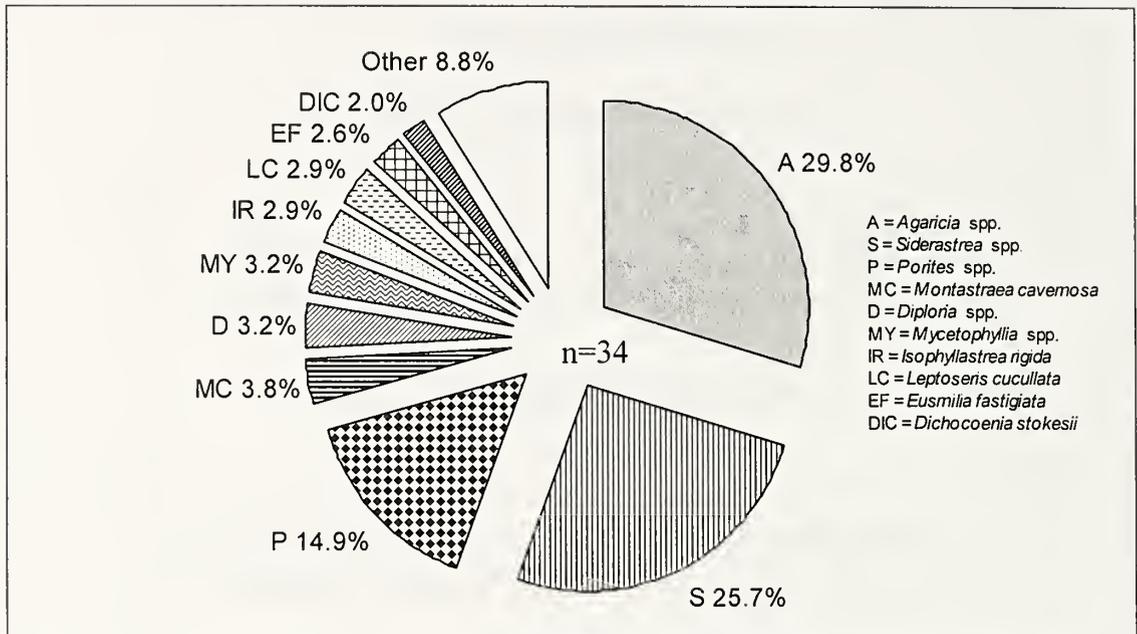
The mean density of stony coral recruits in the different reefs varied from 0.1-0.4/0.0625 m<sup>2</sup>, averaging 0.2 /0.0625 m<sup>2</sup> in both geographic areas (Table 3). In many cases, it was not possible to identify the coral recruits to the species level. The three most important genera, in order of relative abundance, were *Agaricia*, *Siderastrea* and *Porites* (Fig. 4). Only one individual of the herbivorous echinoid *Diadema antillarum* was found in a belt transect in one subreef (Tampalam N).



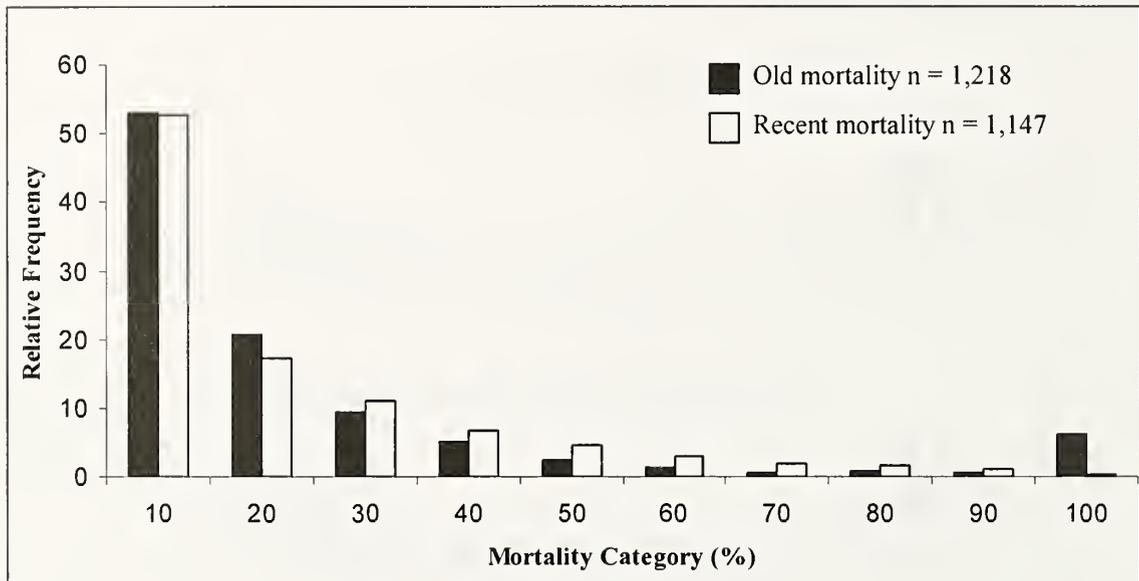
**Figure 3A.** Size-frequency distribution as diameter (dark bars) and height (white bars) of colonies ( $\geq 25$  cm diameter) at 8-12 m on fore reefs in central Quintana Roo, México.



**Figure 3B.** Size-frequency distribution as diameter (dark bars) and height (white bars) of colonies ( $\geq 25$  cm diameter) at 8-12 m on fore reefs in southern Quintana Roo, México.



**Figure 4.** Species composition and mean relative abundance of all stony coral recruits ( $\geq 2$ cm diameter, excluding species that are small as adults) at 8-12 m on fore reefs in the central and southern Mexican Caribbean. Other = *Stephanocoenia intersepta*, *Montastraea* spp., *Madracis* spp., *Meandrina meandrites*, *Manicina areolata*, *Acropora cervicornis*, *Isophyllia sinuosa*, *Colpophyllia natans*, *Millepora alcicornis* and a tiny unknown scleractinian.



**Figure 5.** Frequency distribution as % of old partial colony mortality (black bars) and recent partial colony mortality (white bars) of all stony corals ( $\geq 25$  cm diameter) at 8-12 m on fore reefs in central-southern Quintana Roo, México.

## Coral Condition

Recent partial-colony mortality (hereafter recent mortality) was relatively high, exceeding 15% in 70% (17/24) of the subreefs, with only one reef each in the central (Tampalam) and southern (Xcalak) areas having mean values of <15%. Levels of recent mortality in the two areas were very similar, and the overall mean was 18.0% (sd = 20.0, n = 1146 stony corals).

The two reefs (Mahahual, Xahuayxol) with the lowest mean levels of old partial-colony mortality (hereafter old mortality), as well as the reef with the highest mean old mortality (Xcalak) were all in the southern area. Mean values of old mortality overlapped at the area scale and the overall mean of 19.3% (sd. = 25.4, n = 1218 stony corals) was close to that for recent mortality. Nonetheless, the most frequent percentage class category for both recent and old coral mortality was 0-10% (Fig. 5).

Total (recent + old) partial-colony mortality in the reefs ranged between about 30% at Xahuayxol in the southern area to 43% at Punta Yuyum in the central area. The percentage of "standing dead" stony corals (i.e., 100% mortality of upward facing surfaces of colonies still in the original position of growth) exhibited two pronounced peaks: between Punta Yuyum and Punta Allen in the central area (6-16% of all colonies, n=6 subreefs), and at Xcalak in the southern area (12-24% of colonies, n = 3 subreefs). *Acropora palmata* accounted for about half to two-thirds of the standing dead corals in Punta Allen (10/18) and Xcalak (13/19) but *Diploria* represented half (10/19) of the standing dead colonies in Punta Yuyum. Corresponding values for standing dead on the remaining subreefs ranged from 0-6% (Table 2).

Except for two subreefs in the central area (Punta Allen N, Tampalam N), bleaching was restricted to the southern geographic area, principally between Mahahual N and Xahuayxol C (Fig. 6). "Partly bleached" was the most common condition in these five subreefs affecting 18-35% of the large stony corals. Only in Mahahual, where the highest percentages of stony corals with bleached tissues were found, were all three bleaching categories (pale, partly bleached, bleached) recorded (Fig. 6).

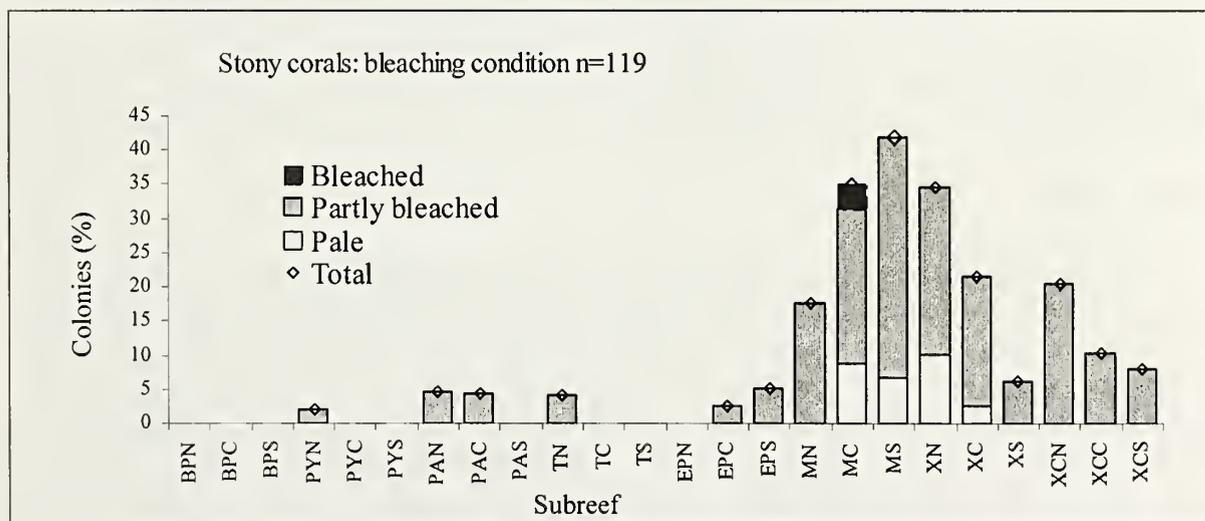


Figure 6. Percentages of all stony corals ( $\geq 25$  cm diameter) affected by bleaching (as pale, partly bleached, bleached) at 8-12 m on fore reefs in central-southern Quintana Roo, México.

Diseased stony corals were relatively rare in the three most southerly of the reefs (being 0% at Xahuayxol) (Table 2). Percentages of diseased corals were highest in Boca Paila (~30%) and Tampalam (25%) in the central geographic area and in El Placer (21%), the most northerly of the southern reefs. After pooling the data for each subreef, the southern area was considerably less affected overall than the central area (mean = 7.1 %, sd = 9 versus mean = 19.4%, sd = 11.7, respectively; for each n = 12 subreefs).

White plague (WPD) was the only disease recorded in Boca Paila, where *M. faveolata* constituted 50-75% of all diseased colonies. WPD was responsible for 66-88% of all diseased colonies in Tampalam (*M. annularis* species complex >> *A. agaricites* and *P. astreoides*) and 62-80 % in El Placer (*M. annularis* species complex >> *S. siderea*, *A. agaricites*, *D. strigosa*, *A. tenuifolia* and *Meandrina meandrites*). Some colonies of *M. meandrites* at Tampalam and El Placer were affected by black-band disease and by what appeared to be a form of red-band disease. Dark-spots disease (in *S. siderea*) and unknown diseases were also seen at Tampalam and a colony of *Diploria strigosa* with signs of a disease resembling yellow-blotch disease was seen at El Placer.

## Algae

Turf algae were the predominant algal group at all but one of the most northerly subreefs (Table 3). The mean relative abundance of macroalgae was higher in the central area and in El Placer (20-41%, n = 15 subreefs) than further south (13-19%, n = 9 subreefs). Crustose coralline algae were more abundant than macroalgae in most (8/9) subreefs in the three most southerly reefs and they were comparable to macroalgae in relative abundance in El Placer and in the central area.

Mean macroalgal canopy heights showed comparatively little variation among reefs, ranging between 3 cm in Xcalak and ~5.5 cm in Tampalam. Mean macroalgal indices (macroalgal relative abundance x canopy height, a proxy for macroalgal biomass) were lowest (<80) in most (seven/nine) of the subreefs at Xcalak, Xahuayxol and Mahahual; the highest values (>110) were found in all subreefs in Punta Allen and Tampalam, in two subreefs at Boca Paila and in El Placer N. Overall, the average macroalgal index was somewhat greater in the central area (mean = 114.8, sd = 61, n = 838 quadrats) than in the southern area (mean = 82.0, sd = 60 cm, n = 875 quadrats).

## Relationships

Significant relationships detected via correlation and regression analyses include: (1) positive relationships between mean coral maximum diameter and live coral cover ( $r = 0.60$ ,  $r^2 = 0.36$ ,  $p < 0.01$ ); (2) positive relationships between mean density of stony coral recruits and (a) mean live coral cover ( $r = 0.50$ ,  $r^2 = 25.4$ ,  $p = 0.01$ ) and (b) mean macroalgal relative abundance ( $r = 0.45$ ,  $r^2 = 20.8$ ,  $p = 0.025$ ); (3) positive relationships between standing dead colonies and nearest human population size ( $r = 0.58$ ,  $r^2 = 33.4$ ,  $p < 0.01$ ); and (4) inverse relationships between mean macroalgal abundance or macroalgal index and mean herbivorous fish density by subreef ( $r = -0.44$ ,  $r^2 = 19.5$ ,  $p = 0.03$ ;  $r = -0.55$ ,  $r^2 = 30.3$ ,  $p < 0.01$ , respectively; fish data from Núñez-Lara et al, this volume). No significant ( $p > 0.05$ ) relationships were found among the other tested variables.

## DISCUSSION

The species composition of the predominant large ( $\geq 25$  cm in diameter) stony corals was relatively homogeneous among the subreefs of any given reef. *Montastraea faveolata* was the most frequently encountered species at 9/12 subreefs in the central area, whereas in the southern area *M. annularis* was the most abundant species at over half (7/12) of the subreefs. As *M. faveolata* grows larger than *M. annularis* (Weil and Knowlton, 1994), it is not surprising that maximum diameters were slightly larger in the central area reefs even though total cover overall was the same in both areas.

The high rates of recent partial-colony mortality in the central area and at El Placer (the most northerly reef in the southern area and, like Tampalam, in the Sian Ka'an subregion) are perhaps explained by the high percentage of diseased corals that were present at the time of our surveys (~9.5-30. %,  $n = 5$  reefs). Delayed effects of the 1998 mass bleaching event may have been partially responsible for the high rates of recent mortality in the three most southerly reefs (Mahahual, Xahuayxol, Xcalak) where relatively few corals showed signs of disease.

Coral bleaching can be manifested in different intensities, can occur locally or over large geographic areas, and can have diverse causes. Mass bleaching events are generally associated with periods in which sea surface temperature (SST) is above average (at least  $1^{\circ}\text{C}$  higher than the summer maximum, Hoegh-Guldberg, 1999). The scleractinians that were bleached during this study were localized in two of the three reefs in the southern area that were surveyed in June-July, 1999 (Mahahual and Xahuayxol; Table 2). During April-July of 1999, SST averaged  $\sim 29.3^{\circ}\text{C}$  in the southern area of Quintana Roo (SST/AVHRR images courtesy of Ocean Remote Sensing Group, Johns Hopkins Applied Physics Laboratory, Garza-Pérez et al., in preparation). We suspect that a minor warming which continued over several months may have triggered the bleaching observed in these two reefs and/or perhaps the stony corals had not yet fully recovered from the effects of the mass bleaching event in 1998.

Damage from the passage of Hurricane Mitch in October 1998 is also possible, although unlikely to have contributed very much to total mortality given the relatively high values for large standing dead colonies of *Acropora palmata* in Xcalak, which is probably more exposed to storm waves than Mahahual and Xahuayxol located in the lee of Banco Chinchorro (Fig. 1). Acroporids had earlier experienced massive mortality around the Caribbean (Goreau et al., 1998), primarily from white-band disease which presumably had also killed the standing dead colonies of *A. palmata* in Xcalak, Punta Allen and, to a lesser extent, in Punta Yuyum. The demise of *Diploria* at Punta Yuyum may either have resulted from some combination of bleaching-related mortality (*Diploria labyrinthiformis* was one of the most affected species during the 1998 mass bleaching event, Garza-Pérez, 1999) or from mortality produced by disease.

It was anticipated that the reefs in Mahahual and Xcalak (southern area) and in Punta Allen (central area) might have the lowest live stony coral cover and/or the highest rates of disease or mortality, given their proximity to long-established villages (~50 years; Bastida-Zavala et al., 2000). In fact, standing dead colonies presented the only significant positive relationship with human population size. Even this relationship could be compromised if colonies that we were unable to identify as standing dead were actually present in higher abundances in the other reefs and/or if *Acropora palmata*

happened to have been relatively more abundant historically at 10 m in the Xcalak and Punta Allen fore reefs than elsewhere.

Nevertheless, in terms of live coral cover, old mortality, and standing dead corals, Xcalak, which is one of the oldest settlements in the Mexican Caribbean, was the most impacted of the eight study reefs. Our results are in agreement with Garza-Pérez's (1999) comparative analysis of the same spur-and-groove habitat in Xcalak, Mahahual, Tampalam, and Akumal. Xcalak was so important in fishing and coconut cultivation at the beginning of the 20th Century that an artificial canal was constructed to connect the fisherman of the coast with the city of Chetumal (the present capital of Quintana Roo). Long-term anthropogenic impacts, such as wastewater discharge and fishing, could have contributed to its current condition. However, Xcalak was also the closest study reef to the prolonged high SST anomaly that resulted in high rates of bleaching-related mortality in Belize during the 1998 ENSO event (see Peckol et al., this volume).

Herbivorous fishes and echinoids are important determinants of the distribution and abundance of benthic algae and stony coral communities in the wider Caribbean (Wood, 1999). Increases in the abundance of macroalgae are correlated with the mass mortality of *Diadema antillarum* throughout the region in 1983-84 and with declines in the abundance of herbivorous fishes (Porter and Meier, 1992; Steneck, 1994). The converse has also been found when macroalgae decline after *D. antillarum* reappears (Aronson and Precht, 2000). Given the almost total absence of *D. antillarum* in our surveys, and the inverse relationships between mean herbivorous fish density (Núñez-Lara et al., this volume) and both the mean relative abundance of macroalgae and macroalgal index, differential fishing inside and outside of Sian Ka'an Biosphere Reserve may have exerted a primary control on the abundance and biomass of macroalgae in the central-southern Quintana Roo reefs. In other words, the somewhat higher relative abundance of macroalgae in the central area compared to the southern area may be a consequence of reductions in the local herbivorous fish community as populations of piscivorous fishes have rebounded within the reserve. Effects of protection on the abundance of benthic organisms and fishes have been demonstrated previously in the Mexican Caribbean (Arias-González, 1998; Arias-González et al., 1999) and other parts of the world (e.g., Edgar and Barrett, 1999).

If the presence of coral recruits is a sign that a reef is maintaining itself, and has the potential to continue to do so in the future, then all the reefs in the study area are in good condition. However, the complete lack of correspondence between genera that are most abundant as large colonies and those present as recruits (Fig. 2 versus Fig. 4) may suggest a dominance of asexual reproduction in the maintenance of the primary reef-building corals, as has been reported elsewhere in the Caribbean (Jackson, 1985). Hence, the 50% decrease in live stony coral cover that had occurred throughout the region between the early 1990's and 1999 (from ~25% to ~12%) and the high rates of recent partial-colony mortality (7-27.5%) seen in our 1999 surveys are cause for concern regarding the future status of these reef ecosystems. Further indications of widespread disturbance are provided by the size-frequency distributions of most species of large corals being skewed towards small-intermediate sizes (Fig. 3A, B).

In general, it can be said that the condition of the stony corals at 10 m depth in the fore-reef spur-and-groove habitat in the central and southern areas of the Mexican Caribbean already has deteriorated from natural and possibly anthropogenic

perturbations, particularly the effects of diseases (in stony corals and *Diadema*) and bleaching. Its fish communities are also being affected by extractive activities, particularly in the southern area (see Núñez-Lara et al., this volume), possibly leading to further deterioration in total reef condition. Thus the introduction of new disturbances related to urbanization and tourist development in the southern area (Daltabuit Godás, 1999) could potentially overwhelm the natural resilience of these coastal coral reefs (i.e., Nyström et al., 2000).

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Table 1. Site information for AGRRA stony coral and algae surveys in central-southern Quintana Roo, México.

Site name	Site code	Nearest human population (#) <sup>1</sup>	Reef type	Latitude (° " N)	Longitude (° " W)	Survey date(s)	Depth (m)	Benthic transects (#)	≥25 cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)
Boca Paila North	BPN		Fringing S&G <sup>2</sup>	20 06 51	87 27 23	Aug. 30 99	11.5	10	6	15.5 ± 4.5
Boca Paila Center	BPC	<20	Fringing S&G	20 06 21	87 27 34	Aug. 31 99	9.5	7	6	19.0 ± 9.5
Boca Paila South	BPS		Fringing S&G	20 05 51	87 27 47	Sept. 01 99	9	10	2	8.5 ± 2.5
Punta Yuyum North	PYN		Fringing S&G	19 58 30	87 27 10	Aug. 27 99	10	11	4.5	10.5 ± 5.5
Punta Yuyum Center	PYC	0	Fringing S&G	19 58 00	87 27 06	Aug. 26,28 99	10	14	4	12.5 ± 6.0
Punta Yuyum South	PYS		Fringing S&G	19 57 30	87 26 52	Aug. 25,28 99	10.5	14	4.5	10.5 ± 6.0
Punta Allen North	PAN		Fringing S&G	19 50 30	87 26 15	Aug. 20,23 99	10.5	14	4.5	11.5 ± 3.5
Punta Allen Center	PAC	<300	Fringing S&G	19 50 00	87 26 36	Aug. 21,23 99	12	15	4.5	14.5 ± 7.5
Punta Allen South	PAS		Fringing S&G	19 49 30	87 26 52	Aug. 22,23 99	11	15	3.5	10.5 ± 4.5
Tampalam North	TN		Fringing S&G	19 09 15	87 32 00	Oct. 2 99	10.5	15	1.5	8.5 ± 5.5
Tampalam Center	TC	<20	Fringing S&G	19 08 45	87 32 10	Sept. 30, Oct. 1,3 99	8	14	5	15.0 ± 9.0
Tampalam South	TS		Fringing S&G	19 08 15	87 32 13	Sept. 30, Oct. 1,3 99	8	12	6	18.0 ± 6.5
El Placer North	EPN		Fringing S&G	18 54 48	87 37 03	Sept. 24-25 99	12	14	2	6.0 ± 2.5
El Placer Center	EPC	<20	Fringing S&G	18 54 08	87 37 24	Sept. 23-24,26 99	12	15	2.5	9.0 ± 3.5
El Placer South	EPS		Fringing S&G	18 53 35	87 37 36	Sept. 22-23,25 99	13	15	4	12.0 ± 4.0
Mahahual North	MN		Fringing S&G	18 43 24	87 41 56	June 25-27, July 22 99	9.5	13	5	17.0 ± 7.0
Mahahual Center	MC	<150	Fringing S&G	18 43 01	87 42 09	June 26,28 99	10.5	13	4.5	17.0 ± 6.0
Mahahual South	MS		Fringing S&G	18 42 30	87 42 20	June 29, July 22 99	10	12	6	16.5 ± 5.5
Xahuayxol North	XN		Fringing S&G	18 30 55	87 45 02	July 9-10,20 99	9.5	14	3.5	11.5 ± 3.0
Xahuayxol Center	XC		Fringing S&G	18 30 25	87 45 13	July 10,20-21 99	11	14	2.5	12.0 ± 5.5
Xahuayxol South	XS	< 5	Fringing S&G	18 29 55	87 45 22	July 18,21 99	10.5	14	3.5	11.5 ± 5.5
Xcalak North	XCN		Fringing S&G	18 13 43	87 49 51	July 13-15 99	11	12	3.5	9.0 ± 4.5
Xcalak Center	XCC	<300	Fringing S&G	18 13 09	87 49 54	July 14-15,17 99	9	11	3.5	9.5 ± 3.0
Xcalak South	XCS		Fringing S&G	18 12 39	87 49 47	July 16-17 99	10	13	3	7.5 ± 4.5

<sup>1</sup>from INEGI (1996)<sup>2</sup>S&G = spur and groove

Table 2. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 25$  cm diameter) by subreefs in central-southern Quintana Roo, México.

Site name	Stony corals		Partial-colony mortality (%)			Live tissues bleached (%)	Stony corals (%)		Diseased
	#	Diameter	Recent	Old	Total		Standing dead	Bleached	
Boca Paila North	59	66.5 $\pm$ 44.0	17.5 $\pm$ 18.0	18.0 $\pm$ 23.5	35.0 $\pm$ 26.5	0	5	0	44
Boca Paila Center	42	73.0 $\pm$ 55.5	14.5 $\pm$ 13.0	16.0 $\pm$ 22.0	30.0 $\pm$ 23.0	0	5	0	28.5
Boca Paila South	22	46.5 $\pm$ 30.5	25.5 $\pm$ 17.0	19.5 $\pm$ 17.0	45.0 $\pm$ 19.5	0	0	0	18
Punta Yuyum North	51	59.5 $\pm$ 40.0	20.5 $\pm$ 23.0	25.5 $\pm$ 31.5	44.0 $\pm$ 32.5	5.0 (n=1)	10	2	8
Punta Yuyum Center	55	53.5 $\pm$ 36.0	23.0 $\pm$ 21.5	17.5 $\pm$ 28.0	39.0 $\pm$ 29.0	0	7.5	0	12.5
Punta Yuyum South	63	66.0 $\pm$ 61.5	18.5 $\pm$ 21.0	28.5 $\pm$ 35.0	44.0 $\pm$ 35.5	0	16	0	8
Punta Allen North	64	51.0 $\pm$ 34.5	19.0 $\pm$ 24.0	23.0 $\pm$ 32.5	39.5 $\pm$ 33.5	25.0 $\pm$ 8.5 (n=3)	12.5	4.5	17
Punta Allen Center	66	65.5 $\pm$ 52.5	16.0 $\pm$ 23.0	14.5 $\pm$ 25.5	29.5 $\pm$ 31.0	0	6	4.5	12
Punta Allen South	55	64.5 $\pm$ 42.5	20.5 $\pm$ 24.0	19.0 $\pm$ 31.0	37.5 $\pm$ 32.5	0	11	0	9
Tampalam North	24	47.0 $\pm$ 24.5	13.0 $\pm$ 13.0	18.5 $\pm$ 13.0	31.0 $\pm$ 19.5	8.0 (n=1)	0	4	37.5
Tampalam Center	68	45.0 $\pm$ 26.0	13.5 $\pm$ 13.5	17.0 $\pm$ 16.5	30.5 $\pm$ 22.0	0	0	0	16
Tampalam South	70	62.0 $\pm$ 51.0	16.0 $\pm$ 17.0	19.5 $\pm$ 17.5	35.0 $\pm$ 24.0	0	1.5	0	21.5
El Placer North	30	38.5 $\pm$ 11.5	15.0 $\pm$ 18.0	20.5 $\pm$ 19.0	35.5 $\pm$ 24.5	0	0	0	16.5
El Placer Center	40	45.0 $\pm$ 25.5	16.0 $\pm$ 16.5	18.5 $\pm$ 22.5	34.0 $\pm$ 27.0	3.0 (n=1)	5	2.5	20
El Placer South	57	56.5 $\pm$ 37.5	22.5 $\pm$ 20.0	18.5 $\pm$ 18.0	40.5 $\pm$ 27.0	4.5 $\pm$ 1.0 (n=3)	2	5.5	26.5
Mahahual North	68	46.5 $\pm$ 31.0	12.0 $\pm$ 19.0	18.0 $\pm$ 21.0	30.0 $\pm$ 25.5	18 (n=12) <sup>1</sup>	1.5	17.5	6
Mahahual Center	57	60.5 $\pm$ 45.0	18.5 $\pm$ 17.5	16.5 $\pm$ 14.0	35.0 $\pm$ 21.0	100 (n=20) <sup>1</sup>	2	35	5.5
Mahahual South	74	67.5 $\pm$ 43.5	27.5 $\pm$ 22.5	17.0 $\pm$ 22.5	43.5 $\pm$ 28.5	45 $\pm$ 39.5 (n=31) <sup>1</sup>	4	42	4
Xahuayxol North	49	48.0 $\pm$ 31.0	18.5 $\pm$ 23.5	10.5 $\pm$ 12.5	29.0 $\pm$ 26.5	39.5 $\pm$ 42.0 (n=17)	0	34.5	0
Xahuayxol Center	37	51.5 $\pm$ 38.0	13.5 $\pm$ 16.5	13.5 $\pm$ 24.0	26.0 $\pm$ 29.0	21.5 $\pm$ 33.0 (n=8)	5.5	21.5	0
Xahuayxol South	48	54.5 $\pm$ 33.0	20.5 $\pm$ 23.5	14.0 $\pm$ 17.0	34.0 $\pm$ 27.0	7.5 $\pm$ 2.5 (n=3)	2	6.5	0
Xcalak North	44	49.0 $\pm$ 29.0	16.0 $\pm$ 16.5	22.0 $\pm$ 34.0	36.5 $\pm$ 34.5	11.5 $\pm$ 6.5 (n=9)	11.5	20.5	4.5
Xcalak Center	38	53.0 $\pm$ 30.5	7.0 $\pm$ 8.5	38.5 $\pm$ 41.0	43.5 $\pm$ 38.5	6.0 $\pm$ 5.5 (n=4)	23.5	10.5	2.5
Xcalak South	37	51.0 $\pm$ 30.5	20.0 $\pm$ 22.0	24.0 $\pm$ 35.5	41.0 $\pm$ 35.5	14.5 $\pm$ 5.5 (n=3)	13.5	8	0

<sup>1</sup> Overestimate, due to some missing data

Table 3. Algal characteristics, density of stony coral recruits and *Diadema anillarum* (mean  $\pm$  standard deviation) by subreefs in central-southern Quintana Roo, México.

Site name	Quadrats (#)	Relative abundance (%)			Macroalgae		Index <sup>1</sup>	Recruits (#/.0625 m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )
		Macroalgae	Turf algae	Crustose coralline algae	Height (cm)	Macroalgae			
Boca Paila North	66	23.0 $\pm$ 7.0	51.5 $\pm$ 10.5	25.0 $\pm$ 9.5	4.0 $\pm$ 1.5	Macroalgae	90	0.1 $\pm$ 0.4	0
Boca Paila Center	30	41.0 $\pm$ 15.5	32.0 $\pm$ 15.5	27.0 $\pm$ 14.5	3.0 $\pm$ 1.0	Macroalgae	130	0.8 $\pm$ 1.0	0
Boca Paila South	50	24.5 $\pm$ 9.0	55.5 $\pm$ 12.5	20.0 $\pm$ 10.0	4.5 $\pm$ 1.5	Macroalgae	113	0.4 $\pm$ 0.8	0
Punta Yuyum North	58	23.5 $\pm$ 7.0	53.5 $\pm$ 8.5	23.0 $\pm$ 7.5	4.0 $\pm$ 2.0	Macroalgae	96	0.1 $\pm$ 0.3	0
Punta Yuyum Center	74	22.0 $\pm$ 7.5	56.0 $\pm$ 10.5	22.0 $\pm$ 8.5	3.5 $\pm$ 2.0	Macroalgae	81	0.2 $\pm$ 0.5	0
Punta Yuyum South	76	20.0 $\pm$ 8.5	56.5 $\pm$ 15.0	23.5 $\pm$ 13.0	3.5 $\pm$ 2.0	Macroalgae	81	0.1 $\pm$ 0.4	0
Punta Allen North	78	22.5 $\pm$ 7.0	49.0 $\pm$ 12.0	25.5 $\pm$ 12.5	5.5 $\pm$ 2.0	Macroalgae	142	0.1 $\pm$ 0.4	0
Punta Allen Center	81	25.5 $\pm$ 6.5	42.0 $\pm$ 10.5	25.5 $\pm$ 10.5	5.0 $\pm$ 2.0	Macroalgae	116	0.1 $\pm$ 0.2	0
Punta Allen South	76	24.5 $\pm$ 9.0	44.0 $\pm$ 13.5	31.5 $\pm$ 12.0	5.0 $\pm$ 2.0	Macroalgae	126	0.1 $\pm$ 0.2	0
Tampalam North	78	23.5 $\pm$ 5.0	57.5 $\pm$ 11.5	19.0 $\pm$ 9.5	6.5 $\pm$ 2.0	Macroalgae	149	0.1 $\pm$ 0.3	0.7
Tampalam Center	87	23.5 $\pm$ 9.0	49.0 $\pm$ 15.5	27.5 $\pm$ 11.0	5.5 $\pm$ 2.0	Macroalgae	126	0.3 $\pm$ 0.7	0
Tampalam South	84	24.5 $\pm$ 9.5	46.0 $\pm$ 17.0	29.5 $\pm$ 13.0	5.0 $\pm$ 2.0	Macroalgae	124	0.3 $\pm$ 0.6	0
El Placer North	73	22.0 $\pm$ 5.5	56.5 $\pm$ 10.5	21.5 $\pm$ 10.0	6.5 $\pm$ 2.5	Macroalgae	135	0.1 $\pm$ 0.4	0
El Placer Center	81	21.5 $\pm$ 6.5	48.5 $\pm$ 14.5	30.0 $\pm$ 11.5	4.5 $\pm$ 2.0	Macroalgae	104	0.3 $\pm$ 0.5	0
El Placer South	88	26.0 $\pm$ 9.0	46.0 $\pm$ 15.0	28.0 $\pm$ 11.0	4.0 $\pm$ 1.5	Macroalgae	107	0.2 $\pm$ 0.4	0
Mahahual North	73	13.0 $\pm$ 11.5	63.0 $\pm$ 16.5	24.0 $\pm$ 16.0	4.0 $\pm$ 2.5	Macroalgae	64	0.4 $\pm$ 0.6	0
Mahahual Center	88	16.0 $\pm$ 12.5	69.5 $\pm$ 15.0	15.0 $\pm$ 13.0	6.5 $\pm$ 4.0	Macroalgae	108	0.4 $\pm$ 0.6	0
Mahahual South	72	18.0 $\pm$ 10.0	56.0 $\pm$ 18.0	26.0 $\pm$ 12.5	4.0 $\pm$ 2.5	Macroalgae	74	0.2 $\pm$ 0.4	0
Xahuayxol North	77	18.0 $\pm$ 6.0	53.5 $\pm$ 10.5	28.5 $\pm$ 10.5	4.0 $\pm$ 1.5	Macroalgae	75	0.2 $\pm$ 0.4	0
Xahuayxol Center	73	18.0 $\pm$ 7.5	57.5 $\pm$ 10.5	24.5 $\pm$ 9.5	4.5 $\pm$ 1.5	Macroalgae	83	0.3 $\pm$ 0.7	0
Xahuayxol South	86	19.0 $\pm$ 6.5	47.0 $\pm$ 14.0	34.0 $\pm$ 13.0	3.5 $\pm$ 1.5	Macroalgae	71	0.1 $\pm$ 0.3	0
Xcalak North	74	14.5 $\pm$ 8.5	55.5 $\pm$ 12.5	30.0 $\pm$ 11.0	3.0 $\pm$ 1.5	Macroalgae	52	0.1 $\pm$ 0.4	0
Xcalak Center	55	16.5 $\pm$ 8.5	48.0 $\pm$ 17.0	35.5 $\pm$ 17.5	3.0 $\pm$ 1.0	Macroalgae	49	0.1 $\pm$ 0.4	0
Xcalak South	69	18.0 $\pm$ 8.5	50.5 $\pm$ 17.0	31.5 $\pm$ 16.0	3.0 $\pm$ 1.5	Macroalgae	53	0.1 $\pm$ 0.3	0

<sup>1</sup>Macroalgal index = relative macroalgal abundance x macroalgal height

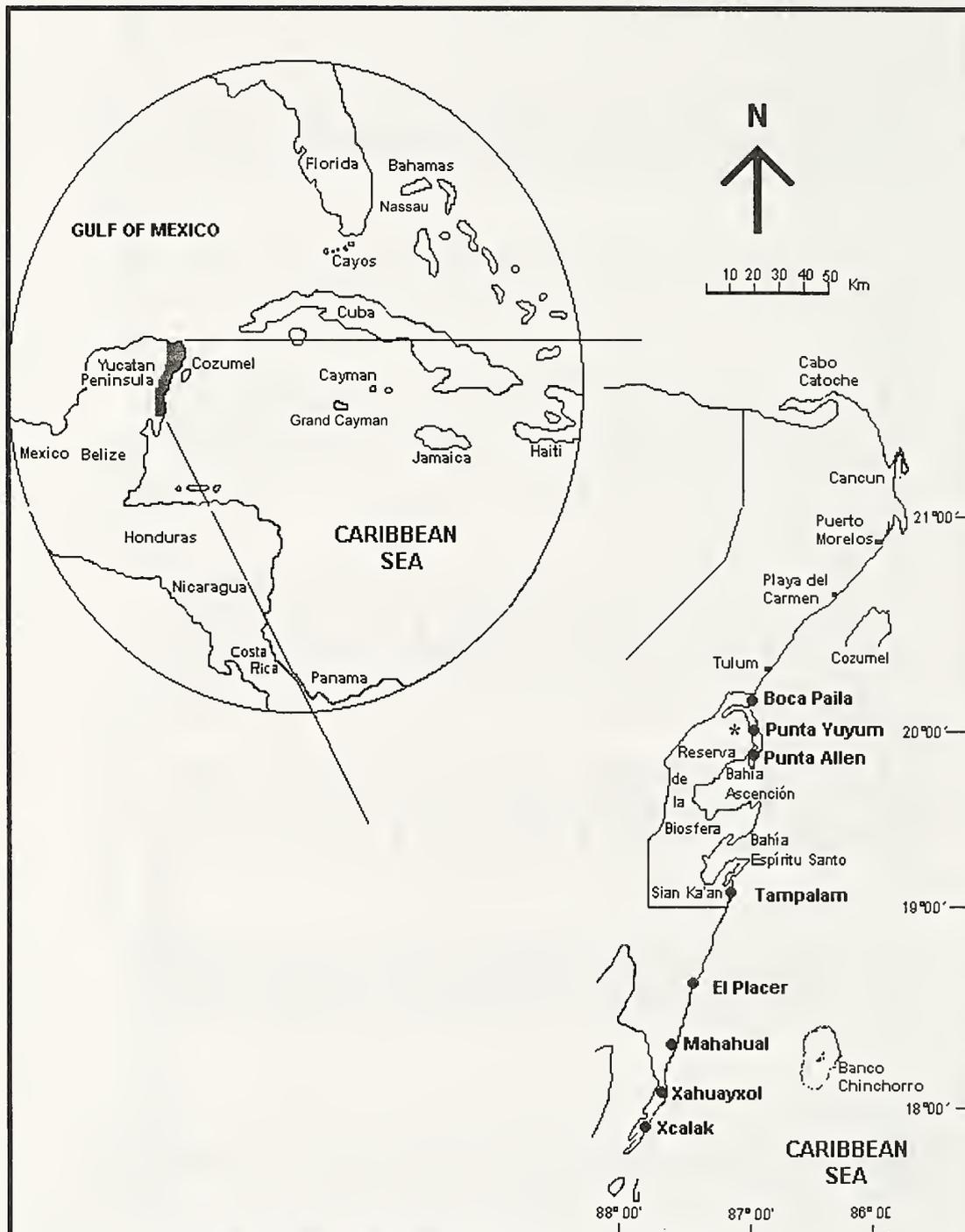


Figure 1. AGRRA survey reefs in central-southern Quintana Roo, Mexico. Modified from Núñez-Lara and Arias-González (1998).

# CONDITION OF CORAL REEF ECOSYSTEMS IN CENTRAL-SOUTHERN QUINTANA ROO (PART 2: REEF FISH COMMUNITIES)

BY

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## ABSTRACT

Increases of fishing and tourism threaten the natural relationships between reef fish communities and their environments. All species of reef fishes were visually assessed in the central and southern Mexican Caribbean in eight fringing reefs, four of which are in a protected biosphere reserve. The sampling design included three spatial scales from tens of meters to tens of kilometers. A total of 9,908 individuals belonging to 128 species and 43 families were identified in 144 belt transects. Zooplankton feeders were the most important trophic group by number of individuals; plant and detritus feeders dominated by number of species. Herbivores were larger in unprotected reefs than in the reserve. Regression analyses showed significant inverse relationships between total fish species density and macroalgal index (a proxy for macroalgal biomass) and, for “large” ( $\geq 25$  cm diameter) stony corals, partial-colony mortality and live/dead ratio. Significant inverse relationships were also found between mean abundance of the plant and detritus feeders guild and macroalgal index and macroalgae abundance. Geomorphological factors and anthropogenic impacts, both positive (protection) in the reserve and negative (fishing and tourism) in unprotected areas, may explain these spatial patterns in reef-fish community structure.

## INTRODUCTION

The reefs of the Mexican Caribbean run along the eastern margin of the Yucatán Peninsula. They are distributed parallel to the coastline of Quintana Roo state as a fringing reef system which originated during Miocene-Pleistocene rifting of the carbonate platform (Weidie, 1985). Eight reefs with similar structural configurations were chosen for the present study (Fig. 1). The reef profile can generally be divided into three main

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zones: reef lagoon, reef crest, and fore reef (mostly spur-and-groove). Four of the reefs are located within the Sian Ka'an Biosphere Reserve, which extends from 19° 05'N to 20° 06'N. The reserve, which was created in 1986 by presidential decree, has approximately 120,000 ha of coastal environments including 37,000 ha of coral reefs that were added in 1998. Natural ecological conditions within the reserve have been conserved due to the long distance from large population centers, limited access, and the restrictions on fishing and tourism. Punta Allen (Rojo Gómez) and Punta Herrero are the reserve's two largest human communities, both having less than 300 inhabitants. The villagers are principally dedicated to lobster fishing and less effort is given to catching bony fishes of which the main species are mojarras (*Gerres* spp.), snappers (lutjanids), barracudas (*Sphyræna barracuda*), grunts (haemulids), and groupers (serranids).

The remaining reefs are located between the southern boundary of the Sian Ka'an Biosphere Reserve and the frontier with Belize. The predominant activity in this area is fishing, mainly for local consumption. Most fish are caught in Banco Chinchorro, a shelf-edge bank reef system located about 45 km seaward of Mahahual town. The principal fishing methods employed are trotlines, traps, gill nets and harpoons [Oficina Regional de Pesca (SEMARNAP), unpublished report]. The Regional Fishing Office for 1997 reported a total fish catch of 335,443 kg from the zone between Punta Herrero, in the southern of Sian Ka'an reserve, and Xcalak, near the Belize border. However, locality-specific information was not provided.

Tourism has recently begun to grow rapidly along the unprotected southern coast of the Mexican Caribbean. The reefs under greatest threat, Mahahual and Xcalak, are in the path of large developers. Tourism potentially constitutes a relatively benign and lucrative use of coral reef resources. However, this benefit can be counteracted by damage and overexploitation (Hawkins and Roberts, 1993). The relationship between tourism and reef fishes is mainly indirect, being felt through the effects on fish habitat (Russ, 1991) including coral breakage and death due to vessel anchors, sedimentation, dredging, and other coastal zone activities (Dollar, 1982; Grigg, 1994; Muthiga and McClanahan, 1997). A direct effect of tourism is the extraction of fishes for consumption by visitors. Changes in community structure are caused by overexploitation of fishes of the high trophic levels (Russ, 1991). Thus, effects of fishing are partially attributable to the high demand of fishes for tourists and partially to the continuous increase in the numbers of fishermen and local inhabitants.

There are relatively few studies of reef fish communities in the Mexican Caribbean (e.g., Fenner, 1991; Díaz-Ruiz and Aguirre-León, 1993; Schmitter-Soto, 1995; Arias-González, 1998; Díaz-Ruiz et al., 1998; Núñez-Lara and Arias-González, 1998). In the present study we tried to detect the main structural forces affecting reef fish communities, and to measure the influence of fisheries and tourism on fishes and their habitat. The data generated represents a basis for comparative analyses of reef fish community structure at different spatial scales.

## METHODS

Reef fishes were visually censused at eight fringing reef localities along Mexico's central and southern Caribbean coast. The reefs were selected on the basis of a mixture of

strategic and representative criteria, including their spatial separation distance (usually 20-30 km), natural geographic barriers (such as the two large bays that divide the Sian Ka'an Biosphere Reserve), and their type of use by humans. For the purpose of this study, every reef was assigned to one of three geographical areas (Fig. 1): Northern Sian Ka'an (NSK) at 19°-20° N (3 reefs about 25 km apart); Southern Sian Ka'an (SSK) at 18°-19° N (2 reefs about 30 km apart—one being outside the reserve); and Southern (S) at 17°-18° N (3 reefs about 30 km apart). The distance parallel to the reef crest that constituted each reef was approximately three kilometers. This distance was subdivided into three 0.9-1 km subreefs (north, center, and south). Subreefs were geographically localized with GPS and described in terms of distance from the coast and degree of exposure to oceanic currents. Six replicate belt transects, each measuring 50 m long by 2 m wide, were swum parallel to the coast at every subreef. Transects were spaced approximately 100 m apart and all surveys were made between 0900 and 1700 hours by one diver (Nuñez-Lara). All transects were made at an average depth of 12 meters in the fore reef. The dominant habitat was spur and groove, except in northern Tampalam, where the calcareous substratum was largely covered with benthic algae, gorgonians, and sponges.

All reef fishes  $\geq 3$  cm in body length within the transects were counted and their sizes estimated in six categories: 3-10 cm, 11-20 cm, 21-30 cm, 31-40 cm, 41-50 cm and  $> 50$  cm. The Atlantic and Gulf Rapid Reef Assessment (AGRRA) fishes constitute a subset of the "all species" data: in this paper, "serranids" are species of *Epinephelus* (including *E. fulvus* but excluding *E. cruentatus*, which is here classified as *Cephalopholis cruentata*) and *Mycteroperca*; "haemulids" and "scarids" (parrotfishes) refer to fishes that are  $\geq 3$  cm in total length.

In order to describe the reef fish community structure, the following ecological descriptors were calculated: species richness, abundance, density, trophic structure and size structure. Three different spatial scales were used for the analysis: subreef (hundreds of meters), reef (kilometers) and area (tens of kilometers). Trophic structure was analyzed by calculating the percentage of individuals and fish species belonging to each of Randall's (1967) feeding categories: plant and detritus feeders; zooplankton feeders; sessile invertebrate feeders; "shelled" invertebrate feeders; generalized carnivores; ectoparasite feeders; and fish feeders.

A multiple regression technique was used to relate the total fish density with the following benthic habitat variables assessed by Ruiz et al. (this volume): total live stony coral cover; total (recent + old) partial-colony mortality, old partial-colony mortality, recent partial-colony mortality, live:dead ratio and maximum diameter for "large" ( $\geq 25$  cm in diameter) stony corals; and relative abundance and macroalgal index (relative abundance  $\times$  height, a proxy for biomass) for macroalgae. Simple regression was used to examine the relationship between total fish herbivore density and the two macroalgal descriptors (index and relative abundance). Densities were  $\log+1$  transformed to meet the assumptions for parametric regression tests. Classification analysis was performed to determine the degree of similarity among subreefs, based on the abundance values of their recorded fish species. The Bray-Curtis (1957) distance index was used as a similarity measure and the Unweighted Pair Grouping Method Average (UPGMA) as a clustering method.

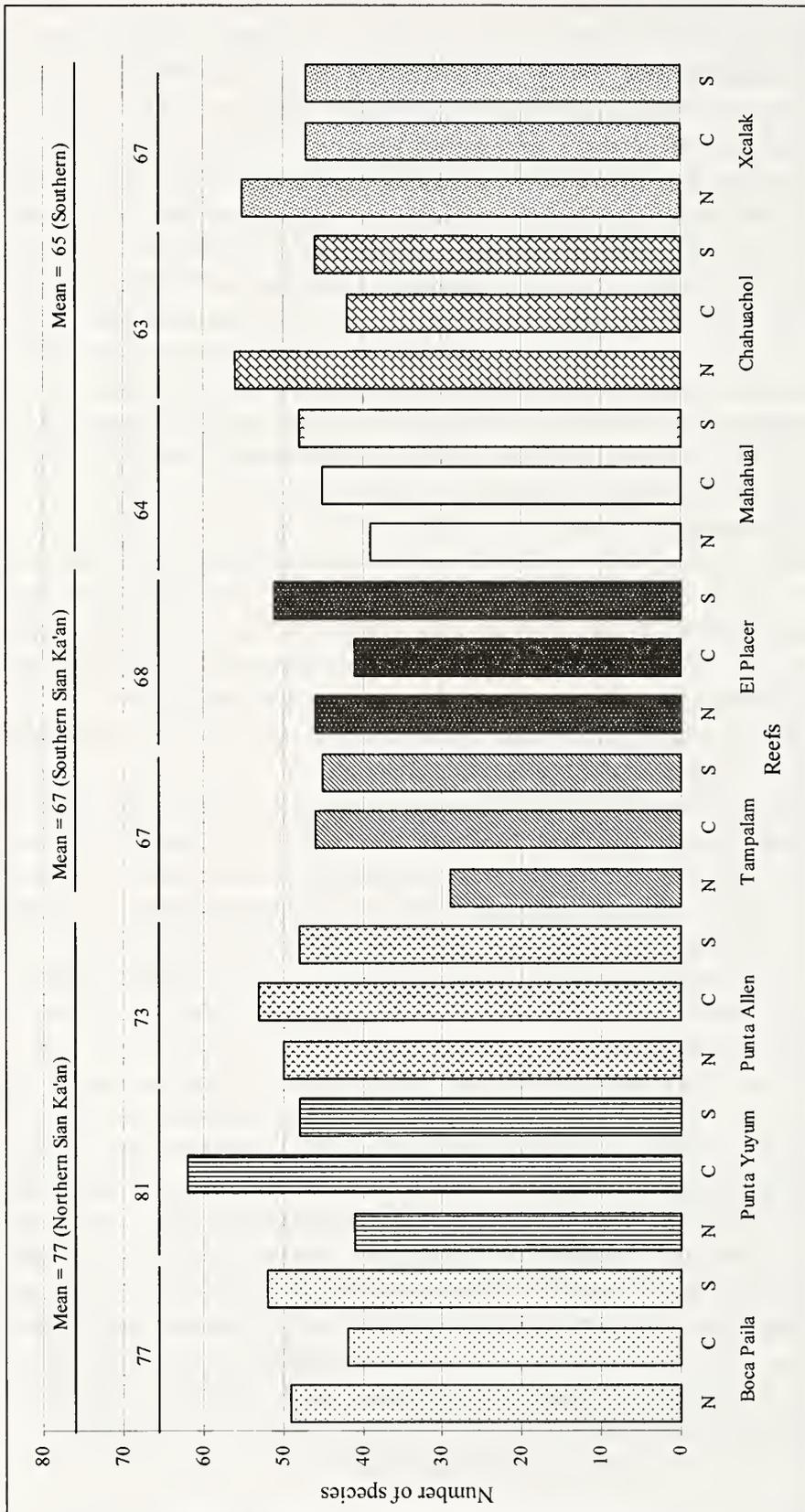


Figure 2. Species richness (all fishes ≥ 3 cm long) by subreef at 12 m depth in central-southern Quintana Roo, Mexico.

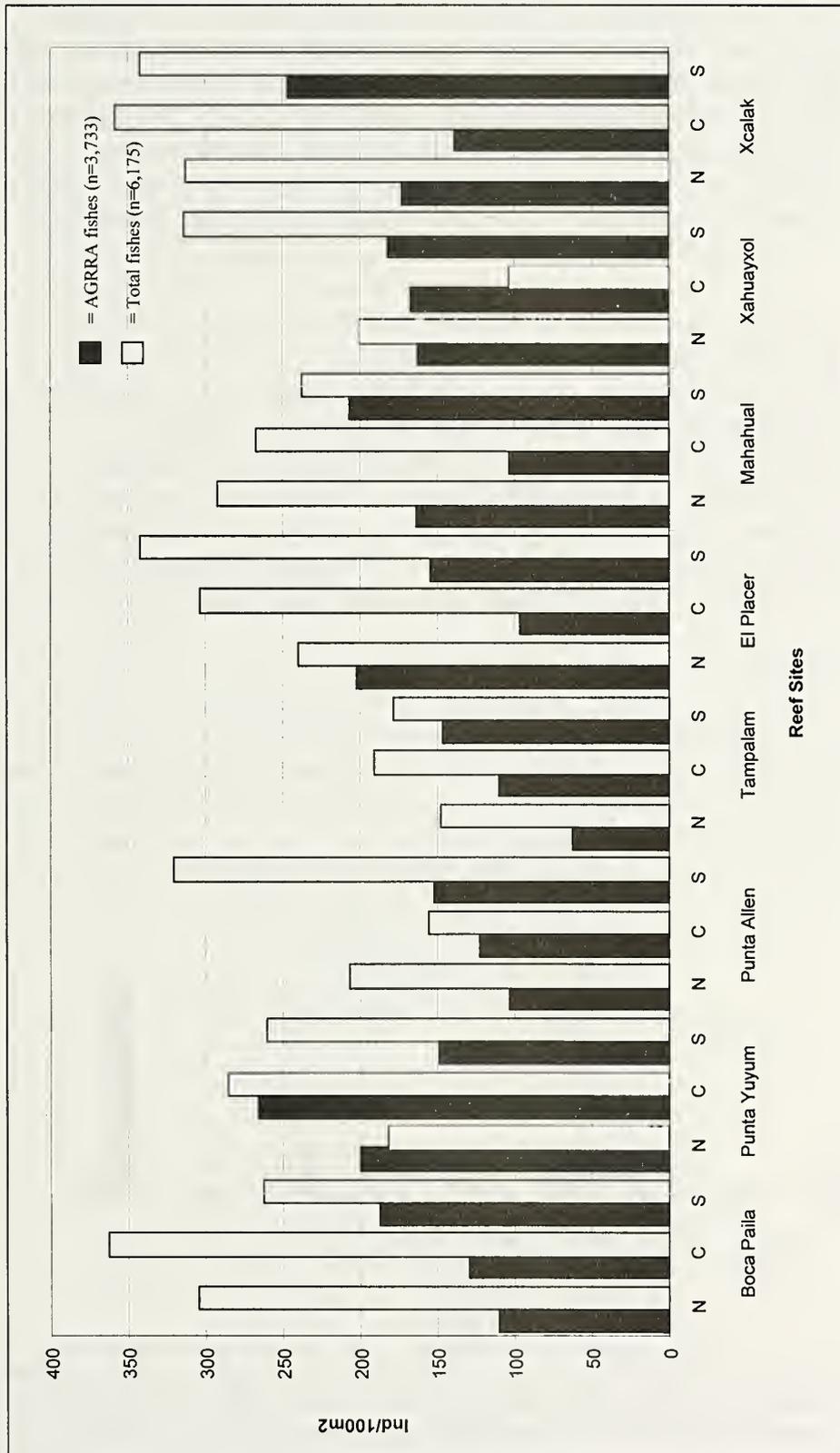


Figure 3. Mean density (no. individuals/100 m<sup>2</sup>) of all fishes ( $\geq 3$  cm long), and of AGRRA fishes, by subreef at 12 m depth in central-southern Quintana Roo, Mexico.

## RESULTS

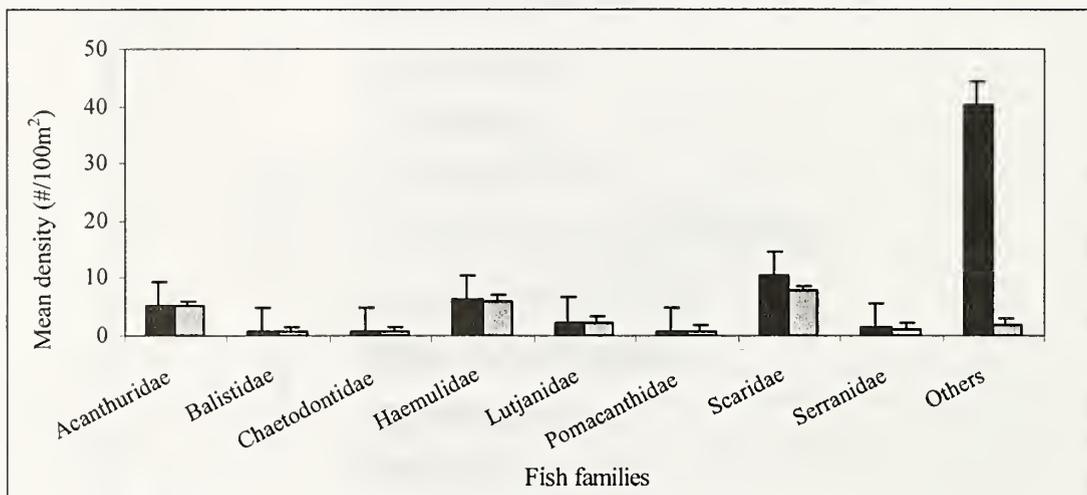
### Species Composition and Richness

A total of 128 reef fish species belonging to 43 families were identified in 144 belt transects. The greatest number was found in the NSK area and the lowest in the S area (Table 1, Fig. 2). Particularly notable was the large number of species belonging to the families Holocentridae (squirrelfish), Serranidae (grouper) and Haemulidae (grunts) in the NSK area and the relatively low number of species from such "typical" reef fish families as Pomacentridae (damselfish), Labridae (wrasse), and Scaridae (parrotfishes) in the SSK area. A somewhat different pattern was evident for AGRRA fishes (Table 1) as species numbers overall were rather similar in the NSK and S areas (23-32,  $n = 18$  subreefs). In the SSK area, the number of AGRRA species was slightly lower at El Placer (21-27,  $n = 3$  subreefs), but much smaller at Tampalam ( $n = 5-15$ ,  $n = 3$  subreefs).

The 25 dominant fish species in terms of sighting frequency and density belonged to the following families: Labridae, Acanthuridae (surgeonfish), Scaridae, Pomacentridae, Haemulidae, Serranidae, Lutjanidae (snappers), Pomacanthidae (angelfish), Holocentridae and Grammatidae (basslet). *Thalassoma bifasciatum* was the most frequently sighted and abundant species in all the studied reefs, followed by *Acanthurus coeruleus*, *Sparisoma aurofrenatum* and *Halichoeres garnoti* (Table 2). Forty percent of these dominant fish species are included in the AGRRA fish list.

### Abundance and Density

A total of 9,908 fishes were counted in the 144 transects, 3,733 of which belonged to the species on the AGRRA list. Total densities were highest in Xcalak S and Punta Yuyum C and lowest in Tampalam N and Xahuayxol C (Fig. 3). In an examination of the key AGRRA families, regardless of scoring mode (all individuals or restricted counts for haemulids, scarids, and serranids), the Scaridae was found to have the greatest density,



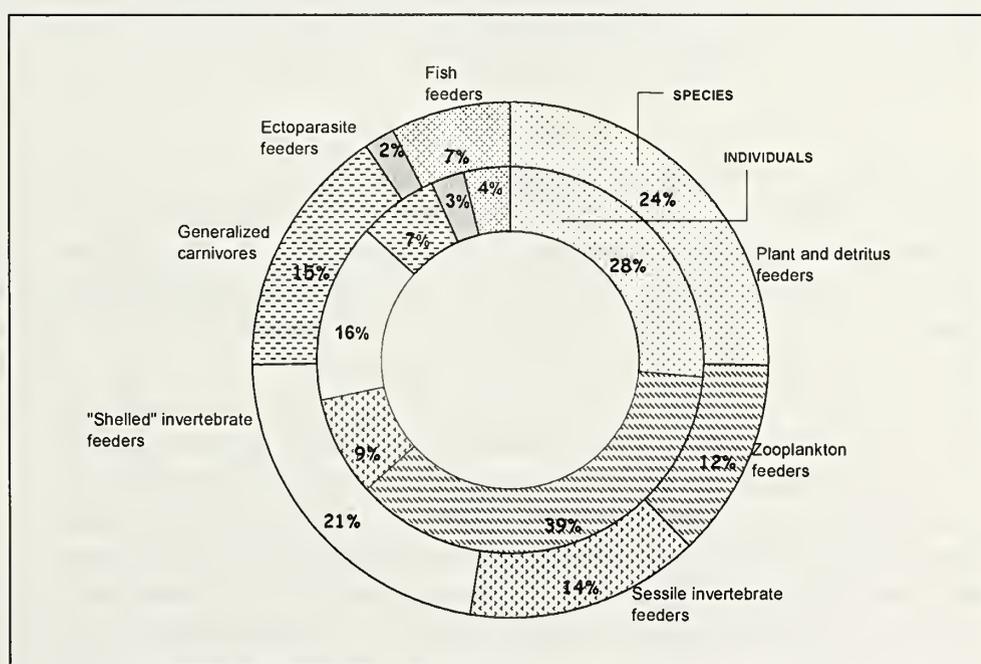
**Figure 4.** Mean density (no. individuals/100m<sup>2</sup> ± se) of all fishes (≥3 cm long), and of AGRRA fishes, by family in central-southern Quintana Roo, Mexico. Other AGRRA fishes = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Microspathodon chrysurus*, *Sphyrna barracuda*.

followed by the Haemulidae and Acanthuridae (Fig. 4). Although less abundant, the Lutjanidae and Serranidae were more plentiful than the Balistidae (leatherjackets), Chaetodontidae (butterflyfish), and Pomacanthidae.

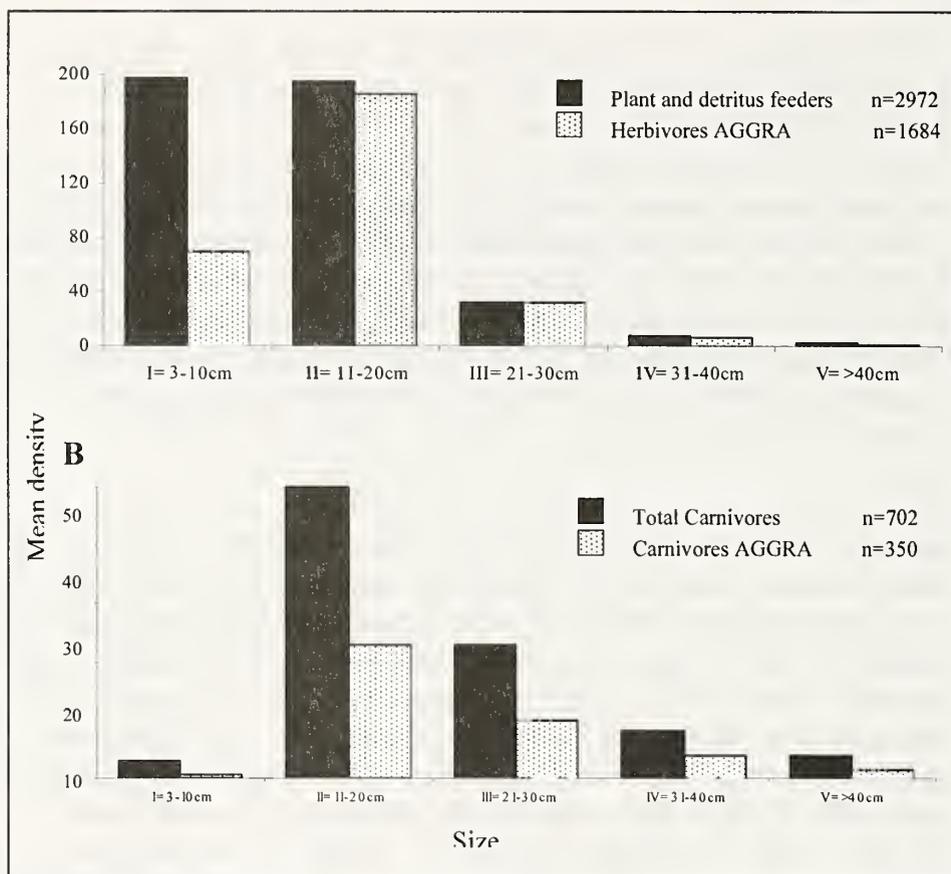
Parrotfishes  $\geq 3$ cm and surgeonfishes were most abundant overall in the S area, although the density of parrotfishes was also high at Punta Allen (in the NSK area). Surgeonfishes were relatively scarce at Mahahual (in the S area) where scarid density was highest (Table 3). The density of snappers was highest in Boca Paila and Punta Yuyum reefs in the NSK area and in Xcalak in the S area. Grunts  $\geq 3$ cm were also abundant in Punta Yuyum (especially) and Boca Paila as well as in El Placer (SSK area). The density of groupers (*Epinephelus*, *Mycteroperca*) was low in all the reefs.

### Trophic Structure

Considering total abundances, the trophic structure of the fish community was dominated by zooplankton feeders (39%), followed by plant and detritus feeders (28%) and shelled invertebrate feeders (16%). In term of total species richness, the three most important of Randall's (1967) trophic groups were the plant and detritus feeders (24%), "shelled" invertebrate feeders (21%), generalized carnivores (15%) and sessile invertebrate feeders (14%). Ectoparasite feeders and fish feeders showed the lowest percent contribution for both individuals and species (Fig. 5). Similar patterns were detected for every area and each of the sampled reefs, with zooplankton feeders dominating by number of individuals and the plant and detritus feeders being the most important group by numbers of species, closely seconded by "shelled" invertebrate feeders (Table 4). Among the most abundant of the herbivorous fish species were *Scarus iserti* (= *S. croicensis*), *Acanthurus coeruleus*, *A. bahianus*, *Sparisoma aurofrenatum* and *S. viride*.



**Figure 5.** Trophic structure (as percent of species and individuals) for all fishes  $\geq 3$  cm long at 12 m in central-southern Quintana Roo, Mexico.



**Figure 6.** Size frequency distribution of (A) all plant and detritus feeders  $\geq 3$  cm (acanthurids, kiphosids, pomacentrids except *Chromis*, and scarids) and AGRRA herbivores  $\geq 3$  cm (acanthurids, scarids, *Microspathodon chrysurus*) and (B) all carnivores  $\geq 3$  cm [carangids, lutjanids, scombrids, select serranids (*Epinephelus* and *Mycteroperca*), sphyraenids] and AGRRA carnivores  $\geq 3$  cm [lutjanids, *Epinephelus* (except for *E. cruentatus*) and *Mycteoperca*] at 12 m in central-southern Quintana Roo, Mexico.

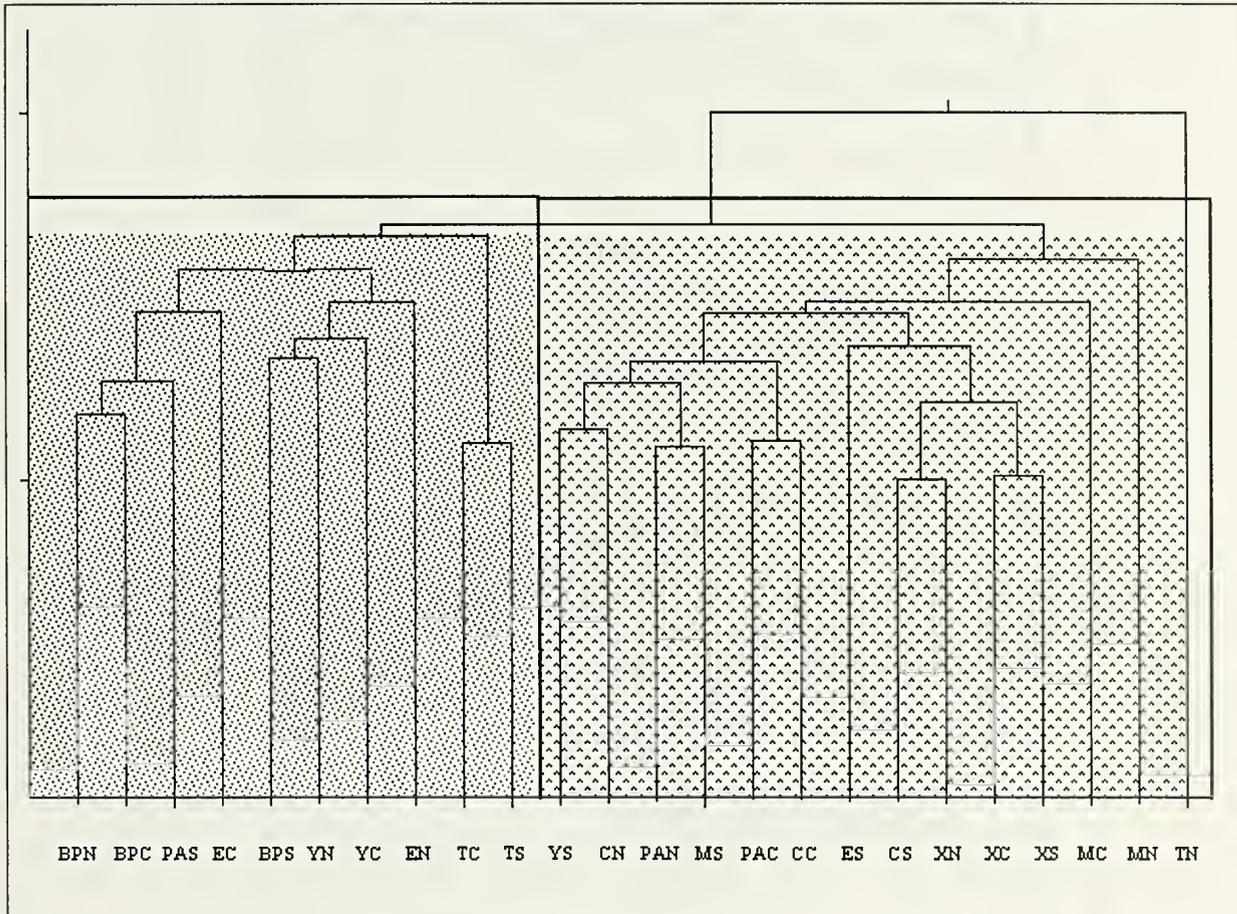
### Lengths

Very few of the surveyed fishes exceeded 30 cm in length. The most common size classes for “all herbivores” (acanthurids, kyphosids, pomacentrids except *Chromis*, and scarids) were equally divided between the 3-10 cm and 11-20 cm length intervals (Fig. 6A). In a more detailed analysis, we observed proportionately more fishes in the 11-20cm size class in the S area (particularly in Mahahual and Xcalak) than in the NSK and SSK areas where smaller herbivores (3-10cm) were relatively more abundant (Table 5). Key herbivores (acanthurids, scarids  $\geq 3$ cm, *Microspathodon chrysurus*) were slightly larger (10-20 cm) overall (Fig. 6A) and also attained their largest sizes in the S area.

Both for “all carnivores” (carangids, lutjanids, scombrids, sphyraenids, plus *Epinephelus* and *Mycteroperca*) and for the AGRRA carnivores (lutjanids, select serranids), the most common size class was 11-20 cm (Fig. 6B). Among the more abundant carnivores (total and AGRRA species) were the relatively small-sized *Epinephelus fulvus*, *Lutjanus apodus* and *L. synagris*. However, carnivores were slightly larger in the reefs in which they were most abundant (Xcalak, Boca Paila and Yuyum) (Table 5).

## Classification Analysis

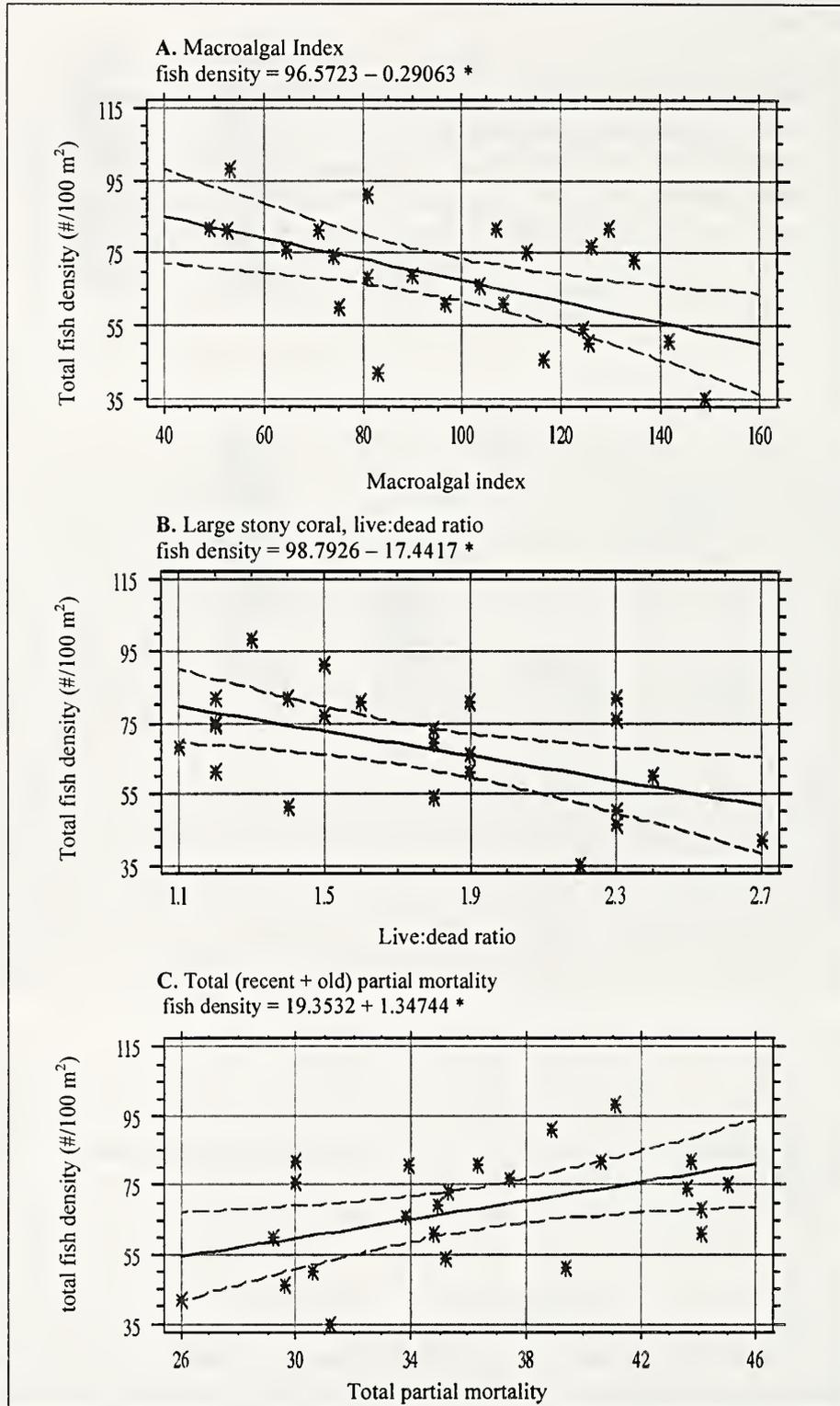
Classification analysis divided the sites into two large clusters according to the affinity criteria based on total fish abundances. The first cluster of 10 subreefs includes only sites located in the NSK and SSK areas. The second cluster includes all nine of the subreefs of the S area, three of the NSK area and El Placer S subreef (SSK area). Tampalam N subreef was markedly different from all the other survey sites (Fig. 7).



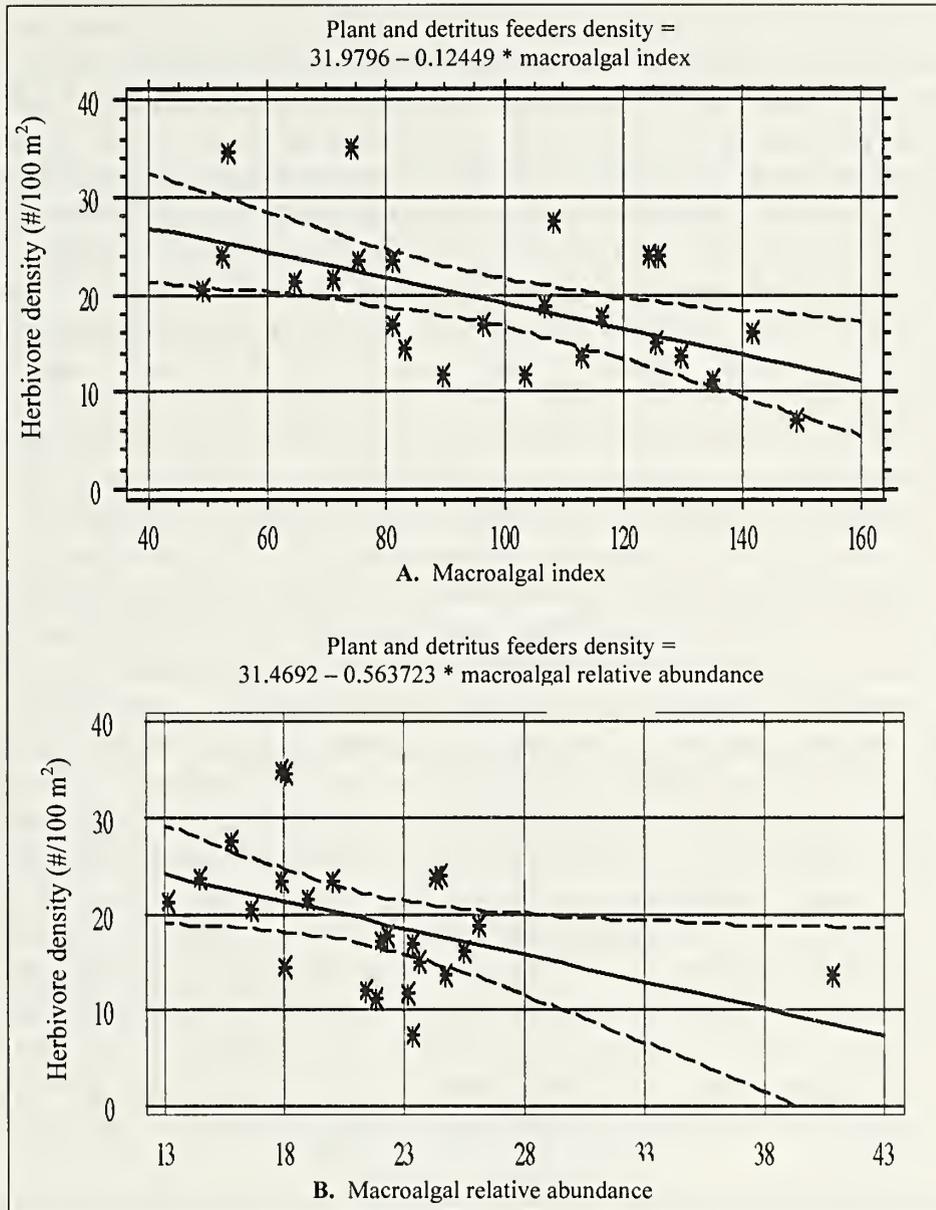
**Figure 7.** Hierarchical classification analysis based on total fish abundance, by subreef at 12 m in central-southern Quintana Roo, Mexico.

## Relationships

Multiple regression analyses between the total density of all the fish species and the benthic habitat variables (Ruiz et al., this volume) showed statistically significant inverse relationships ( $P < 0.05$ ) with the macroalgal index, live/dead stony coral ratio, and total (recent + old) partial-colony mortality (Fig. 8). The  $r^2$  statistic indicates that the model of multiple regression, when all six benthic variables are included, explains 86.96% of the variability in the fish density data. Statistically significant, inverse relationships were also found between the means for the total plant and detritus guild and those for the macroalgal index and relative abundance of macroalgae (Fig. 9).



**Figure 8.** Regression plot and 95% confidence intervals between mean total fish density (no. individuals/100m<sup>2</sup>) and (A) mean macroalgal index ( $P < 0.01$ , 29.2% of the variability in fish density explained), and for large ( $\geq 25$  cm in diameter) stony corals (B) mean live:dead ratio ( $P < 0.05$ , 26.3% explained), (C) mean total (recent + old) partial colony mortality ( $P < 0.05$ , 22.5% explained) by subreef at 12 m in central-southern Quintana Roo, Mexico.



**Figure 9.** Regression plot and 95% confidence interval between mean herbivore (plant and detritus feeders) density (no. individuals/100m<sup>2</sup>) and (A) mean macroalgal index ( $P=0.005 < 0.01$ , 30.3% of the variability in fish density explained) and (B) mean macroalgal relative abundance ( $P=0.03 < 0.05$  19.5% explained), by subreef at 12 m in central-southern Quintana Roo, Mexico.

## DISCUSSION

The principal factors involved in the evolution and maintenance of coral reef fish community structure are historical, biogeographical, geomorphological and bio-ecological (Harmelin-Vivien, 1989). The present study suggests the participation of some of these factors in the regulation of fish communities in the Mexican Caribbean as well as human intervention in the form of both environmental protection and deterioration.

The ecological descriptors used to describe reef fish community structure (species richness, abundance, density) showed a gradient from greater to lesser running from north to south along the Mexican Caribbean coast. The grouping of subreefs in the cluster analysis also approximately followed this pattern. This gradient not only coincides with latitudinal variation but with variation in protection against human exploitation. Indeed, Arias-González's (1998) trophic structural analysis had previously demonstrated that top-level fish production could be two to three times higher in relatively unfished Mexican Caribbean reefs than in those which are unprotected. At first we thought the degree of conservation was the most obvious explanation for the observed geographic gradient found during these surveys. However, it is also possible that other factors aided in determining this pattern.

It is well known that coral reefs have a fragmented environmental distribution and are characterized by diverse substrate types and complexity. The biology of reef fishes is set to the multi-scalar coral reef systems by ecological processes that act upon them and by the architectural patchiness of the reef environment (Sale, 1998). Geomorphological and habitat structural features may be an alternative explanation for the differences found in fish communities, at least for one of the spatial scales investigated in this study. The greatest numbers of species and families were recorded in the NSK reefs, especially in Punta Yuyum and Boca Paila, where coral reef structures visually appear to have a high degree of topographical complexity because the spurs are of high relief, and are covered with a large variety of benthic fauna. Conditions here naturally appear to be particularly favorable for the establishment and persistence of resident reef fishes. At the same time, the Boca Paila and Punta Yuyum reefs receive the least amount of anthropogenic disturbance. Therefore, the fact that these two reefs had higher fish species and family richness, along with higher abundances and larger sized carnivores than those in the SSK and S areas, could be a response to a combination of favorable geomorphological and human factors. Similarly, the relatively low species richness and abundances observed at Punta Allen, which is also located within the NSK, may be explained as a response either to the effects of fishing activity in the community of Rojo Gómez (Punta Allen) and/or to the natural influence of the large freshwater masses associated with La Ascención and Espíritu Santo Bays.

The Tampalam and El Placer reefs in the SSK area experience reduced human activity in the form of few fishing boats (<five per locality) and low numbers of fishermen (5-10 per locality). This fishing activity sometimes occurs immediately over the coral reefs, which can affect fish community structure directly by decreasing the abundance of top predators and indirectly by causing damage to the habitat with fishing gear (Russ and Alcala 1996). Nevertheless, the most probable explanation for the reduced numbers of fish species and individuals at Tampalam was the low structural complexity clearly seen in this system. The spur-and-groove formations on the fore reef were not continuous along the coast, being interrupted by a flat calcareous substratum at Tampalam N, with consequent reduction of suitable habitat spaces for fishes. We conclude that fishing activity was not sufficiently intense and frequent at Tampalam to modify the reef fish community structure. However, habitat structure appeared to be a determining factor, not only of differences among the geographical areas but between the reefs within the SSK area and among the subreefs at Tampalan (Fig. 7).

Fishing and seasonal tourism activities are highest in the S area as reflected in the lower number of fish species and families relative to the NSK and SSK areas. Evident effects of fishing were the loss of intermediate- to large-sized fishes (mainly carnivores) and the clear dominance of the trophic structure by species of plant and detritus feeders. The depletion of large top predators can modify the community structure of reef fishes via an increase in the abundance, size, and biomass of fish prey (Russ, 1991; Jennings and Polunin, 1996; McClanahan, 1997). The three reefs located in the S area are subject to comparable levels of fishing exploitation and have similar geomorphological and habitat structures which probably explains the similar values for their reef fish community descriptors.

A general trend in the size structure of fish species at all three spatial scales was the high abundance of plant and detritus feeders that were <20 cm in length and of carnivores that were between 10 and 30 cm long. As the size of the plant and detritus feeders was greatest in the S area, where large predatory fishes were particularly scarce, their greater lengths here could also be a result of local fishing practices.

The significant inverse relationship that we found between plant and detritus feeders and macroalgae indicates that herbivorous fishes have a measurable effect on the abundance of this important algal group in the Mexican Caribbean. Hence, overfishing of top predators may have ecological effects that cascade through reef ecosystems.

Overall, the results of this work suggest that the general condition and spatial patterns found in the ecological descriptors of reef fish communities at the different spatial scales studied were partially due to effects of human activities (mainly fishing) and partially attributable to geomorphological and habitat structure characteristics. Studies evaluating the condition of reefs and their fish communities appear to be an efficient management tool, given the need for rapid, integrated information useful in short-term planning of coastal resource administration projects. The systematic continuation of this kind of assessment will be beneficial for social, economic, and, of course, environmental purposes.

## ACKNOWLEDGMENTS

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Table 1. Site information for AGRRA fish surveys in central-southern Quintana Roo, México.

Site name	Site code	Reef type	Latitude (°' " N)	Longitude (°' " W)	Survey date	Depth (m)	>25 cm stony corals (#/10 m) <sup>1</sup>	% live stony coral cover (mean ± sd) <sup>1</sup>	50 m fish transects (#)	Species in transects (#) AGRRA <sup>2</sup>	Total	
<b>Northern Sian Ka'an</b>												
Boca Paila North	BPN	Fringing	20 06 51	87 27 23	Aug.30 99	12	6	15.5 ± 4.5	6	30	49	
Boca Paila Center	BPC	Fringing	20 06 21	87 27 34	Aug.31 99	12	6	19.0 ± 9.5	6	24	42	
Boca Paila South	BPS	Fringing	20 05 51	87 27 47	Aug.31 99	12	2	8.5 ± 2.5	6	30	52	
Punta Yuyum North	PYN	Fringing	19 58 30	87 27 10	Aug.27 99	12	4.5	10.5 ± 5.5	6	21	41	
Punta Yuyum Center	PYC	Fringing	19 58 00	87 27 06	Aug.26 99	12	4	12.5 ± 6.0	6	33	62	
Punta Yuyum South	PYS	Fringing	19 57 30	87 26 52	Aug.25 99	12	4.5	10.5 ± 6.0	6	23	48	
Punta Allen North	PAN	Fringing	19 50 30	87 26 15	Aug.20 99	12	4.5	11.5 ± 3.5	6	27	50	
Punta Allen Center	PAC	Fringing	19 50 00	87 26 36	Aug.21 99	12	4.5	14.5 ± 7.5	6	25	52	
Punta Allen South	PAS	Fringing	19 49 30	87 26 52	Aug.22 99	12	3.5	10.5 ± 4.5	6	25	48	
<b>Southern Sian Ka'an</b>												
Tampalam North	TN	Fringing	19 09 15	87 32 00	Sep.30 99	12	1.5	8.5 ± 5.5	6	5	29	
Tampalam Center	TC	Fringing	19 08 45	87 32 10	Sep.30 99	12	5	15.0 ± 9.0	6	11	45	
Tampalam South	TS	Fringing	19 08 15	87 32 13	Sep.29 99	12	6	18.0 ± 6.5	6	15	44	
El Placer North	EPN	Fringing	18 54 48	87 37 03	Sep.25 99	12	2	6.0 ± 2.5	6	27	45	
El Placer Center	EPC	Fringing	18 54 08	87 37 24	Sep.24 99	12	2.5	9.0 ± 3.5	6	21	41	
El Placer South	EPS	Fringing	18 53 38	87 37 36	Sep.23 99	12	4	12.0 ± 4.0	6	25	51	
<b>Southern</b>												
Mahahual North	MN	Fringing	18 43 05	87 41 56	Jun.27 99	12	5	17.0 ± 7.0	6	24	39	
Mahahual Center	MC	Fringing	18 42 35	87 42 09	Jun.26 99	12	4.5	17.0 ± 6.0	6	23	45	
Mahahual South	MS	Fringing	18 42 05	87 42 20	Jun.25 99	12	6	16.5 ± 5.5	6	27	48	
Xahuayxol North	XN	Fringing	18 30 55	87 45 02	Jul.20 99	12	3.5	11.5 ± 3.0	6	31	55	
Xahuayxol Center	XC	Fringing	18 30 25	87 45 13	Jul.21 99	12	2.5	12.0 ± 5.5	6	25	41	
Xahuayxol South	XS	Fringing	18 29 55	87 45 22	Jul.22 99	12	3.5	11.5 ± 5.5	6	26	45	
Xcalak North	XCN	Fringing	18 13 43	87 49 51	Jul.13 99	12	3.5	9.0 ± 4.5	6	32	54	
Xcalak Center	XCC	Fringing	18 13 09	87 49 54	Jul.14 99	12	3.5	9.5 ± 3.0	6	28	45	
Xcalak South	XCS	Fringing	18 12 39	87 49 47	Jul.15 99	12	3	7.5 ± 4.5	6	31	45	

<sup>1</sup>Data from Ruiz et al. (this volume); <sup>2</sup>Excluding any *Epinephelus cruentatus*.

Table 2. Sighting frequency and mean density of the 25 most frequently sighted fish species in "all species" belt transect surveys in central-southern Quintana Roo, México.

\* = AGRRA species.

Species	Sighting frequency (%) <sup>2</sup>	Density (#/100m <sup>2</sup> )
<i>Thalassoma bifasciatum</i>	97	9.76
* <i>Acanthurus coeruleus</i>	97	2.68
* <i>Sparisoma aurofrantum</i>	94	2.35
<i>Halichoeres garnoti</i>	93	2.00
<i>Chromis cyanea</i>	92	9.39
<i>Stegastes partitus</i>	92	4.00
* <i>Acanthurus bahianus</i>	85	1.82
* <i>Sparisoma viride</i>	84	1.88
* <i>Haemulon flavolineatum</i>	78	1.64
<i>Stegastes fuscus</i>	76	1.31
* <i>Scarus iserti</i> (= <i>S. croicensis</i> ) <sup>1</sup>	74	2.96
* <i>Haemulon sciurus</i>	69	2.15
* <i>Epinephelus fulvus</i>	68	0.94
* <i>Ocyurus chrysurus</i>	67	1.06
* <i>Sparisoma chrysopterum</i>	66	1.04
* <i>Microspathodon chrysurus</i>	65	0.90
<i>Stegastes leucostictus</i>	64	0.85
* <i>Haemulon plumieri</i>	60	0.85
* <i>Holacanthus tricolor</i>	52	0.55
* <i>Scarus taeniopterus</i>	51	1.01
<i>Holocentrus adscensionis</i>	45	0.63
* <i>Anisotremus virginicus</i>	41	1.03
* <i>Bodianus rufus</i>	39	0.50
<i>Stegastes variabilis</i>	37	0.52
<i>Gramma loreto</i>	28	0.69

<sup>1</sup>Species names according to Eschmeyer's (1998) revision.

<sup>2</sup>Sighting frequency (%) = percentage of transects in which the species was recorded.

Table 3. Density (mean  $\pm$  standard deviation) of AGRRA fishes, by subreef in central-southern Quintana Roo, México.

Site name	Herbivores (#/100m <sup>2</sup> )		Carnivores (#/100m <sup>2</sup> )		
	Acanthuridae	Scaridae ( $\geq 3$ cm)	Haemulidae ( $\geq 3$ cm)	Lutjanidae	Serranidae <sup>1</sup>
<b><i>Northern Sian Ka'an</i></b>					
Boca Paila N	4.7 $\pm$ 2.4	5.8 $\pm$ 3.5	3.5 $\pm$ 1.5	2.7 $\pm$ 3.1	2.0 $\pm$ 0.8
Boca Paila C	8.0 $\pm$ 7.3	2.0 $\pm$ 1.5	11.7 $\pm$ 8.1	3.0 $\pm$ 3.3	0.7 $\pm$ 1.0
Boca Paila S	7.7 $\pm$ 6.8	3.8 $\pm$ 1.8	13.3 $\pm$ 10.4	6.8 $\pm$ 1.2	1.2 $\pm$ 2.5
Yuyum N	7.5 $\pm$ 2.0	8.0 $\pm$ 5.5	12.0 $\pm$ 7.6	2.5 $\pm$ 2.6	1.5 $\pm$ 2.0
Yuyum C	4.7 $\pm$ 3.7	8.2 $\pm$ 2.5	24.3 $\pm$ 14.9	9.5 $\pm$ 7.6	2.0 $\pm$ 2.5
Yuyum S	4.8 $\pm$ 5.5	11.0 $\pm$ 4.0	7.3 $\pm$ 4.1	1.2 $\pm$ 0.8	1.2 $\pm$ 0.5
Punta Allen N	3.2 $\pm$ 5.1	9.8 $\pm$ 5.7	2.0 $\pm$ 1.2	1.3 $\pm$ 2.2	1.8 $\pm$ 0.8
Punta Allen C	4.2 $\pm$ 1.5	9.8 $\pm$ 5.7	3.0 $\pm$ 1.9	1.0 $\pm$ 0.6	2.3 $\pm$ 2.0
Punta Allen S	3.5 $\pm$ 2.5	15.7 $\pm$ 12.1	6.3 $\pm$ 8.7	1.3 $\pm$ 1.5	1.3 $\pm$ 0.8
<b><i>Southern Sian Ka'an</i></b>					
Tampalam N	5.0 $\pm$ 4.0	2.0 $\pm$ 2.0	5.0 $\pm$ 0.5	0	1.7 $\pm$ 4.0
Tampalam C	4.7 $\pm$ 9.7	6.5 $\pm$ 5.0	7.0 $\pm$ 4.4	2.5 $\pm$ 1.3	0.3 $\pm$ 0.8
Tampalam S	4.0 $\pm$ 8.0	10.8 $\pm$ 5.9	9.0 $\pm$ 6.7	2.7 $\pm$ 2.2	0.2 $\pm$ 0
El Placer N	3.8 $\pm$ 4.2	5.2 $\pm$ 2.9	15.5 $\pm$ 18.7	0.7 $\pm$ 0.5	2.3 $\pm$ 4.2
El Placer C	1.8 $\pm$ 0.8	9.0 $\pm$ 5.1	4.0 $\pm$ 2.5	0.2 $\pm$ 0.6	1.0 $\pm$ 2.0
El Placer S	3.8 $\pm$ 3.7	12.2 $\pm$ 7.4	6.3 $\pm$ 5.7	1.8 $\pm$ 1.5	1.8 $\pm$ 3.5
<b><i>Southern</i></b>					
Mahahual N	2.5 $\pm$ 0.5	15.0 $\pm$ 12.0	1.0 $\pm$ 1.3	1.2 $\pm$ 0	1.5 $\pm$ 1.3
Mahahual C	2.5 $\pm$ 3.5	21.8 $\pm$ 17.8	2.3 $\pm$ 1.5	0.3 $\pm$ 0	0.8 $\pm$ 0.8
Mahahual S	3.3 $\pm$ 3.5	25.0 $\pm$ 13.4	2.3 $\pm$ 1.0	1.0 $\pm$ 0.6	1.5 $\pm$ 0.9
Xahuayxol N	6.7 $\pm$ 7.7	13.0 $\pm$ 4.7	2.8 $\pm$ 1.3	1.0 $\pm$ 10.3	1.8 $\pm$ 0.6
Xahuayxol C	4.5 $\pm$ 4.6	8.3 $\pm$ 4.8	2.5 $\pm$ 0.8	1.0 $\pm$ 0.6	2.8 $\pm$ 6.5
Xahuayxol S	6.2 $\pm$ 4.8	11.3 $\pm$ 6.7	7.5 $\pm$ 6.0	2.0 $\pm$ 3.0	2.3 $\pm$ 3.5
Xcalak N	4.0 $\pm$ 4.0	13.2 $\pm$ 5.1	2.7 $\pm$ 0.6	6.7 $\pm$ 12.4	1.7 $\pm$ 1.0
Xcalak C	5.0 $\pm$ 3.3	10.0 $\pm$ 4.3	4.3 $\pm$ 3.0	2.7 $\pm$ 3.5)	1.2 $\pm$ 2.5
Xcalak S	16.7 $\pm$ 22.4	14.2 $\pm$ 6.4	3.0 $\pm$ 1.1	4.7 $\pm$ 6.0	2.3 $\pm$ 2.7

<sup>1</sup>*Epinephelus* spp. (excluding any *E. cruentatus*) and *Mycteroperca* spp.

Table 4. Mean percentage of species and individuals in Randall's (1967) reef fish feeding categories by reef in central-southern Quintana Roo, México.

Reef name	Plant and detritus feeders (%)		Zooplankton feeders (%)		Invertebrate feeders				Generalized carnivores (%)		Ectoparasite feeders (%)		Fish feeders (%)			
	Spp	Ind <sup>2</sup>	Spp	Ind	sessile (%)	Spp	Ind	"shelled" (%)	Spp	Ind	Spp	Ind	Spp	Ind	Spp	Ind
<b>Northern Sian Ka'an</b>																
Boca Paila	24	17	10	37	12	16	23	16	19	14	1	1	10	3		
Punta Yuyum	20	23	11	30	13	15	22	26	15	11	4	1	10	2		
Punta Allen	23	32	14	38	16	7	18	11	16	7	4	1	8	4		
All Northern Sian Ka'an	<b>23</b>	<b>24</b>	<b>12</b>	<b>36</b>	<b>12</b>	<b>12</b>	<b>21</b>	<b>18</b>	<b>16</b>	<b>11</b>	<b>3</b>	<b>1</b>	<b>7</b>	<b>4</b>		
<b>Southern Sian Ka'an</b>																
Tampalam	27	33	13	32	9	5	24	20	13	7	9	2	3	3		
El Placer	25	18	13	50	13	5	19	19	13	3	6	1	10	4		
All Southern Sian Ka'an	<b>24</b>	<b>26</b>	<b>13</b>	<b>41</b>	<b>11</b>	<b>5</b>	<b>22</b>	<b>20</b>	<b>13</b>	<b>5</b>	<b>1</b>	<b>7</b>	<b>7</b>	<b>3</b>		
<b>Southern</b>																
Mahahual	22	39	11	37	17	9	19	9	18	3	1	1	10	3		
Xahuayxol	25	32	13	38	17	7	21	15	17	4	1	1	6	4		
Xcalak	22	30	10	48	19	4	24	9	15	4	1	1	9	6		
All Southern	<b>24</b>	<b>34</b>	<b>11</b>	<b>41</b>	<b>18</b>	<b>6</b>	<b>22</b>	<b>10</b>	<b>17</b>	<b>4</b>	<b>1</b>	<b>1</b>	<b>8</b>	<b>4</b>		

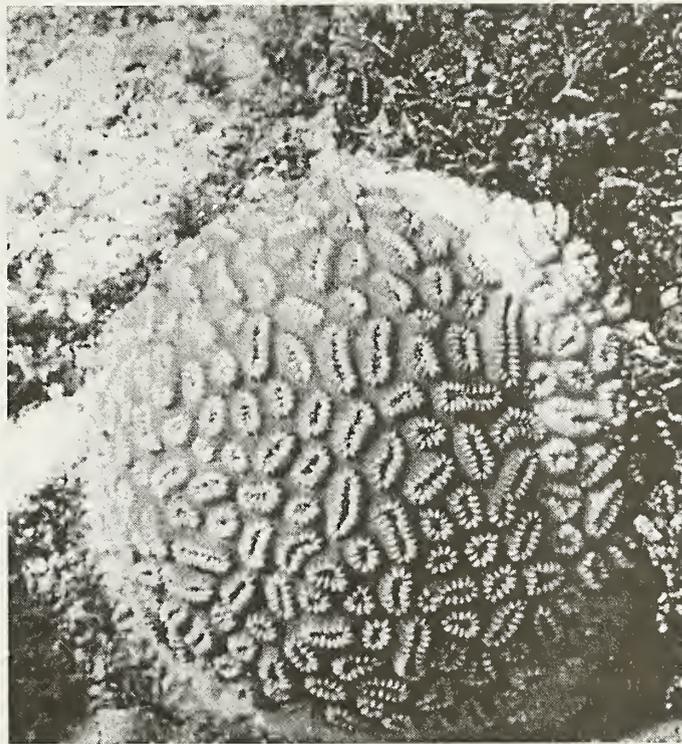
<sup>1</sup>Spp = Species

<sup>2</sup>Ind = Individuals

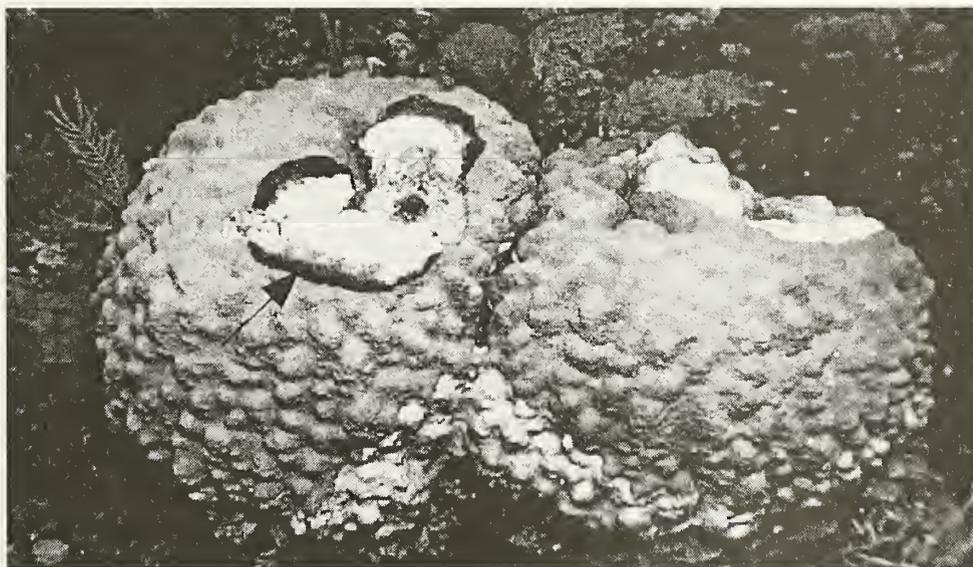
Table 5. Mean abundance by size category for herbivorous and carnivorous reef fishes (all species and AGRRA list) by reef in central-southern Quintana Roo, México.

Reef name	Fish trophic category	Abundance (#/reef)				
		3-10 cm	11-20 cm	21-30 cm	31-40 cm	41-50 cm
<i>Northern Sian Ka'an</i>						
Boca Paila	All herbivores	<b>219</b>	132	28	8	7
	AGRRA herbivores <sup>1</sup>	35	<b>128</b>	28	4	4
	All carnivores	2	<b>70</b>	29	12	6
	AGRRA carnivores <sup>1</sup>	1	<b>52</b>	12	8	3
Yuyum	All herbivores	<b>195</b>	163	33	7	0
	AGRRA herbivores	76	<b>163</b>	33	7	0
	All carnivores	9	<b>86</b>	27	6	1
	AGRRA carnivores	3	<b>65</b>	10	1	0
Punta Allen	All herbivores	<b>176</b>	188	21	7	2
	AGRRA herbivores	92	<b>184</b>	20	7	1
	All carnivores	0	<b>25</b>	19	5	5
	AGRRA carnivores	0	5	<b>10</b>	5	5
<i>Southern Sian Ka'an</i>						
Tampalam	All herbivores	<b>204</b>	111	14	9	3
	AGRRA herbivores	78	<b>107</b>	14	9	1
	All carnivores	0	<b>32</b>	16	3	3
	AGRRA carnivores	0	<b>21</b>	8	2	0
El Placer	All herbivores	<b>285</b>	99	35	7	0
	AGRRA herbivores	<b>97</b>	90	35	7	0
	All carnivores	0	<b>51</b>	12	6	8
	AGRRA carnivores	0	<b>14</b>	0	2	0
<i>Southern</i>						
Mahahual	All herbivores	173	<b>334</b>	65	4	0
	AGRRA herbivores	60	<b>306</b>	65	4	0
	All carnivores	4	<b>33</b>	11	2	3
	AGRRA carnivores	0	<b>9</b>	5	0	1
Xahuayxol	All herbivores	146	<b>218</b>	29	9	0
	AGRRA herbivores	76	<b>199</b>	29	9	0
	All carnivores	6	<b>51</b>	19	1	1
	AGRRA carnivores	0	<b>13</b>	10	0	0
Xcalak	All herbivores	181	<b>317</b>	35	11	0
	AGRRA herbivores	49	<b>316</b>	35	11	0
	All carnivores	6	<b>53</b>	50	25	4
	AGRRA carnivores	0	26	<b>34</b>	22	3
<b>All Reefs</b>	<b>All herbivores</b>	<b>197.4</b>	<b>195.3</b>	<b>32.5</b>	<b>7.8</b>	<b>1.5</b>
	<b>AGRRA herbivores</b>	<b>70.4</b>	<b>186.6</b>	<b>32.4</b>	<b>7.3</b>	<b>0.8</b>
	<b>All carnivores</b>	<b>3.4</b>	<b>50.1</b>	<b>22.9</b>	<b>7.5</b>	<b>3.9</b>
	<b>AGRRA carnivores</b>	<b>0.4</b>	<b>22.8</b>	<b>9.9</b>	<b>4.4</b>	<b>1.3</b>

<sup>1</sup>See Methods for AGRRA species as defined in this paper.



**Plate 8A.** Stony coral diseases are recorded in the AGRRA benthos protocol on the basis of color and host identity: white plague as shown in *Dichocoenia stokesi* and other massive corals; white-band and other white-type diseases in acroporids; black-band in many corals; yellow-blotch (yellow-band) in *Montastraea* spp. and brain corals; and dark spots (*Siderastrea siderea* and *Stephanocoenia intersepta*). (Photo George P. Schmahl)



**Plate 8B.** Diseases presently constitute some of the most severe disturbances on reefs in the wider Caribbean and they are increasing in frequency, intensity and distribution. Signs of disease were present in about 3% (at  $\leq 5$  m) and 5% (at  $> 5$  m) of the stony corals ( $\geq 25$  cm diameter) observed during the AGRRA surveys reported in this volume. Recent mortality in this colony of *Montastraea faveolata* was caused by black-band disease (arrow). (Photo Kenneth W. Marks)

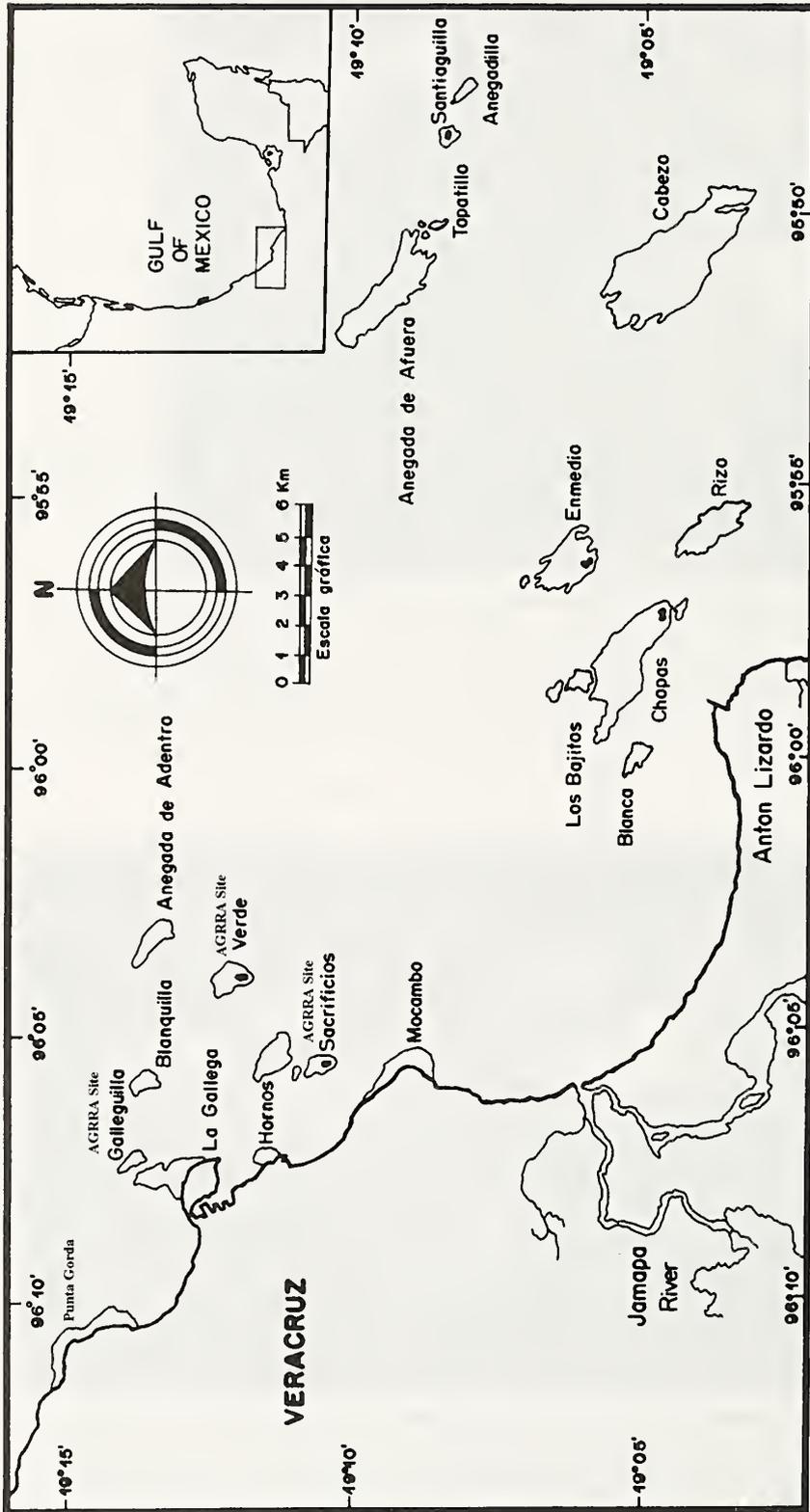


Figure 1. AGRRA survey sites in the Veracruz Reef System.

# CONDITION OF SELECTED REEF SITES IN THE VERACRUZ REEF SYSTEM (STONY CORALS AND ALGAE)

BY

GUILLERMO HORTA-PUGA<sup>1</sup>

## ABSTRACT

Three platform reefs off the city of Veracruz in the Gulf of Mexico were surveyed during July, 1999 with the Atlantic and Gulf Rapid Reef Assessment (AGRRA) benthos protocols at two depth intervals (3-6 m and 9-12 m) in windward fore-reef habitats. Live stony coral cover averaged 17%. Biodiversity was low with 15 taxa of "large" ( $\geq 25$  cm diameter) stony corals of which *Montastraea cavernosa* was numerically the most abundant (35%). Acroporids were almost completely absent. The condition of the large living corals was good with few signs of disease (none in the individually assessed colonies) and little bleaching. Crustose coralline algae were more abundant overall than turf algae while marcoalgae were scarce. As coral recruitment density was extremely low ( $\sim 1.2$  recruits/m<sup>2</sup>), the current potential for recovery of these reefs to historical levels of live coral cover seems poor, even though apparently suitable recruitment sites were available in most sites.

## INTRODUCTION

The 20 coral reefs of the Veracruz Reef System (VRS) are located within 22 km of the coast near the Port of Veracruz in the western Gulf of Mexico (Fig. 1). The reefs developed on the continental shelf after the last glacial period some 9,000 to 10,000 years ago (Morelock and Koenig 1967; Kuhlmann 1975). They have thrived in a naturally turbid environment. Visibility can be  $< 1$  m during the rainy season (June-October) when high concentrations of suspended particles (eroded materials from the mainland) are transported by several rivers (Antigua, Jamapa, Papaloapan) that discharge nearby. The area is also affected by several cold fronts each winter that decrease seawater temperature and increase surf and turbidity.

The VRS reefs have well-developed reef frameworks (Carricart-Ganivet et al., 1993). The windward fore-reef slopes, which face east and north, have the highest stony coral cover and extend to depths of about 30 m (Gutierrez et al., 1993; Vargas-Hernandez et al., 1993). Live stony coral cover is lower, and limited to depths of about 21 m, on the leeward (westward-facing) slopes which, however, have the highest diversity of scleractinians as certain ramose and delicate species only occur in these areas. The reef flats (also called platforms or lagoons) are well-illuminated, shallow (1.0-1.6 m in depth), and dominated by

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macroalgae, the seagrass, *Thalassia testudinum*, and some stony corals. The reef crests or rubble ramparts, which are located at the windward margins of the reef flats, are accumulations of dead coral boulders deposited by strong storm waves and currents (Kühlmann, 1975; Gutierrez et al., 1993).

As in other remote Atlantic reef areas like the Flower Garden Banks and Bermuda, the diversity of reef-building scleractinians is relatively depauperate with 30 zooxanthellate species in the VRS in contrast to the 47 found on Mexican Caribbean reefs in the eastern Yucatan Peninsula (Horta-Puga and Carricart-Ganivet, 1993; Beltrán-Torres and Carricart-Ganivet, 1999; Ruiz et al., this volume). Historically, the main species have been *Acropora cervicornis*, *A. palmata*, *Diploria clivosa*, *D. strigosa*, *Siderastrea radians*, *S. siderea*, *Montastraea annularis*, *M. cavernosa*, *M. faveolata* and *Colpophyllia natans* (Lara et al., 1992; Gutierrez et al., 1993). However, both species of *Acropora* experienced widespread mortality during the 1970s-1980s (Jordán, 1992; Tunnell, 1992), possibly at least in part from disease. At present the bottom is dominated by the long-dead skeletons of important genera (*Acropora*, *Montastraea*, *Colpophyllia*, *Diploria*, *Siderastrea*) which are still recognizable suggesting high mortality rates in recent decades. Populations of *Diadema antillarum* were possibly decimated during the 1983-84 mass mortality event which changed sea-urchin community structure throughout the Western Atlantic. Nowadays, species of *Echinometra* (*E. lucunter* and *E. viridis*) are among the most abundant herbivores in the VRS (see Results).

The reefs of the VRS have been considered to be among the most threatened in the wider Caribbean. Anthropogenic impacts in the area are diverse and include oil spills plus other chemical pollutants, overfishing, unrestricted recreational diving, ship groundings, sewage effluents, dredging, recreational diving, and boat anchoring (Tunnell, 1992; Lang et al., 1998). Environmental stressors have increased in magnitude in recent decades. In response to scientific concern, the entire VRS was decreed a marine protected area by the Mexican Government in 1992 (Diario Oficial, 1992). In 2000, the National Commission of Natural Protected Areas designated the staff of the marine park who are now actively developing a management plan as well as performing surveillance and conservation activities.

The aim of this study was to assess the current condition of the VRS using the AGRRA benthos protocols.

## METHODS

The VRS is divided into a northern and a southern group by the outlet of the Jamapa River. The northern group has three coastal fringing reefs and seven platform reefs rising 25-30 m above the bottom (Carricart-Ganivet and Horta-Puga, 1993). Three platform-type mid-shelf reefs (Fig. 1) were chosen to assess the benthic condition of the northern VRS reef system. Galleguilla was strategically chosen because it is considered highly threatened due to its position near the outlet of Veracruz City's sewage treatment plant and the port docks. Isla Verde, which is furthest offshore, and Isla de Sacrificios are both considered representative platform reefs. The latter is also a strategic site because its reef has been closed to all human activities related to tourism or fishing by local authorities since 1982 and was hoped to be in a process of recovery after having been highly degraded. The surveys were conducted in windward (eastern), fore-reef slope habitats. Two depth zones were selected: the shallow (3-6

m) former *Acropora* zone and a deeper (9-12 m) zone of maximum scleractinian diversity and coverage.

The AGRRA Version 2 benthos protocols (see Appendix One, this volume) were followed from 14-22 July 1999 with the following modifications: (1) colonies that were "standing dead" (completely dead and still in growth position) were not surveyed, i.e., individual assessments were restricted to "large" ( $\geq 25$  cm maximum diameter) corals with at least some living tissues; (2) coral measurements were made to the nearest 5 cm; (3) *Diploria*, *Siderastrea* and *Agaricia* were identified only to the genus level; and (4) *Echinometra* (*E. lucunter* + *E. viridis*) counts were added to the belt transects in all but the deep Isla de Sacrificios site. As the surveys were performed during the local rainy season, water turbidity was high. Visibility at any depth was usually  $< 5$  m, and often  $< 2$  m. In order to standardize the data, we spent a day assessing the same transects and quadrats until all team members consistently recorded similar results.

## RESULTS

### Stony Corals

A team of seven divers (four divers/dive) surveyed 723 large ( $\geq 25$  cm in diameter), living stony corals, examined 662 quadrats, and conducted 138 live stony coral cover transects plus 124 belt transects for sea urchin abundance. Live stony coral coverage varied from 15% in Galleguilla to 21% in Isla de Sacrificios, averaging 17% overall (Table 1). The remaining substratum mostly consisted of coral skeletons that were moderately intact to highly eroded.

We recorded 14 taxa of large living scleractinians and one hydrozoan (Table 2). The total number of taxa per reef varied from 6 in Galleguilla to 14 in Isla de Sacrificios, and from 12 in the deeper zones to 13 in the shallow zones. In terms of numerical abundance, *Montastraea cavernosa* was the predominant large species (overall mean = 35%), dominating both depths in Isla de Sacrificios and the shallow Galleguilla transects. *Colpophyllia natans* was the second most common overall (26.5%) and dominated both depths in Isla Verde. *Siderastrea* was the most common large taxon in the deep Galleguilla site. Depth had relatively little effect on mean numerical abundance with *M. cavernosa*  $>$  *C. natans*  $>$  *Diploria*  $>$  *Siderastrea* in the three shallow zones, and *M. cavernosa*  $>$  *C. natans*  $>$  *Siderastrea*  $>$  *Diploria* in the three deeper zones.

The average numbers of large living stony corals per 10 m transect varied from 5 in Galleguilla and Isla Verde to 6.5 in Isla de Sacrificios (Table 1). Their mean size (as maximum diameter) overall was 59 cm (Table 3). Bleached or partially bleached corals accounted for 3% of the overall total. Black-band disease (in *Siderastrea*, *Montastraea* spp.), tumor neoplasms (in *Diploria*) and dark spots disease (in *Siderastrea*) were seen in the VRS during the survey but no diseases were found in any of the surveyed corals.

Recent partial-colony mortality (hereafter recent mortality) of the large live stony corals averaged  $< 1\%$  of their upward-facing surfaces in five sites and was greatest (mean = 3%) in the shallow Isla de Sacrificios site. The percentage of affected corals varied from 0 to 13%, averaging 4% overall (Table 3). Average values for old partial-colony mortality (hereafter old mortality) varied between 6.5% at Galleguilla shallow and 13% at Isla Verde

shallow, with over 40% of all colonies having areas of old tissue loss. Trends in total (recent + old) mortality were similar to those for old mortality.

Coral recruitment averaged  $0.1/0.0625 \text{ m}^2$  ( $\sim 1.2/\text{m}^2$ ) and was low in all sites (Table 4). Sixty-seven recruits were counted: the numbers/taxa (and percent of total) were *Siderastrea* 43 (64%), *Oculina* 12 (18%), *Porites* and *Agaricia* 3 each (4%), *M. annularis* and *C. natans* 2 each (3%), *S. intersepta* and *M. complanata* 1 (1%) each.

### Algae and Echinoids

In general, crustose coralline algae (overall absolute abundance = 41%), followed by turf algae (overall absolute abundance = 26.5%), dominated the algal quadrats; however, turf algae were somewhat more abundant than crustose corallines at both depths off Galleguilla (Table 4). Macroalgae were notably scanty in the VRS fore-reef zones (overall absolute abundance = 0.5%) and, as their mean height was  $<1$  cm, the absolute macroalgal index (macroalgal abundance  $\times$  macroalgal height) was approximated as 0 to  $<1$ . On average, "barren areas" covered with sediment (terrigenous + endogenous) and lacking conspicuous sessile organisms occupied 13.5% of the substrata and, at the Isla de Sacrificios deeper site, they dominated the available substratum (60%).

Although present in the VRS, only two specimens of *Diadema antillarum* were seen during the survey and none were counted in the belt transects. Densities of *Echinometra* varied between  $10/100\text{m}^2$  in the deep Galleguilla site and  $155/100 \text{ m}^2$  in the shallow Isla de Sacrificios site, averaging  $65/100\text{m}^2$  overall (Table 4).

## DISCUSSION

Live stony coral cover was much higher during the mid 1960's when K hlmann (1975) recorded values of 50% in shallow and 40% in deep areas, respectively, of Blanquilla reef. At that time, *Acropora palmata* covered up to 65% of the available substrata in some shallow VRS reefs and the cover of *A. cervicornis* reached 100% in Enmedio reef (Ranefeld, 1972; K hlmann, 1975). Live *Acropora* spp. accounted for  $<1.5\%$  of the stony coral cover at 3-6 m in 1999. Overall, our data indicate that the VRS reefs had changed little since Tunnell's (1992) report on their condition when coral cover was 17% and 12% in the fore reefs at Enmedio and Cabezo Reefs, respectively. The reduced coral cover in the VRS reefs is a clear indication of serious ecosystem decline, even when it is recognized that nowadays an average of 17% is fairly typical of many Caribbean reef areas (see Kramer, this volume).

That the VRS mortality estimates for colony surfaces in large ( $\geq 25$  cm diameter) stony corals are low (especially for old mortality) in comparison to data collected elsewhere in the western Atlantic (Kramer, this volume) is partially explicable by our exclusion of standing dead corals from these assessments. Nevertheless, in spite of the scarcity of coral diseases and the relatively high availability of crustose coralline algae as a substratum for larval settlement (Morse et al., 1994), the chances for significant recovery of the VRS fore-reef communities seem poor so long as coral recruitment, especially by the broadcasting species that are so important in reef framework construction, is low.

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Table 1. Site information for AGRRA stony coral and algal surveys off Veracruz, México.

Reef name	Site code	Reef type <sup>1</sup>	Latitude (°' " N)	Longitude (°' " W)	Survey date(s)	Depth (m)	Benthic Transects (#0)	≥25 cm live stony corals (#/10 m) <sup>2</sup>	% liveness coral cover (mean ± sd) <sup>2</sup>
<b>Shallow</b>									
Galleguilla	GA3	Wind fore	19 13 53	96 07 37	July 20-22 99	3-6	24	4.5	16.0 ± 11.5
Isla de Sacrificios	IS3	Wind fore	19 10 26	96 05 32	July 18-19 99	3-6	22	7.0	19.5 ± 22.5
Isla Verde	IV3	Wind fore	19 11 50	96 04 06	July 15 99	3-6	28	4.0	14.5 ± 9.0
<b>All shallow</b>	<b>VR 3</b>					<b>3-6</b>	<b>74</b>	<b>5.0</b>	<b>16.5 ± 15.0</b>
<b>Deeper</b>									
Galleguilla	GA9	Wind fore	19 13 53	96 07 37	July 19-20 99	9-12	24	5.0	14.0 ± 8.0
Isla de Sacrificios	IS9	Wind fore	19 10 26	96 05 32	July 16-17 99	9-12	19	5.5	22.5 ± 15.0
Isla Verde	IV9	Wind fore	19 11 50	96 04 06	July 14-16 99	9-12	21	6.0	18.5 ± 8.0
<b>All deeper</b>	<b>VR 9</b>					<b>9-12</b>	<b>64</b>	<b>5.5</b>	<b>18.0 ± 11.0</b>
<b>All Galleguilla</b>	<b>All GA</b>					<b>3-6; 9-12</b>	<b>48</b>	<b>5.0</b>	<b>15.0 ± 10.0</b>
<b>All Isla de Sacrificios</b>	<b>All IS</b>					<b>3-6; 9-12</b>	<b>41</b>	<b>6.5</b>	<b>21.0 ± 19.5</b>
<b>All Isla Verde</b>	<b>All IV</b>					<b>3-6; 9-12</b>	<b>49</b>	<b>5.0</b>	<b>16.0 ± 8.0</b>
<b>All sites</b>	<b>All VR</b>					<b>3-6; 9-12</b>	<b>138</b>	<b>5.0</b>	<b>17.0 ± 13.5</b>

<sup>1</sup>Wind = windward<sup>2</sup>Standing dead corals were not surveyed.Table 2. Relative abundance and species richness of all live<sup>1</sup> stony corals (≥25 cm diameter) by site off Veracruz, México.

Species	Galleguilla %		Isla de Sacrificios %		Isla Verde %		All Reefs %		
	3-6 m	9-12 m	3-6 m	9-12 m	3-6 m	9-12 m	3-6m	9-12 m	Both depths
<i>M. alcicornis</i>		1.0	3.5		1.0	1.0	1.5	0.5	1.0
<i>A. cervicornis</i>						1.0		0.5	<0.5
<i>A. palmata</i>			1.5		9.0		3.0		1.5
<i>Agaricia</i> spp.			0.5	2.0			0.5	0.5	0.5
<i>C. natans</i>	20.5	10.0	12.5	9.0	50.0	60.0	25.5	27.5	26.5
<i>Diploria</i>	4.5	1.0	19.5	5.0	23.0	18.0	16.0	8.5	12.5
<i>M. decactis</i>			0.5				0.5		<0.5
<i>M. annularis</i>				6.0	5.5	4.5	1.5	3.5	2.5
<i>M. cavernosa</i>	59.5	30.0	49.0	56.0	7.5	8.5	40.0	29.5	35.0
<i>M. faveolata</i>				6.0	2.0	1.0	0.5	2.0	1.0
<i>M. franksi</i>				1.0				0.5	<0.5
<i>O. diffusa</i>			0.5				0.5		<0.5
<i>P. astreoides</i>			2.0			1.0	1.0	0.5	0.5
<i>Siderastrea</i>	13.5	58.5	10.5	11.0	3.0	4.5	9.0	25.5	17.0
<i>S. intersepta</i>	2.0			4.0		1.0	0.5	1.5	1.0
<b>All taxa (#)</b>	<b>5</b>	<b>5</b>	<b>10</b>	<b>9</b>	<b>8</b>	<b>10</b>	<b>13</b>	<b>12</b>	<b>15</b>

<sup>1</sup>Standing dead corals were not surveyed.

Table 3. Size and condition (mean  $\pm$  standard deviation) of all live<sup>1</sup> stony corals ( $\geq 25$  cm diameter) by site off Veracruz, México.

Site code <sup>2</sup>	Live stony corals		Partial-colony mortality of live stony corals (%)			Live stony corals (%)				
	#	Diameter (cm)	Recent	Old	Total	Recent mortality	Old mortality	Total mortality	Bleached	Diseased
<i>Shallow</i>										
GA3	111	67 $\pm$ 33	<0.5 $\pm$ 1.0	6.5 $\pm$ 11.5	6.5 $\pm$ 11.5	1.0	34.0	35.0	2.0	0
IS3	153	53 $\pm$ 34	3.0 $\pm$ 9.0	7.0 $\pm$ 13.0	10.0 $\pm$ 14.5	13.0	35.5	48.5	4.5	0
IV3	109	59 $\pm$ 23	0.5 $\pm$ 3.5	13.0 $\pm$ 18.0	13.5 $\pm$ 18.0	3.5	55.0	58.5	0	0
<b>All VR 3</b>	<b>373</b>	<b>59 <math>\pm</math> 31</b>	<b>1.5 <math>\pm</math> 6.5</b>	<b>7.0 <math>\pm</math> 12.0</b>	<b>8.5 <math>\pm</math> 13.5</b>	<b>6.5</b>	<b>41.0</b>	<b>47.5</b>	<b>2.5</b>	<b>0</b>
<i>Deeper</i>										
GA9	123	53 $\pm$ 28	0	9.0 $\pm$ 12.0	9.0 $\pm$ 12.0	0	49.0	49.0	5.5	0
IS9	100	75 $\pm$ 65	<0.5 $\pm$ 1.0	11.5 $\pm$ 17.0	11.5 $\pm$ 17.0	1.0	43.0	44.0	1.0	0
IV9	127	52 $\pm$ 22	<0.5 $\pm$ 4.0	6.5 $\pm$ 11.5	7.0 $\pm$ 12.0	3.0	36.0	39.5	2.5	0
<b>All VR 9</b>	<b>350</b>	<b>59 <math>\pm</math> 42</b>	<b>&lt;0.5 <math>\pm</math> 2.5</b>	<b>9.5 <math>\pm</math> 14.0</b>	<b>9.5 <math>\pm</math> 14.0</b>	<b>1.5</b>	<b>42.5</b>	<b>44.0</b>	<b>3.0</b>	<b>0</b>
All GA	234	59 $\pm$ 31	<0.5 $\pm$ 0.5	8.0 $\pm$ 12.0	8.0 $\pm$ 12.0	0.5	42.0	42.5	4.0	0
All IV	236	55 $\pm$ 53	0.5 $\pm$ 3.5	9.5 $\pm$ 15.0	10.0 $\pm$ 15.5	3.5	45.0	48.5	1.5	0
All IS	253	61 $\pm$ 50	2.0 $\pm$ 7.5	9.0 $\pm$ 14.5	10.5 $\pm$ 15.5	8.5	38.5	46.5	3.0	0
<b>All VR</b>	<b>723</b>	<b>59 <math>\pm</math> 37</b>	<b>&lt;0.5 <math>\pm</math> 0.5</b>	<b>8.5 <math>\pm</math> 12.5</b>	<b>8.5 <math>\pm</math> 12.5</b>	<b>4.0</b>	<b>41.5</b>	<b>46.0</b>	<b>3.0</b>	<b>0</b>

<sup>1</sup>Standing dead corals were not surveyed.<sup>2</sup>.See Table 1 for site codes.

Table 4. Algal characteristics, density of sessile invertebrates, sediment, stony coral recruits and echinoderms (mean  $\pm$  standard deviation) by site off Veracruz, México.

Site code <sup>1</sup>	Quadrats (#)	Absolute abundance (%)					Macroalgae		Recruits (#/.0625 m <sup>2</sup> )	Echinoid transects (#)	<i>Echinometra</i> (#/100 m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )
		Turf algae	Crustose Coralline algae	Sessile Invertebrates	Sediment	Height (cm)	Index <sup>2</sup>					
<b>VR 3</b>												
GA3	105	36.5 $\pm$ 27.0	32.0 $\pm$ 20.5	28.5 $\pm$ 23.0	3.5 $\pm$ 9.0	0	0	0.2 $\pm$ 0.4	28	62 $\pm$ 30	0	
IS3	117	23.0 $\pm$ 20.0	37.5 $\pm$ 24.5	26.5 $\pm$ 24.0	12.0 $\pm$ 22.0	<1.0	<1	0.0 $\pm$ 0.1	27	155 $\pm$ 154	0	
IV3	140	19.5 $\pm$ 20.0	56.5 $\pm$ 35.5	18.0 $\pm$ 19.0	9.5 $\pm$ 18.0	0	0	0.0 $\pm$ 0.2	21	25 $\pm$ 39	0	
<b>All VR 3</b>	<b>362</b>	<b>25.5 <math>\pm</math> 23.5</b>	<b>43.0 <math>\pm</math> 30.0</b>	<b>24.0 <math>\pm</math> 22.0</b>	<b>8.5 <math>\pm</math> 18.0</b>	<b>&lt;1.0</b>	<b>&lt;0.5</b>	<b>0.1 <math>\pm</math> 0.2</b>	<b>76</b>	<b>79 <math>\pm</math> 107</b>	<b>0</b>	
<b>VR 9</b>												
GA9	115	49.5 $\pm$ 17.5	37.5 $\pm$ 16.0	9.5 $\pm$ 9.5	0.5 $\pm$ 6.5	<1.0	<1	0.1 $\pm$ 0.3	21	10 $\pm$ 16	0	
IS9	80	13.0 $\pm$ 18.0	14.5 $\pm$ 18.5	13.5 $\pm$ 15.0	60.0 $\pm$ 36.0	0	0	0.1 $\pm$ 0.3	ND <sup>3</sup>	ND	0	
IV9	105	15.5 $\pm$ 22.5	26.5 $\pm$ 31.5	19.0 $\pm$ 25.0	10.0 $\pm$ 16.5	0	0	0.0 $\pm$ 0.2	24	72 $\pm$ 86	0	
<b>All VR 9</b>	<b>300</b>	<b>0.5 <math>\pm</math> 5.0</b>	<b>39.0 <math>\pm</math> 28.5</b>	<b>14.0 <math>\pm</math> 18.0</b>	<b>19.5 <math>\pm</math> 26.0</b>	<b>&lt;1.0</b>	<b>0.5</b>	<b>0.1 <math>\pm</math> 0.3</b>	<b>45</b>	<b>39 <math>\pm</math> 67</b>	<b>0</b>	
<b>All GA</b>	<b>220</b>	<b>0.5 <math>\pm</math> 4.0</b>	<b>36.0 <math>\pm</math> 18.5</b>	<b>18.5 <math>\pm</math> 19.5</b>	<b>1.5 <math>\pm</math> 6.0</b>	<b>&lt;1.0</b>	<b>0.5</b>	<b>0.1 <math>\pm</math> 0.4</b>	<b>49</b>	<b>34 <math>\pm</math> 36</b>	<b>0</b>	
<b>All IS</b>	<b>197</b>	<b>0.5 <math>\pm</math> 6.5</b>	<b>28.0 <math>\pm</math> 25.0</b>	<b>21.5 <math>\pm</math> 21.5</b>	<b>31.5 <math>\pm</math> 37.0</b>	<b>&lt;1.0</b>	<b>0.5</b>	<b>0.0 <math>\pm</math> 0.2</b>	<b>ND</b>	<b>ND</b>	<b>0</b>	
<b>All IV</b>	<b>245</b>	<b>0.5 <math>\pm</math> 4.0</b>	<b>56.5 <math>\pm</math> 33.5</b>	<b>18.5 <math>\pm</math> 22.0</b>	<b>9.5 <math>\pm</math> 17.5</b>	<b>0</b>	<b>0</b>	<b>0.0 <math>\pm</math> 0.2</b>	<b>45</b>	<b>45 <math>\pm</math> 67</b>	<b>0</b>	
<b>All VR</b>	<b>662</b>	<b>0.5 <math>\pm</math> 5.0</b>	<b>41.0 <math>\pm</math> 29.5</b>	<b>19.5 <math>\pm</math> 21.0</b>	<b>13.5 <math>\pm</math> 26.0</b>	<b>&lt;1.0</b>	<b>0.5</b>	<b>0.1 <math>\pm</math> 0.3</b>	<b>124</b>	<b>63 <math>\pm</math> 96</b>	<b>0</b>	

<sup>1</sup> See Table 1 for site codes.

<sup>2</sup> Macroalgal index = absolute macroalgal abundance x macroalgal height

<sup>3</sup> ND = not done

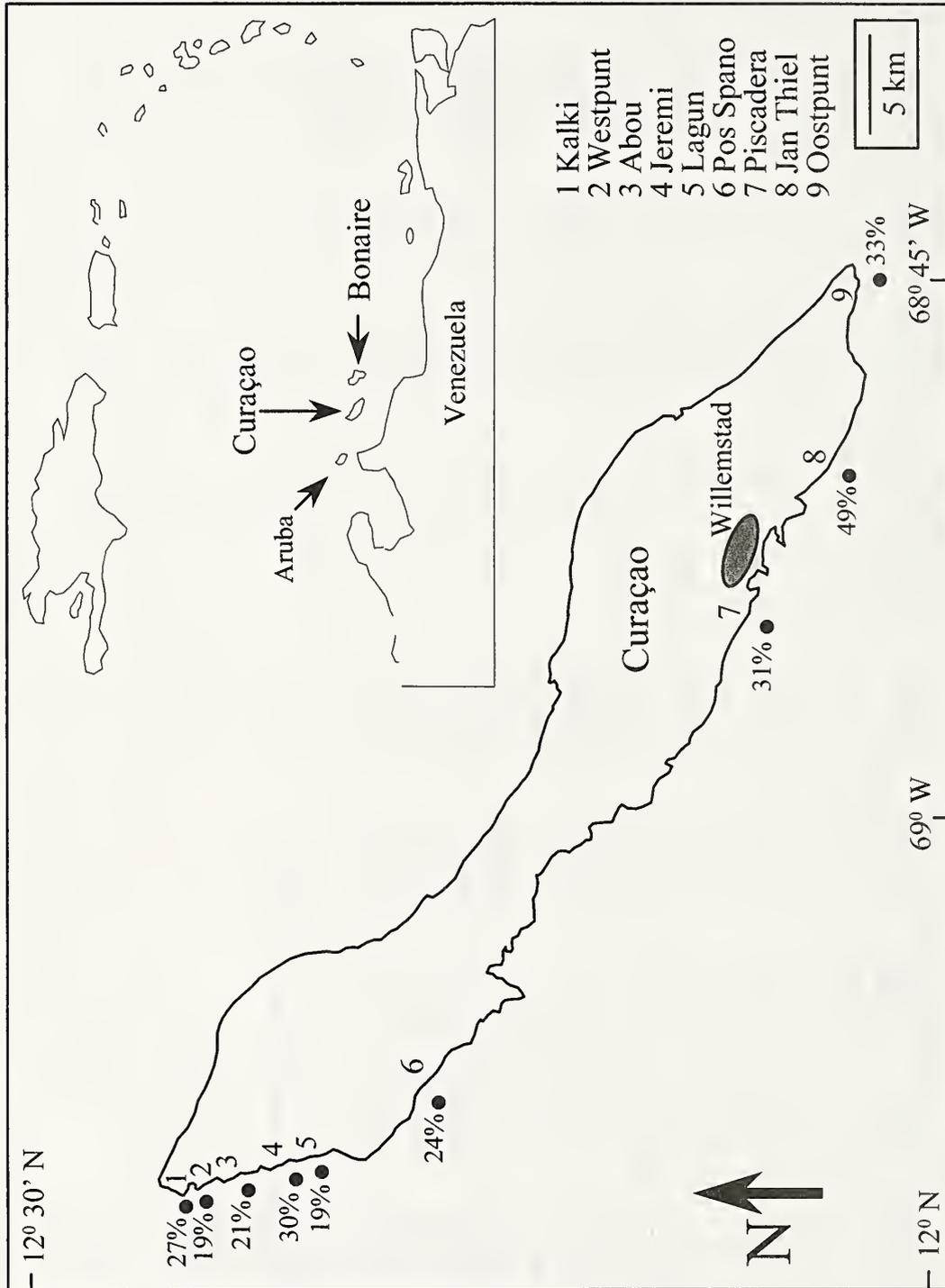


Figure 1. AGRRA survey sites (Kalki, Westpunt, Jeremi, Lagun, Oostpunt) on the leeward coast of Curaçao. Percentages = mean percentage of diseased stony corals (all species and all sizes) in belt transects at 10 m, 15m and 20 m that had been found in 1997 at these sites, and at four other reefs.

# CONDITION OF CORAL REEFS OFF LESS DEVELOPED COASTLINES OF CURAÇAO (PART 1: STONY CORALS AND ALGAE)

BY

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## ABSTRACT

Coral reefs at 10-20 m depth off eastern and western Curaçao, Netherlands Antilles had high abundance and high cover (25-50%) of stony corals, although the latter declined between 1998 and 2000, primarily from impacts associated with Hurricane Lenny in November 1999 and coral disease. Most corals had lost 15-40% of their live tissues and the amount of partial mortality declined with depth. Little recent mortality was observed (0.6% in 2000). Reefs were dominated by the *Montastraea annularis* species complex (46% of all corals  $\geq 20$  cm in diameter), which were 40% larger than other species. Overall, colonies of the *M. annularis* species complex sustained somewhat greater total (recent + old) partial-colony mortality (24%) than other stony corals (19%), and had a higher prevalence of disease. Yellow-blotch disease affected 14.5% of all colonies of the *M. annularis* species complex in January 2000; infected corals had twice as much total partial-colony mortality (44%) as uninfected conspecifics. Shallow reef communities at 8-12 m appear resilient to disturbance, as evidenced by low macroalgal cover, a high abundance of stony coral recruits and juveniles, and declining disease incidence and prevalence overall. However, the high incidence of yellow-blotch disease in the *M. annularis* species complex and the absence of recruits of these species suggests their condition may continue to decline and a shift in species dominance may be underway.

## INTRODUCTION

Curaçao, located 60 km north of Venezuela, forms part of the leeward Netherlands Antilles. The small oceanic island (61 km long; 443 km<sup>2</sup> total) is surrounded by fringing coral reefs, which are better developed along the leeward coast. The shallow reef community, which begins 20 m to 250 m from the shoreline, consists of a terrace that slopes gradually seaward to 7-13 m depth and

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then slopes steeply to a sand terrace at 50-60 m (Pors and Nagelkerken, 1998). Twenty-five years ago, the vertical reef profile was characterized as having a shallow *Acropora palmata* zone dominated by elkhorn coral and gorgonians, fields of *Acropora cervicornis* from 4-5 m depth, and a reef slope dominated by the *Montastraea annularis* species complex, *Agaricia* spp. and *Madracis mirabilis* (Bak, 1975). Coral cover and diversity were high on the reef slope, but decreased rapidly below 35-40 m. Fifty-seven species of scleractinian corals were identified by Bak (1975).

The coral reefs surrounding Curaçao are affected by a number of natural and anthropogenic stressors. The island is located south of the hurricane belt and rough seas are rare on the leeward coast. However, tropical storms pass within 200 km of the island about every four to five years, and associated wave surge has caused considerable damage to the shallow reefs (Van Duyl, 1985; Van Veghel and Hoetjes, 1995). Development and industry are concentrated in Willemstad and along the adjacent southeastern, leeward coast where the majority of Curaçao's population (155,000) resides. In these developed areas coral abundance, cover, and species diversity declined precipitously at 10-20 m on fore reefs between 1973 and 1992 (Bak and Nieuwland, 1995). Much of this change was attributed to sewage discharge and to sedimentation associated with beach construction (Bak and Nieuwland, 1995). The island-wide mass mortality of the herbivorous sea urchin *Diadema antillarum* in 1983 also contributed to a general decrease in the cover of live corals and coralline algae, while turf algae and macroalgae increased in abundance (Bak et al., 1984; De Ruyter van Steveninck and Bak, 1986). Branching acroporids (*A. palmata*, *A. cervicornis*) suffered high mortality in 1980 and 1981 from white-band disease (WBD) (Bak and Criens, 1981; Van Duyl, 1985), but other stony coral diseases were only minor sources of mortality (Bak and Nieuwland, 1995). Bleaching events occurred in 1987 (Williams and Bunkley-Williams, 1990), 1990 (Meesters and Bak, 1993), 1995 (CARICOMP, 1997) and 1998 (A. Bruckner, unpublished data). Yellow-blotch disease (YBD) was first noticed in late 1995 as colonies of the *M. annularis* species complex began to recover from the mass bleaching event. It is not known whether colonies with YBD had bleached during this event (P. Hoetjes, pers. comm.).

While Curaçao's reefs clearly have degraded near its population center, its eastern and western coasts are relatively unaffected by pollution or sedimentation and their reefs are thought to be in better condition (Van Veghel, 1997). Subsistence fishers occur throughout the island, and although spearfishing and coral collection were prohibited in 1976 regulations were not enforced until 1998. The Curaçao Underwater Park, established in 1983 and extending 21 km from the outskirts of Willemstad to the eastern tip of the island, includes a 12 km stretch of coastline that is undeveloped and uninhabited. In total, the park encompasses 600 hectares of coral reef habitat and 436 hectares of inner bays. Hook-and-line fishing is allowed within the park, but spearfishing and anchoring are prohibited. All dive sites have mooring buoys. A Caribbean Coastal Marine Productivity CARICOMP) coral reef site has been monitored here at Spaanse Water since 1994. Lagun and Westpunt are two small communities at the western end of Curaçao where there is

no industry and very limited coastal development. A second underwater park has been proposed for these reefs (Banda Abao reef complex).

The purpose of this study was to utilize the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocol to characterize the reefs off the eastern and western ends of Curaçao, in particular to determine whether diseases are an important factor contributing to coral mortality in the absence of significant human activities. Our data suggest that coral diseases have become more prevalent on Curaçao's reefs during the late 1990s and that one disease in particular is causing significant partial mortality to the most abundant and most important of its reef-building corals. Another unexpected source of mortality was attributed to wave energy associated with Hurricane Lenny in October, 1999. Fish assessments made in June 2000 are reported separately in Bruckner and Bruckner (this volume).

## METHODS

Baseline data were initially obtained in June 1997 by tallying the total number of healthy and diseased colonies of each species of stony coral (scleractinian corals and hydrozoan fire corals) observed within belt transects (2 m wide x 30 m long; 1-4 transects/site) along depth gradients (at 10, 15, 20 m) for nine leeward reefs (Fig. 1). In August 1998 and January 2000, detailed reexaminations were conducted at strategically chosen sites on the less populated eastern and western coasts using the AGGRA Version 2 benthos protocol (Appendix One, this volume), with the following modifications. The minimum diameter of assessed corals was 20 cm and size measurements were recorded to the nearest 5 cm. Smaller colonies of reef-building corals (5-20 cm) were tallied and recorded to species. Scleractinian recruits, defined as <2.5 cm diameter, were recorded to genus only (except for *Montastraea cavernosa*), omitting taxa that do not obtain a large size (e.g., *Scolymia* spp., *Favia fragum*). Colonies of the *Montastraea annularis* species complex were separated according to Weil and Knowlton (1994) as *M. annularis*, *M. faveolata* or *M. franksi*. Forms or morphotypes of *Agaricia agaricites*, *Colpophyllia natans*, *Meandrina meandrites* and *Porites porites* were combined under the respective species. Encrusting forms of *Madracis* were recorded as *M. decactis*.

Recent mortality was defined in this study as any tissue loss occurring within approximately the last 60 days, and signs included: (1) white coral skeleton that lacked algae (surfaces denuded of tissue within the last five-seven days); (2) skeletal areas with readily recognizable corallites that had not been substantially eroded but were colonized by green filamentous algae; or (3) white, exposed skeletal surfaces, or eroded skeletal surfaces with fine filamentous algae that had been physically abraded by fish or other agents but had not yet been colonized by macroalgae or coralline algae. Causes of recent mortality were identified as disease [separated into WBD, YBD, black-band disease (BBD), white plague (WP), dark-spots disease (DSD) or other diseases], corallivory [fish bites, damselfish (*Stegastes planifrons*) algal lawns, or snail predation], overgrowth by algae or an invertebrate (cnidarian, sponge or tunicate), or were recorded as unknown. In January 2000, when toppled corals were observed throughout the reefs, especially at 8-12

m, colonies that had become stabilized were measured and described as above, but were tallied as overturned. The long-dead portions that had been recently exposed through toppling were recorded as old mortality; only skeletal areas that met the criteria described above as recent mortality were tallied as such. Dislodged or overturned corals that had not become stabilized were not included in this survey. To standardize observations, all measurements of mortality were performed by the first author and algal quadrats were completed by the second author. Both authors collected information on colony size. All corals along two pilot transects (Lagun and Jeremi, 10 m length) were measured by both authors and measurements were discussed and compared to ensure consistency prior to the actual surveys.

All statistical analyses were performed with the Systat (version 9.0) program. Comparisons among species, locations, and depths were made using a student's *t*-test (for examination of the *M. annularis* species complex versus all other species pooled) or ANOVA (single-factor or two-factor) and correlations were examined with a Pearson product-moment test. A one- or two-factor ANOVA was also used to examine for differences in coral composition, size frequency distribution and percent partial mortality, surveys from different years on western reefs or between eastern and western reefs. When relevant, post-hoc analyses were performed using a Tukey HSD multiple comparison test. For these tests, coral species were lumped into the following six groups based on colony abundance, mean colony size, susceptibility to disease or predation, colony morphology or sexual reproductive character: (1) *C. natans*; (2) the *M. annularis* species complex; (3) other broadcast spawners with massive morphologies (*Diploria* spp., *M. cavernosa*, *Siderastrea siderea*, *Stephanocoenia intersepta*); (4) small branching corals (*P. porites* species complex, *Eusmilia fastigiata* and *Madracis* spp.); (5) *Agaricia* spp.; and (6) other species (*Porites astreoides*, *M. meandrites*, *Mycetophyllia* spp., *Mussa angulosa*). Data were checked graphically to assure that all assumptions of ANOVA were met; log-transformation for length measurements and arc-sine transformation for percentages were used as appropriate prior to analyses.

## RESULTS

### Stony Corals

AGRRA surveys were performed at 9-13 m on the reef terrace at one eastern (Oostpunt) and three western (Kalki, Jeremi, Lagun) sites during August 1998 (Table 1). Further surveys were made at four western sites (Kalki, Westpunt, Jeremi, Lagun) two months after Hurricane Lenny in January 2000, at ~10 m on the reef terrace and to a maximum depth of ~20 m on the reef slope.

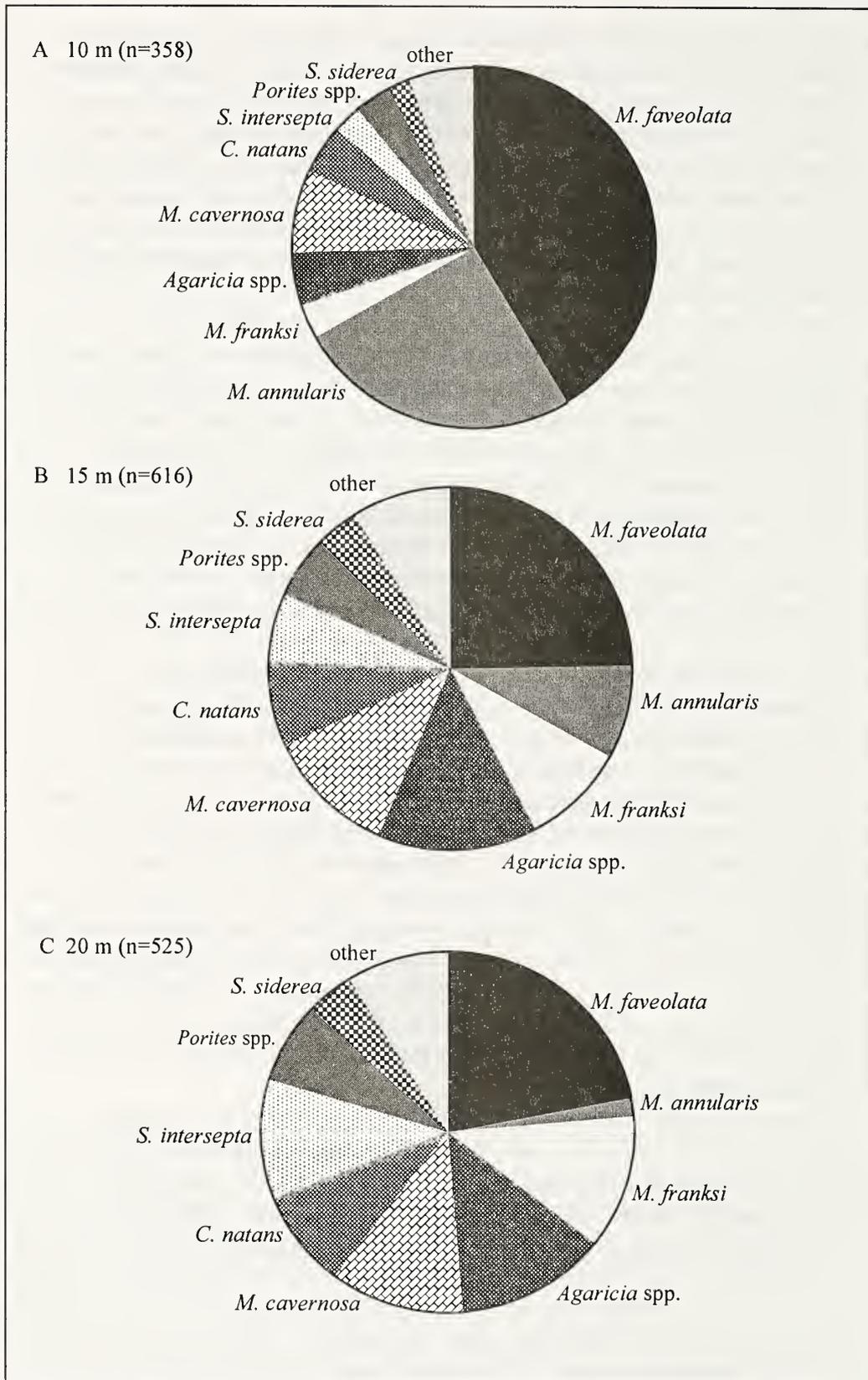
Reefs at 10-20 m were characterized by 18-24 species of scleractinian corals (at least 20 cm in diameter) and the hydrozoan *Millepora complanata*, but were dominated by the *Montastraea annularis* species complex, *Agaricia agaricites*, *Montastraea cavernosa*, *Colpophyllia natans* (Fig. 2) and, at 15-20 m, by *A. lamarki* and *Stephanocoenia intersepta*. In 1998, species composition at 10

m in the six pooled coral groups (see Methods) did not differ among locations (ANOVA,  $p=0.09$ ). Numerically the most abundant corals at all sites and depths (46% of total) belonged to the *M. annularis* species complex which collectively are the primary live cover and structural element of Curaçao's fringing reefs. At the western sites 70% of all corals at 10 m depth consisted of the *M. annularis* species complex, with *M. faveolata* > *M. annularis* > *M. franksi*; 35-42% of all corals at 15 and 20 m (respectively) consisted of these species, with *M. faveolata* > *M. franksi* > *M. annularis*. Similarly, 40% of the corals examined at 10 m in Oostpunt were *M. annularis* and *M. faveolata*.

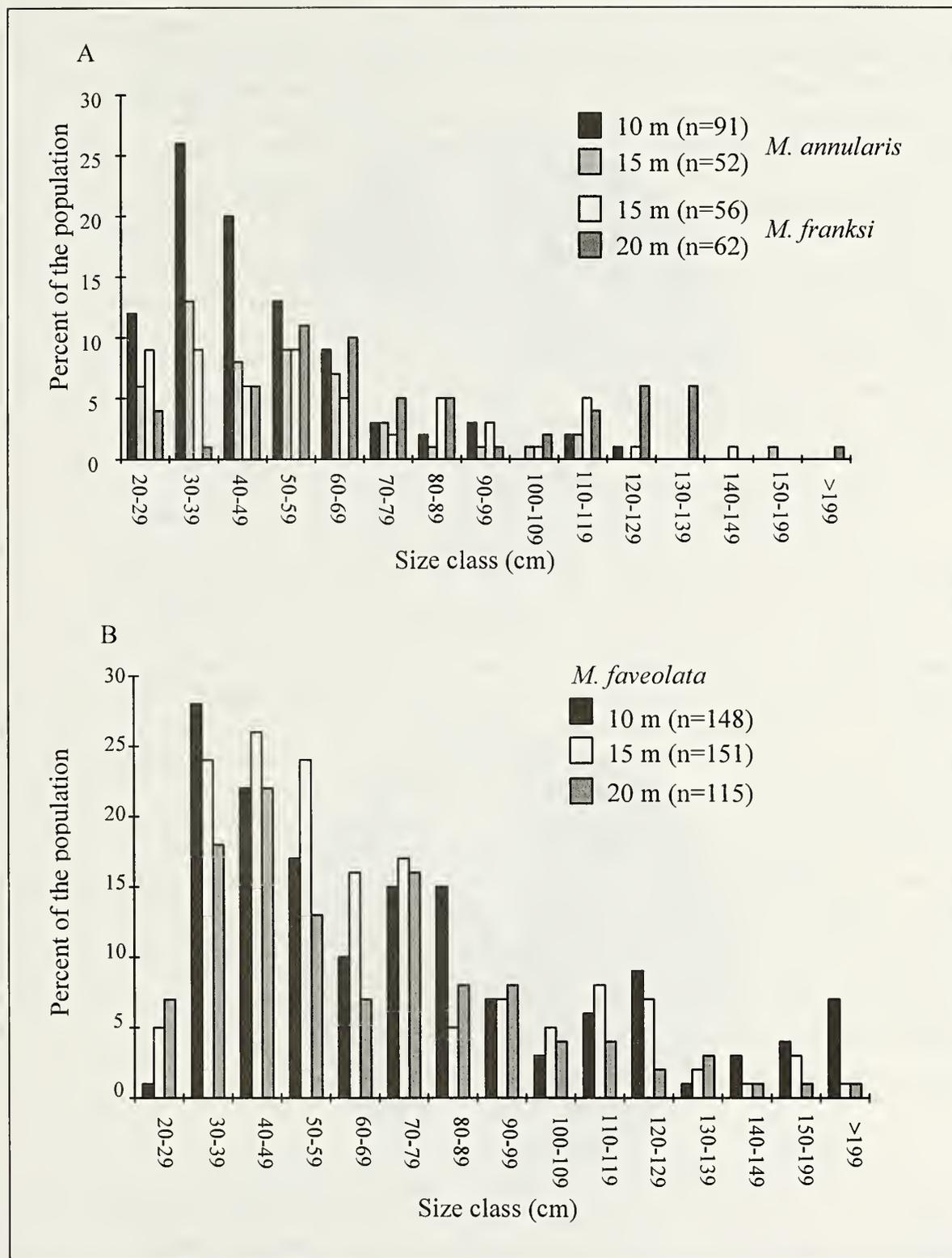
Small, isolated colonies of *Acropora cervicornis* (staghorn coral) and numerous patches of dead staghorn rubble were identified between colonies of the *M. annularis* species complex within transect areas at Oostpunt. On western reefs, the substratum at 7-10 m depth often consisted of dead, consolidated staghorn rubble, but live colonies of *A. cervicornis* were not observed within transect areas nor in the surrounding reefs. Shallow areas (2-4 m) outside of transect areas at Oostpunt had a low abundance of live *A. palmata*. This species was rare or absent in other locations, and the shallows (0-5 m) were nearly devoid of living coral. Large patches of *Madracis mirabilis*, a number of which were several meters in diameter, occurred in 10-15 m on the Kalki reef, and less frequently at other locations.

Colony density (for stony corals of  $\geq 20$  cm diameter) generally ranged from 1.3-2.1 corals per meter (Table 1); data from 2000 indicates that colony density increased with depth ( $r^2=0.61$ ;  $p=0.0016$ ). Coral cover varied among locations, depths and years. In 1998, coral cover ranged from about 20-40% along transects at 10 m depth and was greatest off Kalki. Coral cover at the same depth was substantially lower on the western reefs in January 2000, except for Jeremi, where high variation among the transects, and the presence of several large colonies of *M. faveolata*, may have skewed its mean value (Table 1). The greatest decline in live cover was observed at Kalki where most corals had been removed at 2-12 m from the reef terrace by wave surge. Overall, coral coverage was greater at 15-20 m on reef slopes (Table 1) with the highest percentage occurring at 20 m on Jeremi Reef (49%). Live coral cover was lowest off Westpunt even though this site was minimally impacted by Lenny. Westpunt Reef terminates in sand at 15 m and living corals are absent below this depth.

Transects performed on western reefs in 1998 and 2000 were similar in composition (two-factor ANOVA;  $p=0.59$ ) and diameter of corals ( $p=0.54$ ), with no interaction between years and species ( $p=0.79$ ) with respect to coral size. Colony size recorded during 1998 in the six pooled coral groups did not differ among locations (ANOVA,  $p=0.91$ ). Coral diameter did, however, differ among



**Figure 2.** Species composition and mean relative abundance of all stony corals (>20 cm diameter) in January 2000 at (A) 10 m, (B) 15 m, (C) 20 m, in western Curaçao.



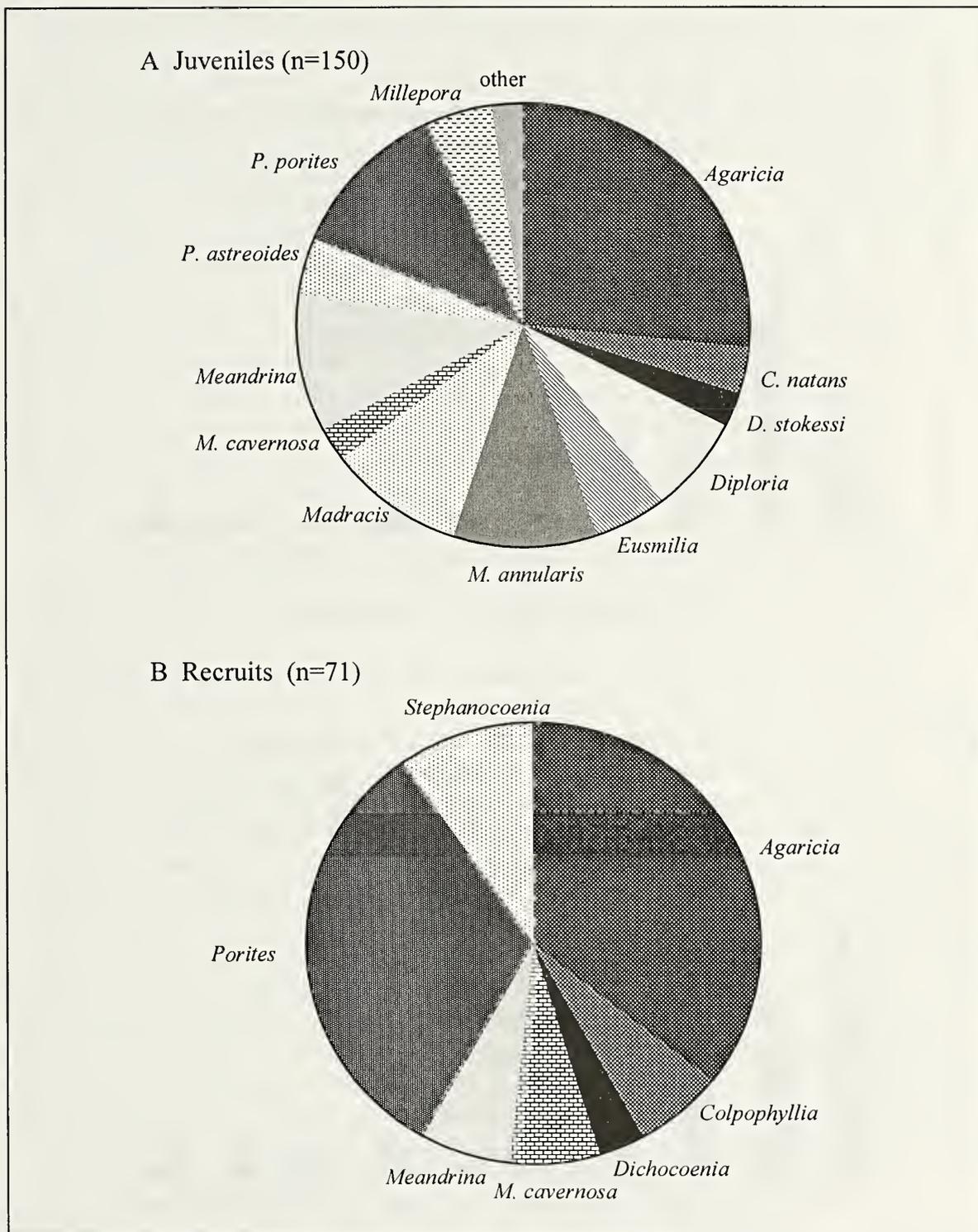
**Figure 3.** Size-frequency distribution of colonies ( $\geq 20$  cm diameter) in January 2000 of (A) *Montastraea annularis* (10 m, 15 m) and *M. franksi* (at 15 m, 20 m), (B) *M. faveolata* (at 10 m, 15 m, 20 m) in western Curaçao.

the species groups (two-factor ANOVA;  $P < 0.001$ ) and an interaction between diameter and location was observed ( $P = 0.009$ ). The largest colonies observed within transect areas were *M. faveolata* and *C. natans*, while most of the brooding species (e.g., *Agaricia* spp. and *Porites* spp.) showed a predominance of the smaller size classes. In January 2000, a large number of the smaller colonies, especially *M. annularis*, were overturned or displaced. The effects of the storm were highly localized, however, as deeper areas and larger corals were minimally impacted. Most corals examined in January 2000 were intermediate in size (30-80 cm diameter, mean=53 cm,  $n=1501$ ; Fig. 3). Colonies of the *M. annularis* species complex were significantly larger in diameter (mean=63 cm) than all other species combined (mean=45 cm) (all depths pooled; t-test,  $t=12.5$ ,  $df=1497$ ,  $p < 0.001$ ). In addition, colonies of *M. faveolata* and *M. franksi* (mean=67 cm) were larger than *M. annularis* (mean=46 cm; ANOVA,  $MS=1.13$ ,  $F=24.7$ ,  $p < 0.001$ ), but no differences were noted among *M. faveolata* and *M. franksi*. Colony size did not differ among depths for the *M. annularis* species complex (ANOVA,  $MS=0.11$ ,  $F=2.35$ ,  $p=0.095$ ) or other species (ANOVA,  $MS=0.03$ ,  $F=1.01$ ,  $p=0.36$ ).

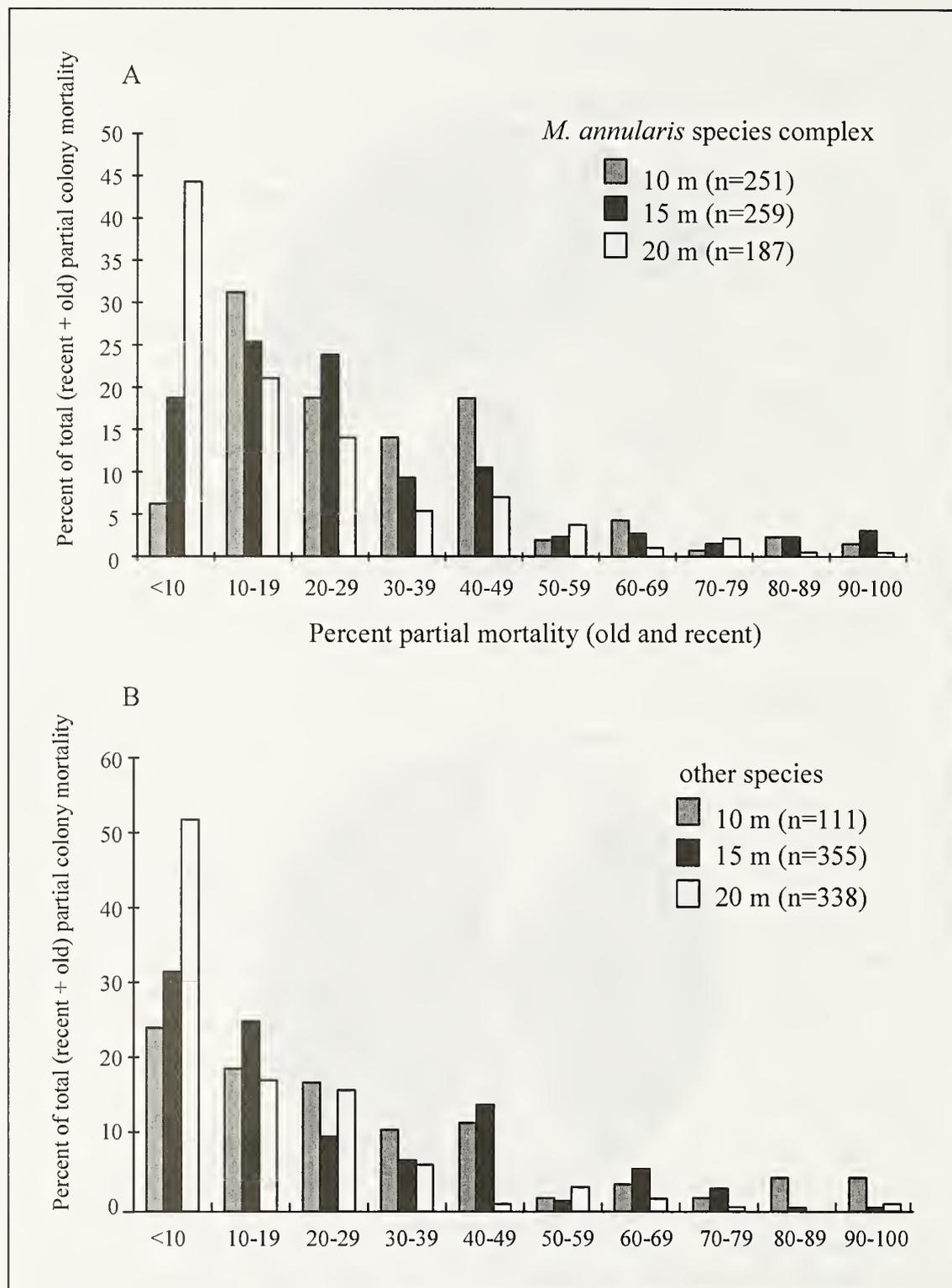
Small corals (5-20 cm diameter) recorded in 1998 along transects (mean abundance=0.5/meter, range=3-9/10 m) consisted predominantly of *A. agaricites* (1.3/10 m), *P. porites* (0.6/10 m), *M. annularis* (0.5/10 m), *M. meandrites* (0.5/10 m), *M. mirabilis* (0.5/10 m) and 16 other species (Fig. 4A). A low abundance of recruits (<2.5 cm diameter) was identified within quadrats at 10 m depth (0.5 - 0.7 recruits/0.0625 m<sup>2</sup>) in 1998 (Table 3), most individuals of which were brooders including *Agaricia* and *Porites*. However, broadcasters like *M. cavernosa*, *Dichocoenia*, *Colpophyllia*, *Stephanocoenia* and *Meandrina* were also observed (Fig. 4B). An absence of recruits of the *M. annularis* species complex was noted, even though these were the dominant corals on all reefs. Recruits were also recorded in 2000 but lower numbers were seen (mean=0.12 recruits/0.0625m<sup>2</sup> at 10 m depth). In shallow transects (9-12 m), recruits were not observed on reef substrata with a high cover of macroalgae (Jeremi and Lagun) or on substrates that had been exposed relatively recently (e.g., at Kalki). Recruits were observed on long-dead coral skeletons and reef substrata not directly affected by the hurricane with a higher number at 15 m on reef substrata (up to five recruits/0.0625m<sup>2</sup> mean=0.61) than on denuded coral (especially *M. faveolata*, *M. annularis*, *M. franksi*) skeletons (mean=0.13) (t test,  $t=3.5$ ;  $df=108$ ;  $p < 0.001$ ).

### Coral Condition

In this study we examined the condition of 1,939 scleractinian and hydrozoan reef-building corals (1998 and 2000, all sites and depths pooled). Overall, in 32% of all corals total (recent + old) partial-colony mortality (hereafter total partial mortality) affected less than 10% of their planar surface area. Mean values of total partial mortality at each site (all corals pooled, 1998 and 2000) ranged from 15-32% with less than 20% of all corals missing more than half their tissues (Fig. 5). Distinct differences in percent total partial mortality were also noted among the six species groups (all years and depths are pooled,  $p < 0.001$ ).



**Figure 4.** Species composition and mean relative abundance in August 1998 of (A) all “juvenile” (5-20 cm diameter) stony corals and (B) all recruits (<2.5 cm diameter, excluding species that are small as adults) at 10 m, 15 m and 20 m in western (four sites) and eastern (one site) Curaçao.



**Figure 5.** Frequency distribution in January 2000 of total (recent + old) partial colony mortality of all colonies ( $\geq 20$  cm diameter) of (A) the *Montastraea annularis* species complex, (B) other species, in western Curaçao.

From 20-40% of each small clump of *A. cervicornis* consisted of dead branches. Colonies of the widely distributed *M. annularis* species complex overall exhibited a significantly greater amount of total partial mortality than all other species (pooled) except *A. cervicornis* (all sites, depths and years are pooled, mean partial mortality=24%; t-test,  $t=6.2$ ,  $df=1497$ ,  $p<0.0001$ ). No relationship was observed between percent tissue loss and colony diameter for the *M. annularis* species complex ( $r^2=0.002$ ;  $p=0.28$ ) or other pooled species ( $r^2=0.04$ ), possibly due to the high amount of variation observed within each size class.

In 1998, the amount of total partial mortality in the six pooled species groups was found to vary among species (two-factor ANOVA;  $P<0.001$ ), with minor differences among locations ( $p=0.066$ ) and a significant species-location interaction with respect to total partial mortality ( $p<0.001$ ). Total partial mortality was greatest among the *M. annularis* species complex and branching corals (*Porites porites*, *Eusmilia fastigiata*, *Madracis* spp.) while the group of brooding species had the lowest percentage (Tukey test). Post-hoc analysis indicates that total partial mortality in 1998 was slightly higher on western reefs (21% versus 19% at Oostpunt), yet the greatest amount overall (35%) was recorded for the *M. annularis* species complex at Oostpunt.

In 2000, a large number of small-to-intermediate-sized corals were dislodged or overturned in shallow water (7-13 m depth), and numerous unattached colonies had been transported down the reef slope to 15-20 m or deeper. Overturned colonies identified along transects that were 20 cm or larger had a mean size of 49.5 cm (maximum=160 cm). Overturned colonies were substantially larger on Jeremi (mean=60.5 cm) than on Lagun (mean=39.5 cm), but the total proportion of overturned colonies versus those that were unaffected was higher at Lagun (26%) than on Jeremi (20%). A small number (<3% of all corals examined) experienced total partial mortality; most of the corals that survived had sustained a low amount of recent partial-colony mortality (hereafter recent mortality) with the exception of areas on the colony now in contact with the substrata. The amount of total partial mortality in 2000 varied significantly among depths for the *M. annularis* species complex (ANOVA,  $MS=5628$ ,  $F=24.8$ ,  $p<0.001$ ) and other species (ANOVA,  $MS=4103$ ,  $F=13.2$ ,  $p<0.001$ ) with the greatest loss at 10 m depth and the least at 20 m (Table 2). Whereas the amount of recent mortality at 10 m on western reefs was greater in 1998 than 2000 (3.7% versus 0.6% of colony surfaces, respectively), the amount of total partial mortality at 10 m was significantly greater in 2000 (mean=29%) than in 1998 (mean=21%; two-factor ANOVA,  $p=0.003$ ). Minor differences in percent total partial mortality were noted among the six species groups ( $p=0.06$ ) but there was no interaction between the survey period and species ( $p=0.175$ ). Overall, *S. intersepta*, the *M. annularis* species complex, *S. siderea* and *E. fastigiata* respectively exhibited the greatest amount of total partial mortality (22-31%), while the lowest values were observed in *M. meandrites*, *D. strigosa*, *D. stokesii*, *P. astreoides* and *P. porites* (10-13%). The other dominant species (*C. natans*, *M. cavernosa*, and *Agaricia* spp.) exhibited 17-19% total partial mortality.

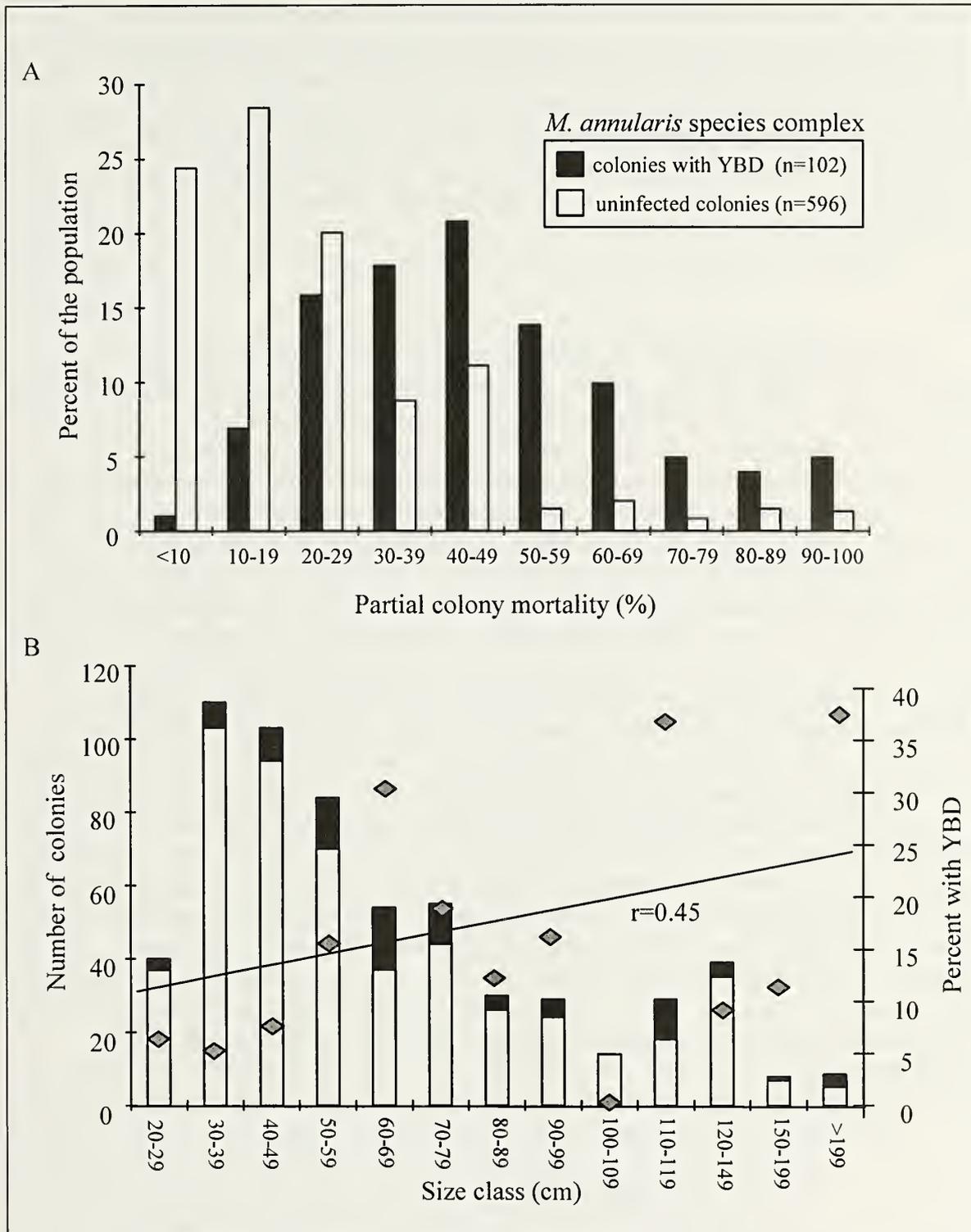
The most common sources of recent mortality were coral diseases, in particular YBD, DSD, WP, WBD and red-band disease (Table 4). On all reefs, a high percentage of the *M. annularis* species complex exhibited signs of YBD; this condition was not recorded in other taxa. WBD was only observed at the Oostpunt,

susceptible acroporid corals being absent at the western sites. A high prevalence of DSD was noted in 2000. It is not known whether DSD was common in 1998 because the affected species (*S. intersepta*, *S. siderea*) were predominantly recorded at 15 and 20 m, at depths which had not been surveyed in 1998.

Coral diseases appeared to be most common in 1997 (Bruckner and Bruckner, 1999; unpublished data), and have since declined. The highest prevalence of YBD was observed along the east coast, near Willemstad (30.6% of the 264 colonies of the *M. annularis* species complex examined at Piscadera; 49%, N=165 at Jan Thiel) and at Oostpunt (37.5%, N=474), with fewer infections seen off the western coast (24%, N=607). In 1998 fewer infections were observed in western sites at 10 m depth (15.6-19.2% of the *M. annularis* species complex) while Oostpunt had a higher prevalence of YBD (68% of all colonies of the *M. annularis* species complex). Although the occurrence of new YBD infections was lower in January 2000 on western reefs, many older infections still plagued colonies (total=14.5% of 698 of the *M. annularis* species complex) as indicated by the large amount of partial mortality immediately adjacent to the YBD-affected tissues. Moreover, colonies of the *M. annularis* species complex with YBD had lost a significantly greater percentage of their tissues than had unaffected conspecifics (mean loss=44% versus 20%, respectively; t-test,  $t=10.6$ ,  $df=127$ ,  $p<0.0001$ ; Fig. 6A) and were larger in size (mean diameter=72 cm) than uninfected corals (61 cm). Diameter was correlated to incidence of YBD ( $r^2=0.45$ ,  $p=0.0001$ ): 25% of all colonies greater than 0.5 m diameter were infected, versus only 7% of the smaller colonies (Fig. 6B). BBD and WP were also more prevalent during the summer of 1998 (3.6%) than in January 2000 (0.1%); however, these differences may relate to seasonal variations associated with water temperature rather than an overall decline in disease.

An additional source of coral mortality was attributed to predation. Stoplight parrotfish (*Sparisoma viride*) bites were observed on all reefs. Focused biting was primarily observed among *M. annularis* and *C. natans* and spot-biting affected these and 12 other species. Overall, the most extensive skeletal and tissue destruction from *S. viride* occurred on *M. annularis* (9.8% of all colonies) at 10 m in Oostpunt in 1998. Lesions from fish bites were very common in 1997 and 1998 (4.4% of all colonies of *M. annularis*), and affected 2.2% of all colonies examined at 10 m depth in 1998. Although fewer colonies appeared to be affected by fish predation on western reefs in 2000, affected colonies were highly aggregated and areas exhibiting focused biting in 1998 also had affected colonies in 2000. Predation by the snail *Coralliophila abbreviata* (in 5/14 surveys) and overgrowth by sponges and the tunicate *Trididemnum solidum* (4/14 surveys) were also noted (Table 4).

Bleaching was not recorded during the August 1998 and January 2000 surveys. However, bleached colonies were observed in November 1998, primarily among *Agaricia* spp. and the *M. annularis* species complex (personal observations). During this bleaching event about 30% of the *M. annularis* species complex became pale, especially on their upper surfaces, but did not turn



**Figure 6.** Relationship between the presence (or absence) in January 2000 of YBD and (A) percent of total (recent + old) partial colony mortality, (B) size frequency distribution, for the *M. annularis* species complex ( $\geq 20$  cm diameter) in western Curaçao. Data are pooled from 10 m, 15 m and 20 m. Grey diamonds in (B) indicate the percent of each size class affected by YBD; a best fit line and r-value are presented.

completely white. Approximately 80% of all colonies of *A. lamarki* in deeper water (18-25 m depth) were completely bleached (white). Bleaching affected other species but was less prevalent. We tagged and photographed 30 of these corals (*A. lamarki* and the *M. annularis* species complex) in November 1999 and reexamined them in January 2000. Over 75% (n=26) of these regained full pigmentation and did not experience any mortality. In the other four colonies, tissue mortality affecting 5-30% of the surface of each plate was noted but the remaining live tissues had regained most pigmentation.

## Algae

Algal communities at 10 m in 1998 were dominated by crustose coralline algae (50 %) and sparse turf algae (38 %) that did not trap considerable amounts of sediment. Macroalgae were codominant on exposed coral skeletons and at the bases of coral heads (Table 3). Where present, macroalgae consisted primarily of *Dictyota* spp., *Halimeda* spp. and *Lobophora variegata*, and were usually less than 1.5 cm in height. The algal community was very different in January 2000, possibly due to the effects of Hurricane Lenny two months previously. At Kalki, the newly exposed reef substrata consisted primarily of cemented *A. cervicornis* skeletons that were being colonized by fine filamentous turf algae; macroalgae and crustose corallines were rare. Mats of cyanobacteria, which had been observed here and at Jeremi in August 1998, were uncommon at both sites. At Lagun, Jeremi and Westpunt, a dense growth of red algae (*Liagora* spp., *Trichogloea* spp., *Trichogloeopsis pedicellata* and other similar fleshy algae; mean height  $5 \pm 21.5$  cm) occupied much of the exposed substrata in January 2000. On Jeremi, these algae occupied 43% of open substrata at 10 m depth but were not growing on living corals. Dead coral surfaces (i.e., areas on colonies denuded of tissue) had similar algal communities in 1998 and 2000 composed predominantly of fine filamentous turfs and crustose corallines with sparse macroalgae. In most cases, patches of macroalgae (*Dictyota* spp, *Lobophora* spp. and *Halimeda*) occurred at the base of coral heads, between lobes of living coral, and in crevices, while up to 80% of the exposed surfaces were colonized by turfs and crustose coralline algae. *Diadema antillarum* was not observed along transects, although signs of grazing were apparent on exposed coral surfaces and herbivorous fish (mean densities in 1998: surgeonfish, 5/100 m<sup>2</sup> and parrotfish, 4/100 m<sup>2</sup>) were recorded in belt transects (30 m long x 2 m wide) (Bruckner and Bruckner, this volume).

## DISCUSSION

Coral reefs examined off Curaçao in 1998 and 2000 were dominated by intermediate to large-sized massive scleractinian taxa, including *M. faveolata*, *M. annularis*, *M. franksi*, *S. intersepta*, *S. siderea* and *C. natans*, and smaller colonies of *P. astreoides* and *A. agaricites*. *Acropora palmata* and *A. cervicornis* were prevalent in the 1970s, declined in the early 1980s as a result of a regional disease epizootic (Van Duyl, 1985), and were uncommon (Oostpunt, Jan Thiel, Piscadera)

or were not observed (western reefs) during these surveys. Living coral cover ranged from 17-49% with the exception of one shallow site on the western coast that lost >95% of its live stony corals as a result of a hurricane in 1999. Most colonies over 20 cm (69%) had experienced total partial mortality that affected at least 10% of their planar surface area, with a mean tissue loss in all colonies of 22%. The major sources of mortality noted in this study were coral diseases (prevalent in all sites and years) and hurricane damage (affecting western reefs in 2000). *Diadema antillarum* was rare or absent in all sites but a low abundance of macroalgae and turf algae suggests that herbivorous fish are effectively controlling algal populations.

In general, the eastern reefs were in poorer condition than the western reefs. A precipitous decline in coral abundance, cover, and species diversity had been reported on reefs near Willemsted in previous studies (e.g., Bak and Nieuwland, 1995). In 1997, these areas had the highest prevalence of disease with up to 49% of the  $\geq 20$  cm diameter colonies affected in one location. Oostpunt, a protected area located off the uninhabited eastern end of Curaçao, was reported to be in good shape as recently as 1995 (Van Veghl, 1997). This site experienced a mass bleaching event in the fall of 1995 and present surveys revealed a high incidence of disease in 1997 (38%) and 1998 (21%) and other biotic disturbances (especially focused biting by *S. viride*). Colonies at this site had also sustained a higher percentage of total partial mortality, and there were more entirely dead colonies, than observed at the same depth on western reefs.

The amount of total partial mortality differed among species, and was highest in the slowly reproducing, large, massive broadcasters (*M. annularis* species complex, *S. siderea*, *S. intersepta*, *C. natans*) that dominated eastern and western reefs. In particular, colonies of the *M. annularis* species complex, which were the most abundant and largest corals at all sites and depths, had experienced a high degree of total partial mortality. They were also affected most severely by disease: between 7-49% of all colonies were observed with signs of disease in surveys conducted in 1997, 1998 and 2000. The main disease affecting the *M. annularis* species complex was YBD, a condition that causes relatively slow rates of mortality (1-2 cm spread per month), but may affect individual colonies for several years (Bruckner and Bruckner, 2000).

The prevalence of large colonies of the *M. annularis* species complex that are hundreds of year old and few colonies less than 30 cm in diameter suggests that recruitment events of significance have not occurred among these species in several decades. These large, ecologically-dominant colonies have exhibited high rates of survivorship, and are presumably well adapted to deal with chronic disturbances such as predation, bioerosion and disease (Bythell et al., 1993). However, this may no longer be the case for these reefs as recent disease epizootics have primarily plagued these species. Overall, larger colonies of the *M. annularis* species complex were infected with YBD more frequently than small colonies. In addition, colonies with active signs of YBD have lost a mean of 44% of their tissue area in January 2000 or roughly twice that of uninfected colonies of the same species. Chronic YBD infections on these reefs may have serious ramifications for the persistence of the *M. annularis* species complex, as continued

tissue loss and fission may significantly reduce the reproductive potential of these colonies and the proportion of small, non-breeding colonies in the population may increase. Furthermore, several decades or more may be required for their replacement because of their slow rate of growth, delayed reproduction, and infrequent, episodic recruitment (Szmant, 1991).

Although coral reefs surveyed in this study have a high prevalence of coral disease and have been impacted by a recent hurricane, Curacao's reefs appear to be relatively resilient to recent disturbances and have a high potential for continued growth and sexual recruitment. Crustose coralline algae are prevalent on exposed reef substrata; macroalgae and turf algae have remained sparse with the exception of a benthic algal bloom in shallow water that lasted for three months after Hurricane Lenny and has since disappeared. Most locations have relatively high cover of live coral (over 25%), a high abundance of small corals less than 20 cm (3-9/10 m) and new recruits are present (0.5-0.7 recruits/.0625 m<sup>2</sup>). Longer-term effects associated with the hurricane appear to be minimal. Most colonies overturned during Lenny have become stabilized; fragmented and overturned corals contained substantial amounts of surviving tissue which were not diseased, bleached or injured. In addition, surveys from January 2000 indicate that the number of active and new YBD infections have declined and other diseases were less abundant.

Reefs at the western end of the island have experienced coral mortality as a result of disease and hurricane damage but ensuing mortality appears to be declining, coral cover remains high and remaining corals are in relatively good condition. In light of an ongoing accelerated decline of reefs located near human population centers, including those off populated coastlines of southeastern Curaçao (Bak and Nieuwland, 1995), conservation efforts should be directed towards the island's more remote western end. The western reefs may serve as a refuge for important reef-building species and a source of larvae for these and other reefs down current. Recent prohibitions on spearfishing, along with the proposed establishment of a marine park for western Curaçao, are two steps forward that may help ensure the long-term persistence of its reef ecosystems.

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Table 1. Site information for AGRRA stony coral and algal surveys off Curaçao, Netherlands Antilles. 1998 sites are italicized.

Site name	Reef type	Latitude (N° ' ' '')	Longitude (W° ' ' '')	Survey date	Depth (m)	Benthic transects (#)	>20 cm stony corals (#/10 m)	% live stony coral cover (mean ± se)
<i>Eastern Oospunt</i>	<i>Fringing</i>	<i>12 01</i>	<i>68 44 59.8</i>	<i>Aug 15 98</i>	<i>10</i>	<i>7</i>	<i>15</i>	<i>27 ± 2.9</i>
<i>Western Lagon</i>	Fringing	12 19 6.9	69 09 54.0	Aug 13 98	10	9	13	25 ± 3.0
				Jan 13 00	10	9	14	17 ± 1.5
				Jan 12 00	15	9	16	38 ± 4.5
				Jan 10 00	20	9	18	35 ± 1.2
Jeremi	Fringing	12 19 44.3	69 09 1.2	Aug 14 98	10	7	16	31 ± 3.9
				Jan 10 00	10	9	17	29 ± 1.5
				Jan 14 00	15	9	19	36 ± 3.7
				Jan 12 00	20	9	21	49 ± 1.9
Westpunt	Fringing	12 21 37.2	69 09 42.1	Jan 14 00	10	6	13	21 ± 1.6
				Jan 14 00	15	9	16	25 ± 1.4
Kalki	Fringing	12 22 31.3	69 09 29.8	Aug 14 98	10	7	15	40 ± 4.1
				Jan 13 00	10	6	0.8	1.6 ± 0.9
				Jan 13 00	15	9	17	37 ± 1.8
				Jan 11 00	20	9	18	38 ± 2.1

Table 2. Size and condition (mean  $\pm$  standard error) of all stony corals ( $\geq 20$  cm diameter), by site off Curaçao.

Site name	Year	Depth (m)	Stony corals		Partial-colony surface mortality (%)			Stony corals (%)		
			(#)	Diameter (cm)	Recent	Old	Total	Standing dead	Diseased with Damselfish	
<b>Eastern</b>										
Oostpunt	1998	10	110	56 $\pm$ 3.8	3.5 $\pm$ 0.6	16 $\pm$ 2.3	19 $\pm$ 2.5	7	21.5	3
<b>Western</b>										
Lagun	1998	10	118	51 $\pm$ 2.0	2.0 $\pm$ 0.4	17 $\pm$ 2.0	19 $\pm$ 2.3	2	10	1
	2000	10	131	49 $\pm$ 2.2	0.5 $\pm$ 0.2	27 $\pm$ 1.9	28 $\pm$ 1.9	0	15	1
	2000	15	145	57 $\pm$ 2.6	0.5 $\pm$ 0.2	23 $\pm$ 2.0	24 $\pm$ 2.0	2	7	0.5
	2000	20	165	52 $\pm$ 1.7	0.5 $\pm$ 0.2	16 $\pm$ 1.4	17 $\pm$ 1.4	2	8	2.5
Jeremi	1998	10	110	53 $\pm$ 2.7	3.0 $\pm$ 0.8	22 $\pm$ 2.1	25 $\pm$ 2.3	2	12	0
	2000	10	152	64 $\pm$ 3.7	0.5 $\pm$ 0.1	28 $\pm$ 1.7	29 $\pm$ 1.7	0	9.5	3.5
	2000	15	171	47 $\pm$ 2.1	0.5 $\pm$ 0.2	21 $\pm$ 1.3	22 $\pm$ 1.4	0	8	0.5
	2000	20	191	51 $\pm$ 2.1	0.5 $\pm$ 0.1	16 $\pm$ 1.4	17 $\pm$ 1.4	1	8	4.5
Westpunt	2000	10	79	60 $\pm$ 4.3	0.5 $\pm$ 0.2	29 $\pm$ 2.5	30 $\pm$ 2.5	0	10.5	0.5
	2000	15	142	53 $\pm$ 2.2	1.0 $\pm$ 0.3	25 $\pm$ 1.9	26 $\pm$ 1.9	0	8	12
Kalki	1998	10	107	54 $\pm$ 3.3	3.0 $\pm$ 0.5	16 $\pm$ 2.0	19 $\pm$ 2.2	0	16	17
	2000	15	156	53 $\pm$ 2.3	1.0 $\pm$ 0.2	18 $\pm$ 1.6	19 $\pm$ 1.6	0	10.5	3.5
	2000	20	169	54 $\pm$ 2.6	0.5 $\pm$ 0.9	16 $\pm$ 1.5	17 $\pm$ 1.5	0	11.5	0.5
<b>All sites</b>	1998	10	445	53 $\pm$ 1.8	3.0 $\pm$ 0.5	18 $\pm$ 1.1	21 $\pm$ 1.2	3	14.6	5
<b>Western sites</b>	1998	10	335	52 $\pm$ 1.6	2.7 $\pm$ 0.3	19 $\pm$ 1.2	21 $\pm$ 1.3	1	12.5	6.3
	2000	10	362	57 $\pm$ 2.0	0.6 $\pm$ 0.3	28 $\pm$ 1.1	28 $\pm$ 1.1	0	11.4	2
	2000	10-20	1501	53 $\pm$ 0.8	0.6 $\pm$ 0.1	21 $\pm$ 0.5	22 $\pm$ 0.6	0.3	9.4	3

Table 3. Algal characteristics and stony coral recruit abundance (mean  $\pm$  standard error), by site off Curaçao.

Site name	Year	Depth	Quadrats (#)	Relative abundance (%)			Macroalgal		Recruits (#/0.0625 m <sup>2</sup> )
				Macroalgae	Turf algae	Crustose coralline algae	Height	Index <sup>2</sup>	
<b>Eastern</b>									
<i>Oostpunt</i>	1998	10	30	15 $\pm$ 4.1	38 $\pm$ 6.0	47 $\pm$ 5.9	1.3 $\pm$ 0.1	15	0.50 $\pm$ 0.15
<b>Western</b>									
Lagun	1998	10	35	14 $\pm$ 2.6	35 $\pm$ 4.5	52 $\pm$ 4.1	1.2 $\pm$ 0.1	16	0.74 $\pm$ 0.17
	2000	10	16	23 $\pm$ 5.1	53 $\pm$ 6.8	24 $\pm$ 6.6	1.0 $\pm$ 0.1	24	0.14 $\pm$ 0.08
	2000	15	50	26 $\pm$ 2.6	49 $\pm$ 2.4	25 $\pm$ 1.8	1.5 $\pm$ 0.1	39	0.28 $\pm$ 0.11
Jeremi	1998	10	44	7 $\pm$ 1.3	38 $\pm$ 3.4	55 $\pm$ 3.3	0.7 $\pm$ 0.1	3	0.59 $\pm$ 0.14
	2000	10	24	43 $\pm$ 4.3	42 $\pm$ 4.2	15 $\pm$ 1.6	8.5 $\pm$ 1.2	366	0.44 $\pm$ 0.18
	2000	15	32	17 $\pm$ 2.7	57 $\pm$ 3.3	26 $\pm$ 2.4	1.3 $\pm$ 0.1	17	0.16 $\pm$ 0.08
Kalki	1998	10	30	21 $\pm$ 3.0 <sup>1</sup>	35 $\pm$ 5.5	44 $\pm$ 5.2	1.3 $\pm$ 0.12	14	0.67 $\pm$ 0.15
	2000	10	20	7 $\pm$ 2.2	83 $\pm$ 4.3	10 $\pm$ 2.3	0.5 $\pm$ 0.14	4	0.24 $\pm$ 0.09
	2000	15	34	15 $\pm$ 3.0	55 $\pm$ 3.3	30 $\pm$ 2.9	1.3 $\pm$ 0.14	19	0.33 $\pm$ 0.14
<b>All sites</b>	1998	10	139	12 $\pm$ 1.5	37 $\pm$ 2.4	50 $\pm$ 2.3	1.1 $\pm$ 0.05	11.2	0.58 $\pm$ 0.08
<b>Western sites</b>	2000	10	57	27 $\pm$ 4.8	56 $\pm$ 3.8	17 $\pm$ 2.6	4.9 $\pm$ 0.90	152	0.20 $\pm$ 0.09
	2000	15	116	20 $\pm$ 1.8	57 $\pm$ 1.7	23 $\pm$ 1.4	1.4 $\pm$ 0.07	28	0.26 $\pm$ 0.14

<sup>1</sup>Includes 9% cyanobacteria.<sup>2</sup>Macroalgal index = relative macroalgal abundance x macroalgal height

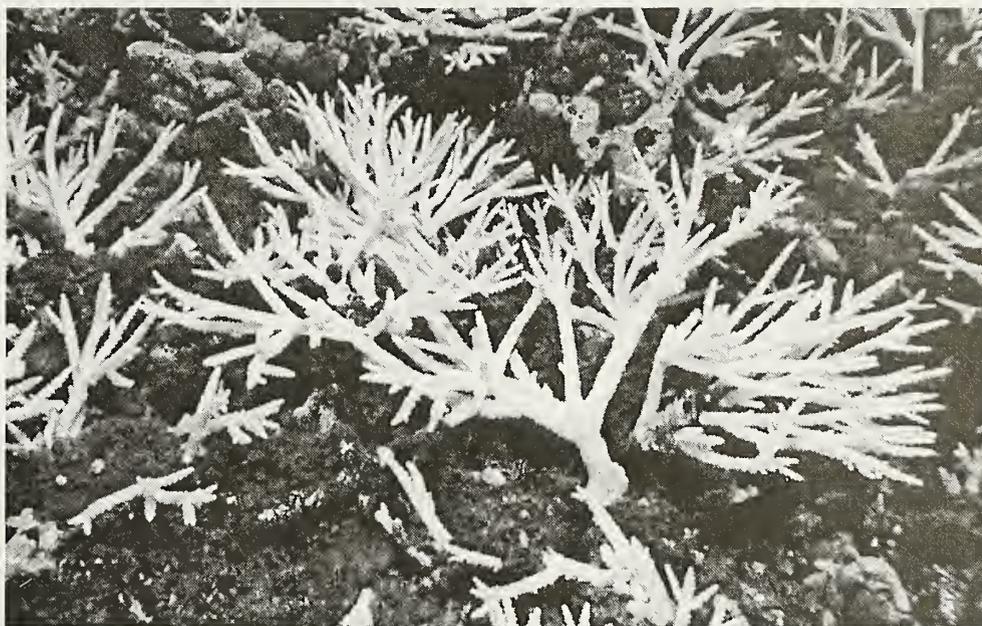
Table 4. Causes of recent mortality for all scleractinians ( $\geq 20$  cm diameter), as numbers and percentages of all colonies, by site off Curaçao.

Site name	Year	Depth (m)	Corals (#)	YBD <sup>1</sup> # (%)	DSD <sup>2</sup> # (%)	WP <sup>3</sup> # (%)	BBD <sup>4</sup> # (%)	PFB <sup>5</sup> # (%)	Snail <sup>6</sup> # (%)	Overgrowth <sup>7</sup> # (%)
<b>Eastern</b>										
Oostpunt	1998	10	110	17 (16.5)	0	5 (5)	0	4 (4)	3 (3)	0
<b>Western</b>										
Lagun	1998	10	118	12 (10)	0	0	0	3 (2.5)	0	5 (4.0)
	2000	10	131	15 (11.5)	2 (1.5)	1 (1)	1 (1)	3 (2.5)	3 (2.5)	2 (1.5)
	2000	15	145	10 (7)	0	0	0	0	0	0
	2000	20	165	8 (5)	5 (3)	0	0	6 (3.5)	0	0
Jeremi	1998	10	110	10 (9)	0	3 (2.5)	0	3 (2.5)	4 (3.5)	3 (2.5)
	2000	10	152	14 (9)	0	0	0	0	0	0
	2000	15	171	10 (5.5)	0	0	0	0	0	0
	2000	20	191	8 (4)	6 (3)	1 (0.5)	0	1 (0.5)	1 (0.5)	0
Westpunt	2000	10	79	11 (14)	0	0	0	6 (7.5)	0	0
	2000	15	142	7 (9)	1 (0.5)	0	0	0	0	0
Kalki	1998	10	107	7 (6.5)	2 (2)	6 (5.5)	2 (2)	0	4 (3.5)	0
	2000	15	156	11 (7)	5 (3)	0	0	1 (0.5)	0	0
	2000	20	169	7 (4)	12 (7)	0	0	1 (0.5)	0	1 (0.5)
<b>All sites</b>	1998	10	445	13.0	0.4	3.2	0.5	2.2	2.5	1.8
<b>Western sites</b>	2000	10	1501	6.7	2.1	0.1	0.07	1.2	0.3	0.2

<sup>1</sup>YBD = yellow-band disease; <sup>2</sup>DSD = dark-spot disease; <sup>3</sup>WP = white-plague disease; <sup>4</sup>BBD = black-band disease; <sup>5</sup>PFB = parrotfish bites;

<sup>6</sup>Snails = *Coralliphila abbreviata* predation; <sup>7</sup>Overgrowth = overgrowth by a sponge or tunicate.

Note that some conditions, such as YBD and DSD, affect a relatively small number of species; infection rates are substantially higher among individual species than at the community level.



**Plate 9A.** Each stony coral, as in these colonies of *Acropora cervicornis*, is assessed for the presence and intensity of any bleaching that is related to mass bleaching events. (Photo Kenneth W. Marks)



**Plate 9B.** Bleaching is characterized according to the approximate severity of tissue discoloration as *pale* (discoloration of coral tissues); *partly bleached* (patches of fully bleached or white tissue); and *bleached* (tissues are totally translucent, and the white skeleton is visible), as shown for this *Porites*. (Photo Larry Benvenuti)

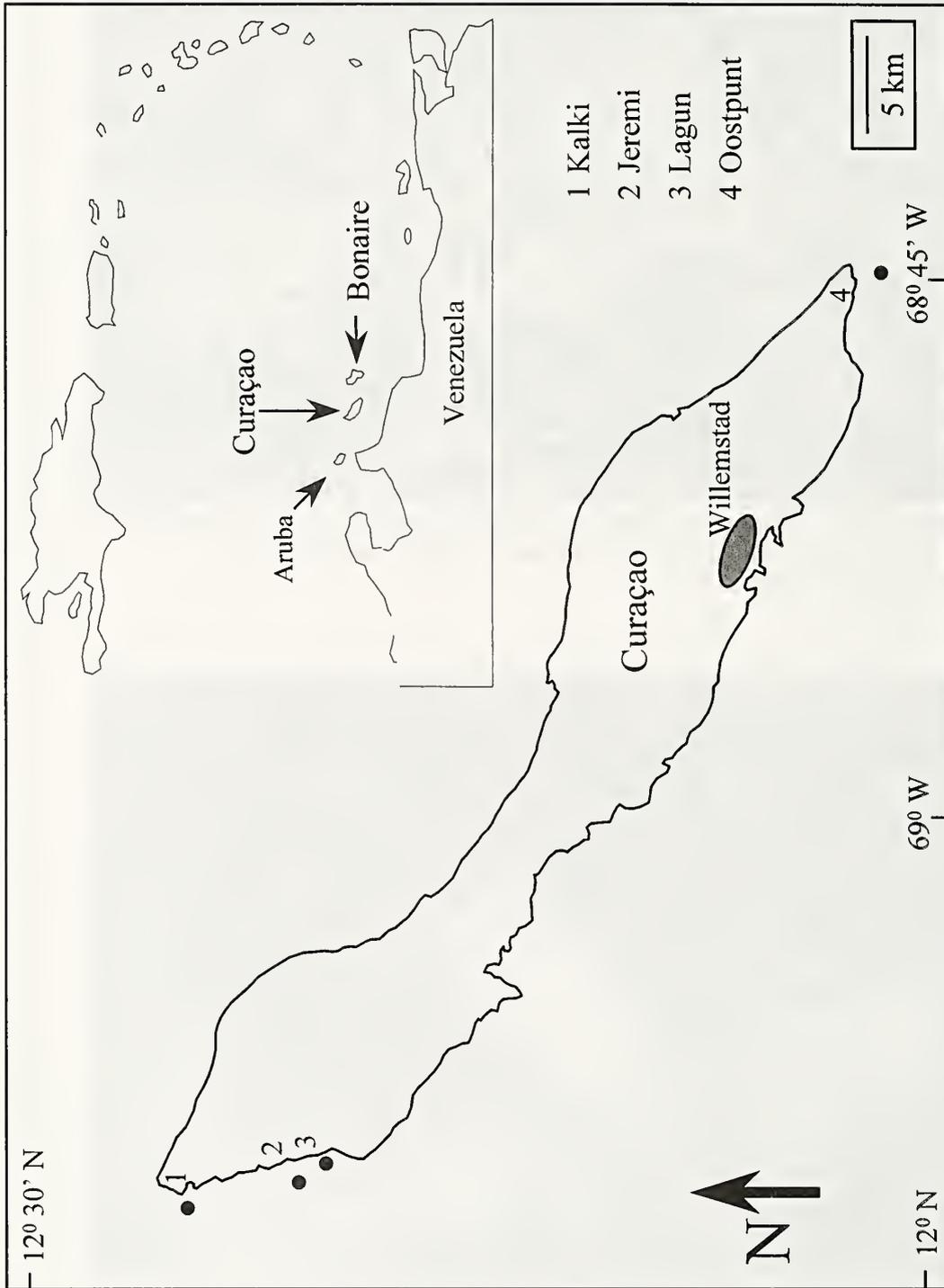


Figure 1. AGRRA survey sites (Kalki, Jeremi, Lagun, Oostpunt) on the leeward coast of Curaçao.

# CONDITION OF CORAL REEFS OFF LESS DEVELOPED COASTLINES OF CURAÇAO (PART 2: REEF FISHES)

BY

ANDREW W. BRUCKNER<sup>1</sup> and ROBIN J. BRUCKNER<sup>2</sup>

## ABSTRACT

Reef fish populations at 9-12 m depth in more remote eastern and western Curaçao were characterized by low abundance, size, and diversity of large-bodied species. Families often contained only a few of the species previously reported in Curaçao and many commercially important food species were absent. Grunts, parrotfishes, surgeonfishes and butterflyfishes were most abundant. The densities of most species averaged less than eight individuals/100 m<sup>2</sup> and their mean sizes were 15-25 cm. Most reefs contained one or two tiger grouper (*Mycteroperca tigris*) and barracuda of larger sizes (~25-35 cm) and several medium- to large-sized (20-28 cm) parrotfish. No major differences were apparent between an underwater park in Oostpunt and the western reefs. The cumulative impacts of heavy fishing pressure, lack of enforcement of a spearfishing ban, along with increased urbanization and degradation of coastal nursery areas, may have contributed to the decline of Curaçao's reef fishes.

## INTRODUCTION

Curaçao, Netherlands Antilles is a small (61 km long; 443 km<sup>2</sup> area) oceanic island situated 60 km north of Venezuela (Fig. 1). The island is oriented on a northwest/southeast axis and has a narrow submarine shelf that slopes steeply into deep water. Fringing reefs surround the island but are best developed along the leeward (south) coast. The island is sparsely populated (155,000 people) with an economy based on an oil refinery and other industries, offshore banking, and tourism. There is a small commercial fishery that targets pelagic species with an estimated catch of 90-180 metric tons/year (Woodley et al., 1997). Most reef fishes for restaurant and grocery markets are imported from Venezuela although subsistence fishing is widespread along the leeward coast.

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Reef conditions and reef fish populations have degraded near population centers due to extensive coastal development, industrial activities, raw sewage discharge, sedimentation, and vessel impacts (Woodley et al., 1997). Reefs at the eastern and western ends of the island are thought to be in better condition as they are well removed from urban and industrial development. Two small fishing villages (Lagun and Westpunt) are found at the western end of Curaçao, but coastal development is minimal and population density is low. The Curaçao Underwater Park extends 21 km eastward from the outskirts of Willemstad (Princess Beach Hotel) to Oostpunt and includes a 12-km stretch of coastline that is undeveloped and uninhabited. The park also contains an extensive (436 hectares) partially enclosed bay (Spaanse Water) where most of the island's mangroves and *Thalassia testudinum* seagrass beds are found (Debrot et al., 1998).

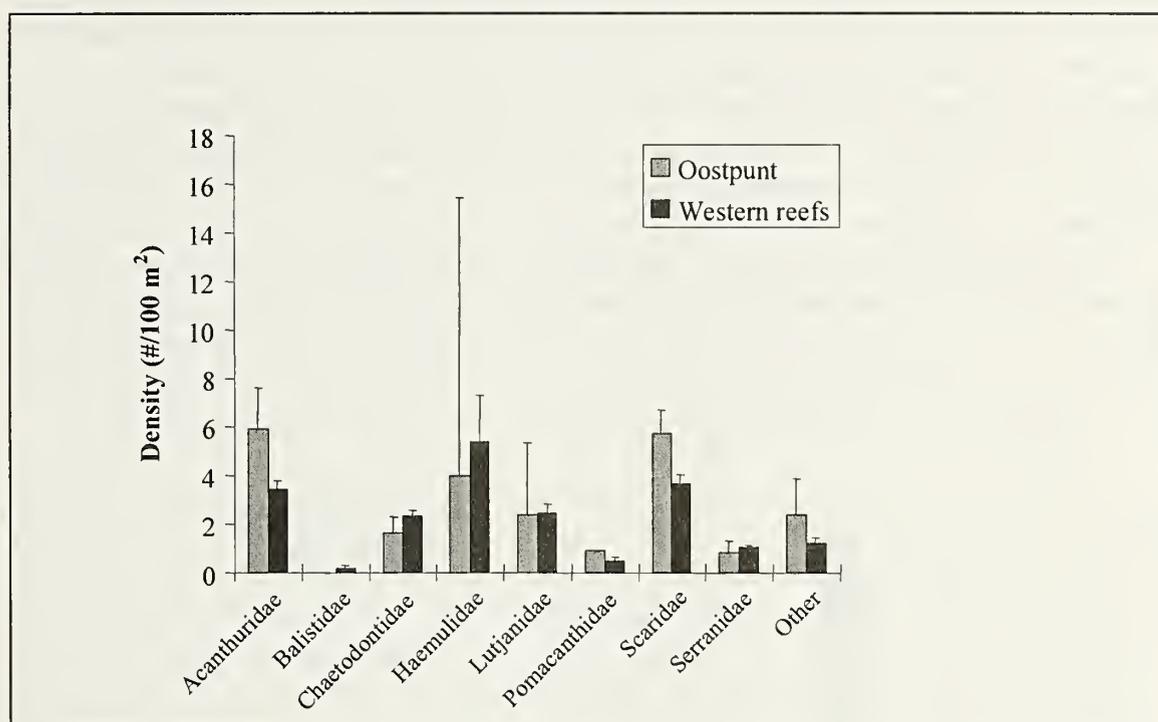
Few quantitative data are available on the reef fish communities of Curaçao (Nagelkerken, 1974; 1977; 1980). However, they are reported to be overfished as a result of heavy artisanal fishing pressure and the use of spear guns, fish traps, and gill nets (Van't Hof et al., 1995). The purpose of this study was to collect baseline information on the status of fish populations in more remote locations of Curaçao at the time of initial enforcement of spearfishing prohibitions and to determine whether differences existed between the marine-park and open-access areas. The Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocol was used to assess the diversity and population dynamics of reef fish with a focus on commercially and ecologically important large-bodied species.

## METHODS

Limited estimates of the abundance and size of reef fish populations were made in four strategically chosen fringing reefs off the less populated, southeast (Oostpunt) and southwest (Jeremi, Lagun and Kalki) coasts of Curaçao by one diver (R. Bruckner) in August 1998. Belt transects were conducted as described in the AGRRA Version 2 fish protocol (Appendix One, this volume) except that temporal constraints restricted the number of transects to three-four at each reef. All transects (30 m long, 2m X 2m window) were run parallel to depth gradients at 9-12 m depth in the same general area as the benthic surveys (Bruckner and Bruckner, this volume). Counts of serranids (groupers) were restricted to species of *Epinephelus* and *Mycteroperca*. Scarids (parrotfishes) and haemulids (grunts) less than 5 cm in total length (TL) were not tallied and only one damselfish, *Microspathodon chrysurus*, was recorded. Previous training dives (July 1998; n=10 dives) were conducted in Puerto Rico using styrofoam fish models (for estimating sizes) and practice transects to obtain consistency in sampling area, swimming speed, and species identification. *Reef Fish Identification* by Humann (1997) was used to confirm species identities.

## RESULTS

Most families of large-bodied reef fishes included in the AGRRA protocol were represented in the remote reefs of eastern and western Curaçao although the species richness and abundance of each were relatively low at all sites (Table 1; Fig. 2). Lagun and Oostpunt had the highest numbers of species (26), while Jeremi had the highest density (57 individuals/100 m<sup>2</sup>). Herbivores were prominent in all reefs. Overall, parrotfishes were more common than surgeonfishes (acanthurids), but there was a notable absence of schools containing more than 6-10 adult ( $\geq 10$  cm TL) fish. *Sparisoma viride* and *Scarus taeniopterus* were the dominant parrotfishes; *S. rubripinne*, *S. aurofrenatum* and *S. croicensis* were also common and juveniles (<10 cm TL) of these and other species were abundant in all locations.

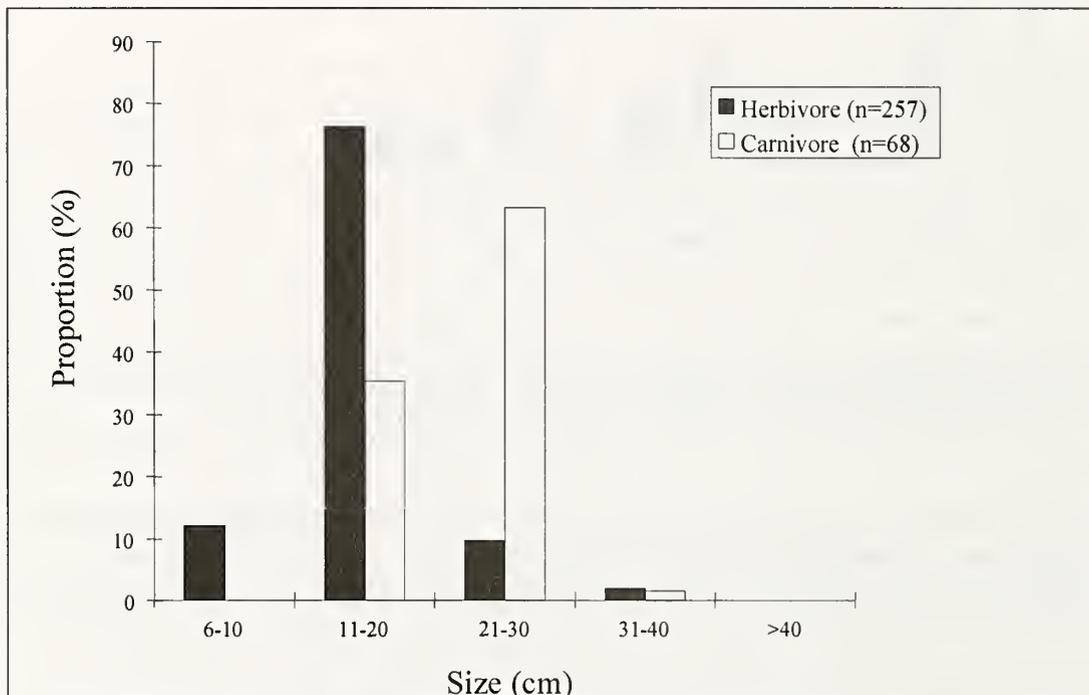


**Figure 2.** Mean fish abundance (no. individuals/100 m<sup>2</sup> ± sd) for AGRRA fishes in eastern and western Curaçao. Other = *Bodianus rufus*, *Microspathodon chrysurus*, *Sphyraena barracuda*.

There were no substantial differences between Oostpunt and western reefs in the overall abundance of reef fishes [pooled by family; two-factor Analysis of Variance (ANOVA); MS=0.8, F=0.95, p=0.36] although the abundance of individual families did vary (ANOVA; MS=6.6, F=8.1, p=0.004). Post-hoc analysis of major groups (pooled into herbivores, major predators, and other predators; Tukey HSD multiple comparison test) indicated that herbivores were significantly more abundant at Oostpunt with minor, non-significant differences among other groups (Fig. 2). The most abundant group of fish observed at Jeremi, Oostpunt and Lagun overall were grunts, which formed large resting

schools at the base of large coral heads, the dominant species being *Haemulon flavolineatum* and *H. aurolineatum*. Commercially important *H. plumieri* were encountered only at Oostpunt. Grunts were relatively uncommon at Kalki although three species (*Anisotremus surinamensis*, *H. sciurus* and *H. flavolineatum*) were observed. Oostpunt had a larger number and species of angelfishes (pomacanthids) and butterflyfishes (chaetodontids) than were recorded within transects on other reefs. No angelfish were recorded within transect areas at Jeremi whereas *Chaetodon capistratus* and *C. striatus* were present in all four locations. Among the leatherjackets (balistids), *Cantherines pullus* was recorded within a belt transect only on Jeremi. Filefish (*C. pullus*, *C. macrocerus*, *Aluterus scriptus*) were found in areas away from transects at the same depth on all reefs, and *Melichthys niger* commonly occurred in the water column at Oostpunt. *Balistes vetula* were also absent from transects in all locations but individuals were observed among gorgonians in shallow water (3-5 m depth) at Oostpunt. In addition, large schools of *Caranx ruber* and *C. crysos* were frequently sighted in the water column but not within transects. Important top predators were generally uncommon. Each reef had one-two individuals each of tiger grouper (*Mycteroperca tigris*) and barracuda (*Sphyraena barracuda*). The most abundant snapper was *Lutjanus apodus* which occurred on all reefs but was most common at Lagun. Other predators recorded on most reefs at a low abundance include *Epinephelus guttatus*, *E. cruentatus*, *Lutjanus apodus*, *L. griseus* and *L. mahogoni* and, at Oostpunt only, *Ocyurus chrysurus*.

The mean total length of all fishes recorded within the belt transects was generally 15-25 cm (Table 2). The largest predators were *M. tigris*, *S. barracuda* and *L. apodus* (25.5-35.5 cm). Large stoplight parrotfish (*S. viride*) were identified at Jeremi (28 cm), Oostpunt (24 cm) and Kalki (21.5 cm), but this species was smaller at Lagun (18 cm) and



**Figure 3.** Size frequency distribution of herbivores (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) and carnivores (all lutjanids, select serranids) in eastern and western Curaçao.

other species of parrotfish were mostly small (12-20.5 cm). Most grunts were small (8-15.5 cm) with the exception of a single *Haemulon macrostomum* at Oostpunt (25.5 cm) and one *A. surinamensis* (25.5 cm) at Kalki (Table 2). The size frequency distribution of commercially important carnivores (snappers and select groupers) and key herbivores (parrotfish, surgeonfish and yellowtail damselfish) are presented in Figure 3.

## DISCUSSION

The ecologically and commercially important species of reef fishes included in the AGRRA species list exhibited a low diversity, abundance and size in the four locations examined in Curaçao. The dominant families seen during these surveys were Haemulidae, Scaridae and Acanthuridae, although one snapper (*Lutjanus apodus*) was abundant at a single location in western Curaçao. Typically, each family was represented by one-four species with the exception of parrotfishes, for which three-six species occurred in each reef. In each reef the top predators consisted of one-two tiger groupers and one-two barracudas, although these were fairly small in total length (25-35 cm). Other medium-sized fishes that were often equally or more abundant than the AGRRA-listed species included goatfish (Mullidae), squirrelfish and soldierfish (Holocentridae), and wrasses (Labridae) (R. Bruckner, unpub. obs.). From the historical data available for Curaçao, it is apparent that many commercially important groupers, snappers, grunts, hogfish (*Lachnolaimus maximus*) and other species that were formerly common on these reefs (Nagelkerken, 1974, 1977, 1980, 1981) were absent or occurred at very low numbers during the present survey.

While reef fish populations are known to be heavily overfished throughout Curaçao, reefs of Oostpunt are reported to be in the best condition (Van't Hof et al., 1995). Oostpunt's reefs are located off an undeveloped coastline that is upstream from terrestrial sources of pollution. In addition, fish populations are offered limited protection by their inclusion in a marine park and reefs are in close proximity to important nursery areas. The mangroves, seagrass beds and shallow-reef communities in Spaanse Water were identified as the most important nursery biotope for many of the species found on fringing reefs near Oostpunt, and 32 species on the AGRRA list were identified here (Nagelkerken et al., 2001). However, there did not appear to be substantial differences between reef fish populations at Oostpunt and those at the western end of the island in terms of species composition and size. In addition, it was difficult to approach most fish in all locations with exception of small resting schools of grunts, butterflyfishes, and the small herbivores.

Several factors appear to have contributed to the island-wide decline of reef fish populations. Heavy fishing pressure and the use of harmful gear (fish pots, gillnets and spearguns) are widespread and, until recently, were not managed (Van't Hof et al., 1995). Although Oostpunt is designated as a park, the island lacks no-take marine-protected areas (MPAs) and hook-and-line fishing is permitted within park boundaries. Until 1998, spearfishing was prevalent throughout Curaçao even though it had been prohibited more than 20 years earlier. As early as 1984, researchers reported a decline in abundance and size of groupers compared to the neighboring island of Bonaire which was thought to be

caused by spearfishing (Pors and Nagelkerken, 1998). The other major factor affecting fish populations may be the loss or degradation of important nursery areas. Rapid urbanization of Spaanse Water has been associated with the degradation of mangroves and seagrass beds and a precipitous decline of its coral populations (Debrot et al., 1998). The long-term effects of these changes on reef fish populations are largely unknown. However, in a recent survey within Spaanse Water, groupers were completely absent and other commercially important species, such as white grunts (*Haemulon plumieri*), margates (*H. album* and *Anisotremus surinamensis*) and most snappers, occurred at very low densities (Nagelkerken et al., 2001).

This study provides only a very limited snapshot of four reefs in the more remote locations of Curaçao and more extensive surveys are necessary to obtain a representative picture of the overall diversity, size and abundance of commercially and ecologically important reef fish. Nevertheless, these data may provide a baseline of their status at the onset of a full enforcement of the spearfishing prohibition. The elimination of spearfishing represents a critical step towards protecting and restoring Curaçao's reef fish populations but there is a need for other tools such as the establishment of no-take marine protected areas (Bohnsack et al., in press). It is hoped that this study will help convince resource managers and policy makers of the need for additional management measures for Curaçao's reefs (Van't Hof et al., 1995).

## ACKNOWLEDGMENTS

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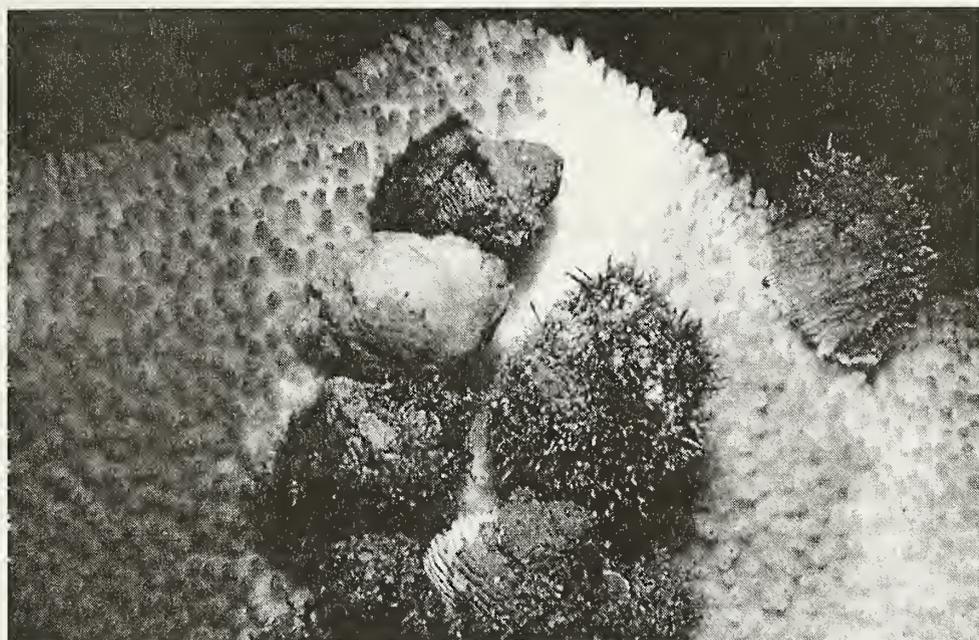
Table 1. Site information for AGRRA fish surveys in Curaçao, Netherland Antilles.

Site name	Reef type	Latitude (° ' " N)	Longitude (° ' " W)	Survey date	Depth (m)	% live stony coral cover (mean ± se)	AGRRA fishes		Species (#)
							30 m transects (#)	Mean density (#/100 m <sup>2</sup> )	
Oostpunt	Fringing	12 02 29.4	68 44 59.8	Aug 15 1998	9-12	27 ± 2.9	3	80.6	26
Lagun	Fringing	12 19 6.9	69 09 54.0	Aug 13 1998	10-11	25 ± 3.0	4	62.5	26
Jeremi	Fringing	12 19 44.3	69 09 1.2	Aug 14 1998	10-11	31 ± 3.9	3	94.4	24
Kalki	Fringing	12 22 31.3	69 09 29.8	Aug 14 1998	9-12	40 ± 4.1	3	54.4	21

Table 2. Number of species, plus density and length (mean ± standard deviation) of AGRRA fishes in four reefs off Curaçao.

Family name	Site Names											
	Oostpunt			Lagun			Jeremi			Kalki		
	Spp. (#)	Density (#/100 m <sup>2</sup> )	Length (cm)	Spp. (#)	Density (#/100 m <sup>2</sup> )	Length (cm)	Spp. (#)	Density (#/100 m <sup>2</sup> )	Length (cm)	Spp. (#)	Density (#/100 m <sup>2</sup> )	Length (cm)
Acanthuridae	3	17.8 ± 11.1	15.5 ± 0	3	9.2 ± 4.0	14.1 ± 2.9	2	8.9 ± 8.6	15.5 ± 0	3	8.3 ± 6.7	15.0 ± 1.9
Balistidae	0			0			1	0.6	25.5	0		
Chaetodontidae	4	6.7 ± 3.3	10.5 ± 3.5	2	4.2 ± 2.9	8.8 ± 2.4	3	6.1 ± 1.0	8.0	3	8.9 ± 4.2	12.8 ± 3.7
Haemulidae <sup>1</sup>	5	20.0 ± 6.0	14.7 ± 3.2	5	16.7 ± 22.6	15.7 ± 1.4	4	43.3 ± 20.2	12.6 ± 4.5	3	3.9 ± 5.4	17.7 ± 4.2
Lutjanidae	3	7.2 ± 6.7	21.7 ± 4.9	3	10.4 ± 8.2	22.7 ± 4.5	2	3.9 ± 6.7	24.1 ± 3.5	2	3.9 ± 3.5	21.2 ± 4.9
Pomacanthidae	3	2.8 ± 1.0	17.5 ± 4.0	1	0.8	20.5 ± 5.0	0			1	0.6	15.5
Scorpaenidae <sup>1</sup>	3	17.2 ± 8.2	19.9 ± 5.7	6	16.3 ± 8.3	15.6 ± 4.6	7	25.0 ± 4.4	18.7 ± 8.9	5	23.3 ± 5.4	16.2 ± 3.0
Serranidae <sup>2</sup>	2	1.7 ± 1.7	25.5 ± 0	3	2.9 ± 1.6	25.5 ± 7.6	3	3.3 ± 1.7	20.5 ± 7.6	2	1.7 ± 1.4	18.8 ± 4.0
<i>Microspathodon</i>	1	5.6 ± 5.1	15.5 ± 0	1	1.3 ± 2.5	15.5 ± 0	0			1	2.2 ± 1.9	8.0 ± 0
<i>Barracuda</i>	1	0.6	25.5	1	0.4	35.5	1	1.1	25.5	1	1.1	25.5
<i>Bodianus</i>	1	1.1	15.5	1	0.4	15.5	1	2.2 ± 3.8	35.5 ± 0	1	0.6	15.5
All	26	80.6		26	62.5		24	94.4		21	54.4	

<sup>1</sup>≥ 5 cm only<sup>2</sup>*Epinephelus* spp. and *Mycteroperca* spp.



**Plate 10A.** Areas of recent mortality and adjacent coral tissues are examined closely for evidence of predation by corallivores, such as these *Coralliophila abbreviata* snails that preferentially feed on *Acropora palmata* (as shown) and other acroporids. (Photo Andrew W. Bruckner)



**Plate 10B.** Threespot damselfish (*Stegastes planifrons*) create algal gardens within the living tissues of this *Acropora palmata* and other species of stony corals which they defend against intrusion by other herbivores. Damselfish predation is reported in AGRR by the presence of an aggressive fish or its gardens. (Photo Kenneth W. Marks)

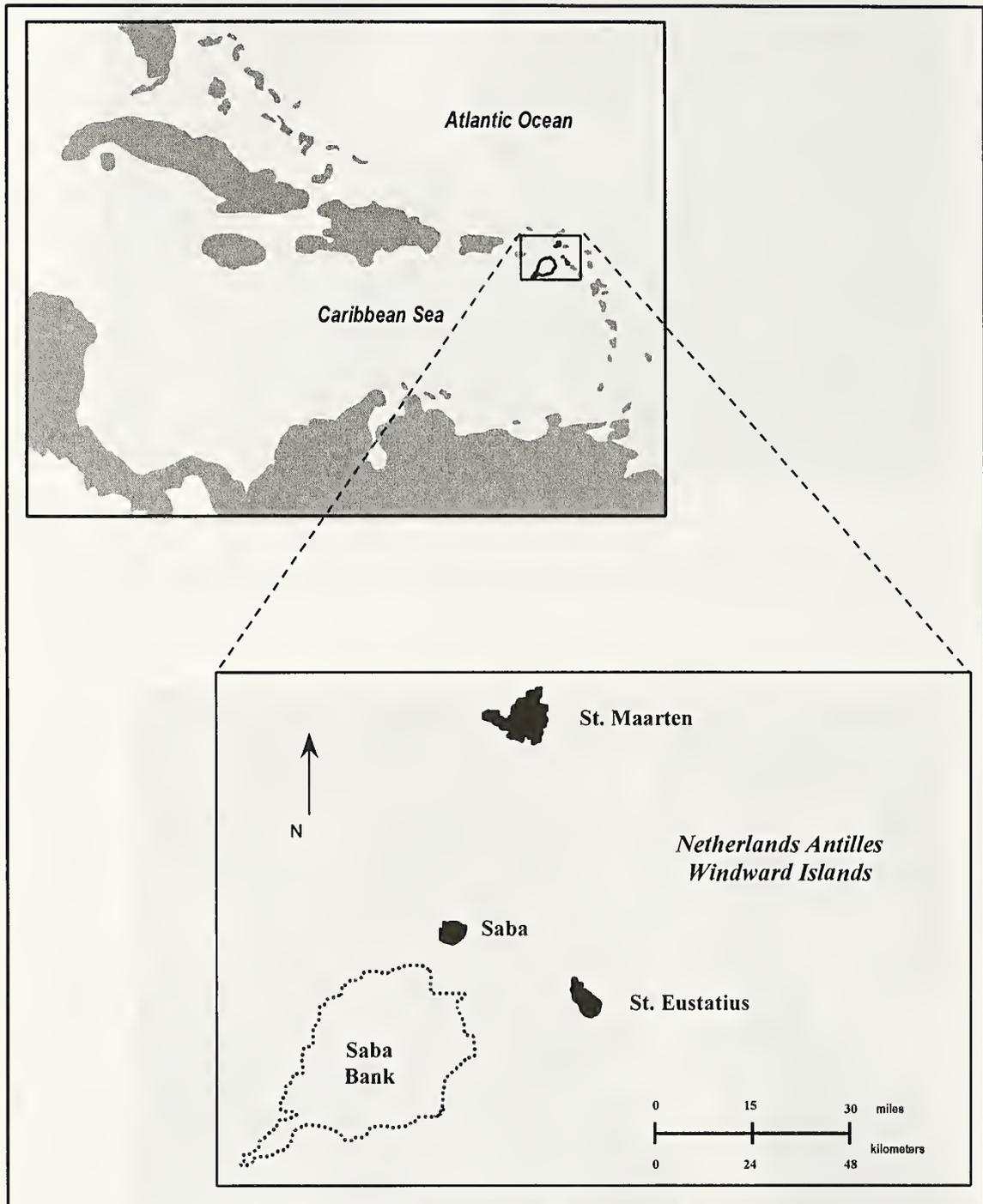


Figure 1. Location of the windward Netherlands Antilles.

# A POST-HURRICANE, RAPID ASSESSMENT OF REEFS IN THE WINDWARD NETHERLANDS ANTILLES (STONY CORALS, ALGAE AND FISHES)

BY

KRISTI D. KLOMP<sup>1</sup> and DAVID J. KOOISTRA<sup>2</sup>

## ABSTRACT

Reefs of the windward Netherlands Antilles (Saba, Saba Bank, St. Eustatius, St. Maarten) were assessed at 24 sites in late 1999. The Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocol was used with modifications to detect recent hurricane impacts. Live coral cover averaged 18%. The assemblage of  $\geq 10$  cm stony corals was primarily composed of small-sized colonies (mean diameter  $\approx 37$  cm) of which the *Montastraea annularis* complex was the most abundant (30% of colonies). Overall,  $\approx 1\%$  of the individually surveyed colonies had been physically damaged by Hurricane Lenny but injury levels were higher in Saba (2.6%). Bleaching was noted in  $>23\%$  of colonies at the time of the assessment with the greatest percentage occurring on St. Maarten (44%) and the lowest on Saba Bank (9%). Total (recent + old) partial mortality of reef-building corals averaged less than 18% although levels were higher (26%) in *Colpophyllia natans*. Coral recruitment densities were relatively consistent (mean  $\approx 5$  recruits/m<sup>2</sup>) across sites. Commercially significant fish species (i.e., serranids, lutjanids, haemulids  $\geq 5$  cm) were present with mean densities of 4.5 individuals/100 m<sup>2</sup>. High biomass (mean  $\approx 5.8$  kg/100 m<sup>2</sup>) of grazing, herbivorous fishes (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) partially explains the relatively low macroalgal cover (mean  $\approx 7\%$ ) throughout this area. Saba's fish community had a greater total biomass than those in the other three geographic areas (mean  $\approx 11$  kg/100 m<sup>2</sup> versus 7 kg/100 m<sup>2</sup>). While the coral reefs of St. Maarten show signs of disturbance (i.e., increased bleaching and sedimentation), those of Saba, Saba Bank, and southern St. Eustatius have been relatively little disturbed by coastal development and remain potential sources of marine life. Nevertheless, reef development in the windward Netherlands Antilles is limited by frequent hurricanes.

## INTRODUCTION

The windward Netherlands Antilles (N. A.) are located at the northern arc of the leeward Lesser Antilles in the Caribbean Sea (Fig. 1). Included in the windward N. A.,

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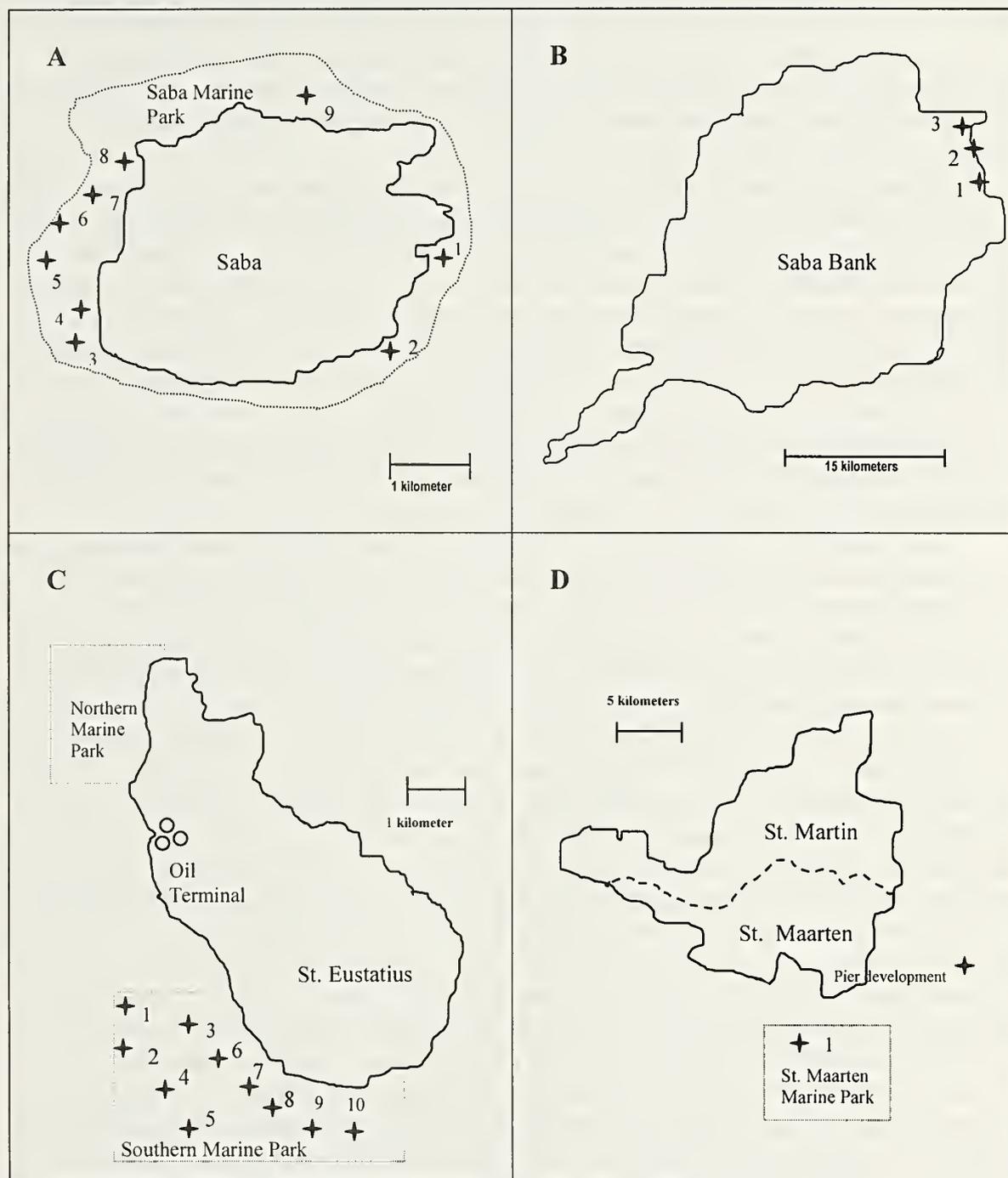
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and separated from each other by distances of 20-50 kilometers, are Saba (Fig. 2A), Saba Bank (Fig. 2B), which is totally submerged, St. Eustatius (Fig. 2C), and the southern portion of St. Martin/St. Maarten (Fig. 2D). As each represents a distinctly different area in terms of geophysical aspects and degree of human impact, they are described separately below.

Saba (17°36'N, 63°15'W) is an isolated volcanic island of late Pleistocene to mid-Holocene origin (Westermann and Kiel, 1961). Saba does not rest on a carbonate shelf but a narrow submarine platform fringes the island before sloping steeply to the deep sea floor. Devoid of sandy beaches, Saba's coastline is formed of steep, rocky cliffs. Rapid erosion of this coastline and Saba's constricted shelf do not allow extensive reef development. Coral communities circumfuse the island occurring on granite boulders, pinnacles, and lava formations that extend seaward from the island. Close to shore, scattered coral colonies encrust large boulders that have eroded from the steep, coastal embankments. Extending seaward along Saba's gently sloping (<10°) insular shelf are lava-flow formations aligned perpendicular to the coastline. These volcanic features, often separated from each other by sand channels, provide a basal structure for the development of coral communities. In some areas where corals have, almost contiguously encrusted lava flows, the topography superficially resembles spur-and-groove formations.

Saba's villages are located above 240 m altitude (Buchan, 1998) and coastal development is limited to a small harbor that accommodates the marine-park office, the island's dive operators, and a recently closed rock-crushing plant. While Saba has become a popular dive destination in recent years, there appears to be minimal human impact on its reefs. (Although the authors acknowledge that Saba's coral communities do not fit the narrow definition of coral reefs in the technical sense, we have elected to use the term throughout this paper for the sake of fluidity). However, Saba's corals are physically damaged by hurricanes (most recently Hurricanes Luis in 1995, Georges in 1998, Lenny in 1999), especially the *Acropora palmata* in some shallow, high-surge zones. An area surrounding the island from the high-water mark down to a depth of 60 meters is actively regulated by the Saba National Marine Park which was established in 1987.

Saba Bank (17°25'N, 63°30'W) is a submarine plateau that has been described as a sunken atoll (Vaughan, 1919; Van der Land, 1977). Its eastern edge lies about 3-5 km SW of Saba. Saba Bank is 60 to 65 km long and 30 to 40 km wide and, with a total surface area of about 2200 km, ranks among the largest atolls in the world. Rising about 1000 meters above the surrounding sea floor, it reaches a plateau at a depth of about 15 m. Here aggregations of gorgonians and sparsely spaced, small (< 25 cm diameter) scleractinians are arranged linearly in tracts that are 20-40 m wide and separated by 10-20 m-wide sand channels. Depths vary between 7 and 20 m near its eastern and southeastern rim. The western ridge reportedly is deeper (50 m) and dominated by sand. From the bank's eastern rim at ≈20 m depth, the bottom terrain slopes seaward to meet a terrace before it drops precipitously beyond 35 m depth. Corals are more densely spaced along



**Figure 2.** AGRRA survey sites for (A) Saba, (B) Saba Bank, (C) St. Eustatius, (D) St. Maarten, in the windward Netherlands Antilles. See Table 1 for site codes.

this slope (i.e., between 20-35 m depth) than at our sampling depths of 15-20 m. As depth increases, colonies become progressively larger and, for many of the *Montastraea* and *Agaricia*, increasingly flattened in morphology. Van der Land (1977), who referred to this coral-covered slope as the "front reef," noted that Saba Bank was an island during the last glacial period and until at least about 5,000 years ago. He hypothesized that this windward "front reef" originated, when sea levels were 20-30 m below present, as a fringing reef of Saba Bank's island.

Because a considerable distance separates Saba Bank from land, human impact on its corals is restricted to that imposed by fishing activity, boats and ships. Part of the Bank is within the geographic scope of Saba island's jurisdiction as territorial sea; the larger part falls within the limits of a possible Exclusive Economic Zone (EEZ) within the competency of the Netherlands Antilles. At present, only a limited EEZ has been claimed by the Netherlands Antilles in the form of an Economic Fisheries Zone. Van der Land (1977), Meesters et al. (1996) and Dilrosun (1999) have ascertained that, in isolated areas, Saba Bank's reefs are rich in terms of cover and diversity of reef-building corals but that its fish stocks appeared to be declining. However, its reefs have remained largely unexplored. Hence, Saba Bank provides us with a rare opportunity to study the regional potential for reef development in the absence of coastal processes.

Sint Eustatius (17°29'N, 62°57'W) is situated on a submerged platform shared with the islands of St. Kitts and Nevis. The island consists of two extinct volcanoes, an older (late Pliocene) eroded volcano in the northwest and a younger volcano with evidence of activity as recent as the early Pleistocene epoch (Adey and Burke, 1976) in the south. A narrow platform (<2 km wide) surrounds most of the island. Fringing reefs occur mid-shelf off its southern and northwestern coasts. The remainder of the shelf is a flat, sandy plateau with limited potential for coral development. Lava flow formations from the southern volcano extend seaward and, along with large volcanic boulders, provide a high-relief (≈3-4 m) structural base for reefs on the island's southern side. These reef structures, each about 30-50 m wide, are separated by sand channels of ≈50 m width and frequently occur in a series. St. Eustatius' most robust reefs in terms of coral cover occur mid-shelf near the island's southernmost tip. Here the underlying substrata on which the corals occur is carbonate in nature but it is not apparent from casual observation whether these are entirely of carbonate composition or whether they embody inorganic, pyroclastic fragments. While typically spanning an area similar in size (30-100 m in any direction), they are not as consolidated as lava flows nor do they exhibit the uniform orientation ("seaward") of lava flow. However, these formations are equal in vertical relief (rising 3-4 m from the sand-covered bottom) and, where disjunct, are often cemented together with mature (>100 cm diameter) colonies of *Montastraea*. Corals also fringe the rocky northwestern coast near shore where broken skeletons of *A. palmata* and live stands of *Millepora* occur on boulders in the high-surge zones. Further from shore, scattered coral heads occur along a reef flat, though most are dead and covered with sediment.

The coastline of St. Eustatius is relatively undeveloped with the exception of its capital city, Oranjestad, and a trans-shipment oil terminal, both located on the western coast. The St. Eustatius Marine Park was established in January 1998 and is operated by the St. Eustatius National Park Foundation (STENAPA). Two areas, the Northern Marine Park and the Southern Marine Park, are actively protected by St. Eustatius Marine Park

officials and fishing activity is restricted in these zones. St. Eustatius' reefs have been impacted by hurricanes in recent years and those in the Northern Park have been severely impacted by anthropogenically enhanced sedimentation.

Saint Martin/Saint Maarten (18°N, 63°W) is situated on the Anguilla Bank, a microplate shared by the islands of Anguilla and St. Barthelemy. The Anguilla Bank is comprised of an old (early to mid-Tertiary period) basal rock of fine mudstones and the islands represent an uplifted veneer of young (Pleistocene to Recent) reef limestones and sandstones (Oxenford and Hunte, 1990). A fringing reef flat occurs mid-shelf ( $\approx 2$ -3 km) off the southern coast of the island and fringing reefs are found adjacent to small islets located at similar distances from St. Maarten's southeastern coast. The reefs off the southern coast emerge from the surrounding hardbottom at depths of about 8-15 m. Most of this system is characterized by low-relief, scattered corals but a spur-and-groove system of intermediate relief ( $< 2$  m) occupies a small area. These reefs are routinely exposed to strong offshore currents. Remnants of *Acropora palmata* are apparent within the spur-and-groove zone but most colonies are either "standing dead" (i.e., entirely dead but still attached to the substratum) or, if still living, have spread horizontally without having attained much height above the substratum. The reefs which fringe the small islets on St. Maarten's southeastern side grow compactly along a slope ( $< 30^\circ$ ) with no lagoonal area separating them from the islets.

Tourism is a primary contributor to St. Maarten's economy. St. Maarten is a major port of call for Caribbean cruise ships and, at the time of this survey, operations were under way to expand harbor capacity to accommodate additional vessels. As the island's infrastructure struggles to meet the demands of development, St. Maarten's reefs are increasingly threatened by devegetation, siltation, sewage input, recreational boating (Nijkamp et al., 1995), and anchor damage (Smith et al., 1997). Additionally, Smith et al. (1997) reported that heavy seas during Hurricane Luis (1995) resulted in resuspended sand, smothering reefs and damaging *A. palmata* in shallow water. The Marine Park of St. Maarten, established in 1997 in conjunction with the Nature Foundation St. Maarten, encompasses a portion of its reefs. At the time of this assessment, the marine park was in its initial stages of devising a management plan. One of our survey sites (SXM01), which lies within the marine park boundaries (Fig. 2D), has been the site of two cruise ship groundings in recent years.

The windward N. A. are frequently impacted by hurricanes. During the previous decade, a succession of severe storms (Hugo in 1989, Luis and Marilyn in 1995, Bertha in 1996, Georges in 1998, José and Lenny in 1999) had come near enough to impose harm to the reefs. Prior to Hurricane Luis in 1995, *Acropora palmata* was the predominant scleractinian on many of Saba's shallow ( $< 10$  m) reefs (i.e., at sites 1, 2, 8, 9; Fig. 2A). Although some recovery of *A. palmata* was evident following Hurricane Luis, they sustained further damage during Hurricanes Georges (1998) and Lenny (1999). Hence, at many Saba sites, *A. palmata* has been replaced by gorgonians and *Millepora* spp. (Kooista, personal observations).

Our surveys were conducted less than two weeks after the passage of Hurricane Lenny, a category 4 (Safir/Simpson scale) hurricane, which coursed between St. Maarten and Saba on November 18 and 19, 1999. Coupled with the impact of Hurricane José (category 2), which had earlier passed through the windward N. A. islands in October

1999, we were provided with an opportunity to assess damages associated with recent hurricanes.

## METHODS

The surveys were conducted in the windward N. A. (Fig. 1) during November and December 1999. Data were collected from nine sites along the coast of Saba (Fig. 2A), three sites along the eastern rim of Saba Bank (Fig. 2B), 10 sites along St. Eustatius' southwestern shore (Fig. 2C), and two sites along St. Maarten's southern and southeastern shore (Fig. 2D). These sites were among those chosen during a reconnaissance evaluation in October 1999 to be representative of reefs within the area and to strategically accommodate marine park management objectives. Due to reduced visibility associated with storm-induced rainfall and runoff, we omitted sampling where conditions were prohibitive including the reefs in St. Eustatius' Northern Marine Park.

Sites were surveyed using the AGRRA protocols version 2.1 (see Appendix One, (this volume)). The benthic surveys were made by four divers. In quantifying individual stony corals, we modified the method to include all colonies  $\geq 10$  cm in maximum diameter to ensure adequate sample sizes. Coral sizes were measured to the nearest cm. Additionally, we recorded incidence of recent hurricane damage along each benthic transect as indicated by overturned, broken, or abraded coral colonies. Stony corals were searched for damselfish algal gardens and the numbers of benthic damselfish along the transect line were also counted. The *Montastraea annularis* complex was treated as a single species with three morphologically distinct formae. Sediment in the algal quadrats was wafted away by hand before scoring the relative abundance of crustose coralline algae. Macroalgal heights were measured to the nearest cm. Two divers conducted all the fish surveys between 9:00 a.m. and 4:00 p.m. Counts of serranids (groupers) were restricted to species of *Epinephelus* and *Mycteroperca*; scarids (parrotfishes) and haemulids (grunts) less than 5 cm in length were not tallied. Roving Diver observations averaged 30-45 minutes each. Fish biomass was estimated using length-weight relationships given in Appendix Two (this volume). Species recorded during the roving dives were assigned to trophic guilds developed by Schmitt et al. (1998) based on the major source of food in their diet determined from prior studies (Randall, 1967). Field guides included Humann (1992, 1993, 1994), Goodson (1976), Littler et al. (1989), and Smith (1948).

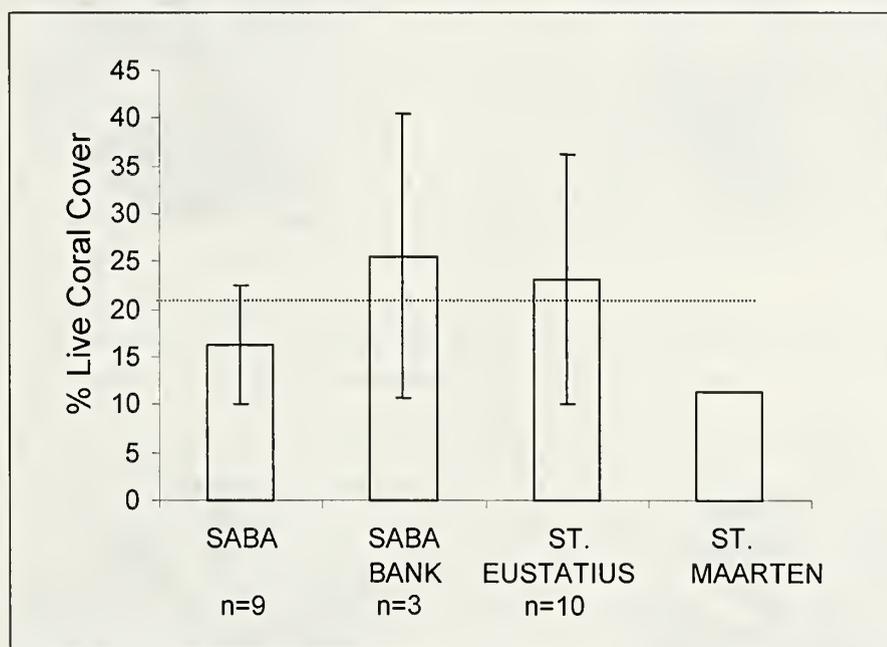
Parameters were statistically analyzed to detect differences between the four geographical areas using Statistica software, Version 5.1 (StatSoft, Inc., 1998). To reduce heteroscedasticity of variance, a single-factor, Kruskal-Wallis nonparametric Analysis of Variance (ANOVA) by rank was used. Kruskal-Wallis nonparametric ANOVA by rank was also used to detect differences among sites within each of the four geographical areas. The Tukey HSD for unequal N was used, when relevant, as an *ad hoc* multiple comparison test. Significance was tested at  $\alpha = 0.05$ . Coral species composition and coral size distribution were analyzed using a chi-square analysis to detect proportional differences among the four geographical areas. Where no significant differences were found, the data were combined as a single value for the windward N. A. Fish densities and biomass were calculated for sites within the four areas and collectively for the

windward N. A. A linear regression was performed to examine the relationship between the biomass of herbivorous fishes (i.e., acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) and macroalgal abundance.

## RESULTS

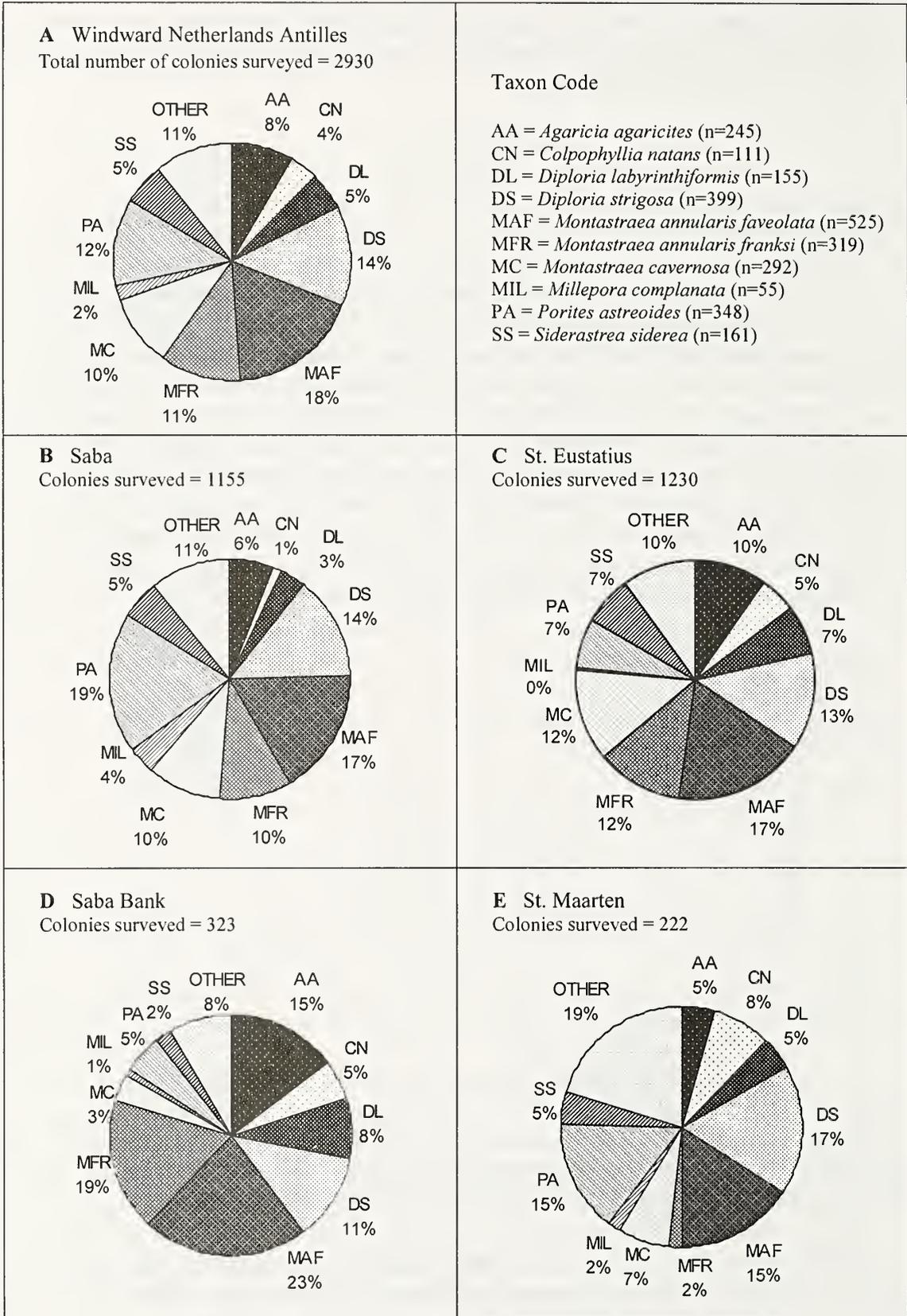
### Stony Corals at the Reef Sites

The 284 benthic transects were made in depths of 6-20 m at the 24 survey sites (Table 1). Mean live coral cover for the windward N. A. (Fig. 3) was 18% (se = 5.7). Though not significantly different from the other three geographical areas (ANOVA,  $p = 0.33$ ), the lowest live coral cover ( $11\% \pm 2.4$  se) was found in St. Maarten. Significant, between-site differences in live coral cover (Table 1) were evident in the three remaining areas (for each ANOVA,  $p < 0.001$ , Tukey  $p < 0.05$ ). Sites SAB03 and SAB09 in Saba (mean =  $8\% \pm 1.1$  se) were significantly lower than the other Saban sites (mean =  $20\% \pm 3.9$  se), but comparable in live coral coverage to the St. Maarten sites (ANOVA,  $p = 0.12$ ). All three sites on Saba Bank differed significantly from each other. Sites EUX01 through EUX08 in St. Eustatius were statistically similar (mean =  $16\% \pm 1.6$  se) and substantially lower than EUX09 and EUX10 (mean =  $46\% \pm 1.5$  se).



**Figure 3.** Mean live stony coral cover (percent  $\pm$  se) for each geographical area in the windward Netherlands Antilles. Dotted line represents overall mean. n=number of sites.

The 2,930 colonies of stony corals, each at least 10 cm in diameter, that were encountered along the transect lines represented 24 species of stony corals (23 scleractinians and the hydrozoan *Millepora complanata*). *Montastraea* was the most abundant stony coral genus throughout the windward N. A. (40% of all individually surveyed colonies). *Montastraea annularis faveolata* was the predominant taxon (19%



**Figure 4.** Species composition and mean relative abundance of all stony corals ( $\geq 10$  cm diameter) in (A) windward Netherlands Antilles, (B) Saba, (C) St. Eustatius, (D) Saba Bank, (E) St. Maarten.

of all colonies), and nine species accounted for nearly 90% of the “major reef-building corals” (Fig. 4). *Acropora palmata*, most of which were located near Green Island at SAB09, comprised <1% of these corals. Only two colonies of *A. cervicornis* were encountered during our entire survey.

Results of a chi-square analysis for proportional similarity suggest that the individually surveyed corals in Saba and St. Eustatius are statistically similar in species composition (chi-square,  $p = 0.35$ ). Likewise, those of Saba Bank and St. Eustatius show similar proportions (chi-square,  $p = 0.31$ ). Marginally significant similarities in species composition were detected between Saba and St. Maarten (chi-square,  $p = 0.11$ ). Significant similarities were lacking for the three remaining geographic comparisons (each chi-square,  $p < 0.015$ ).

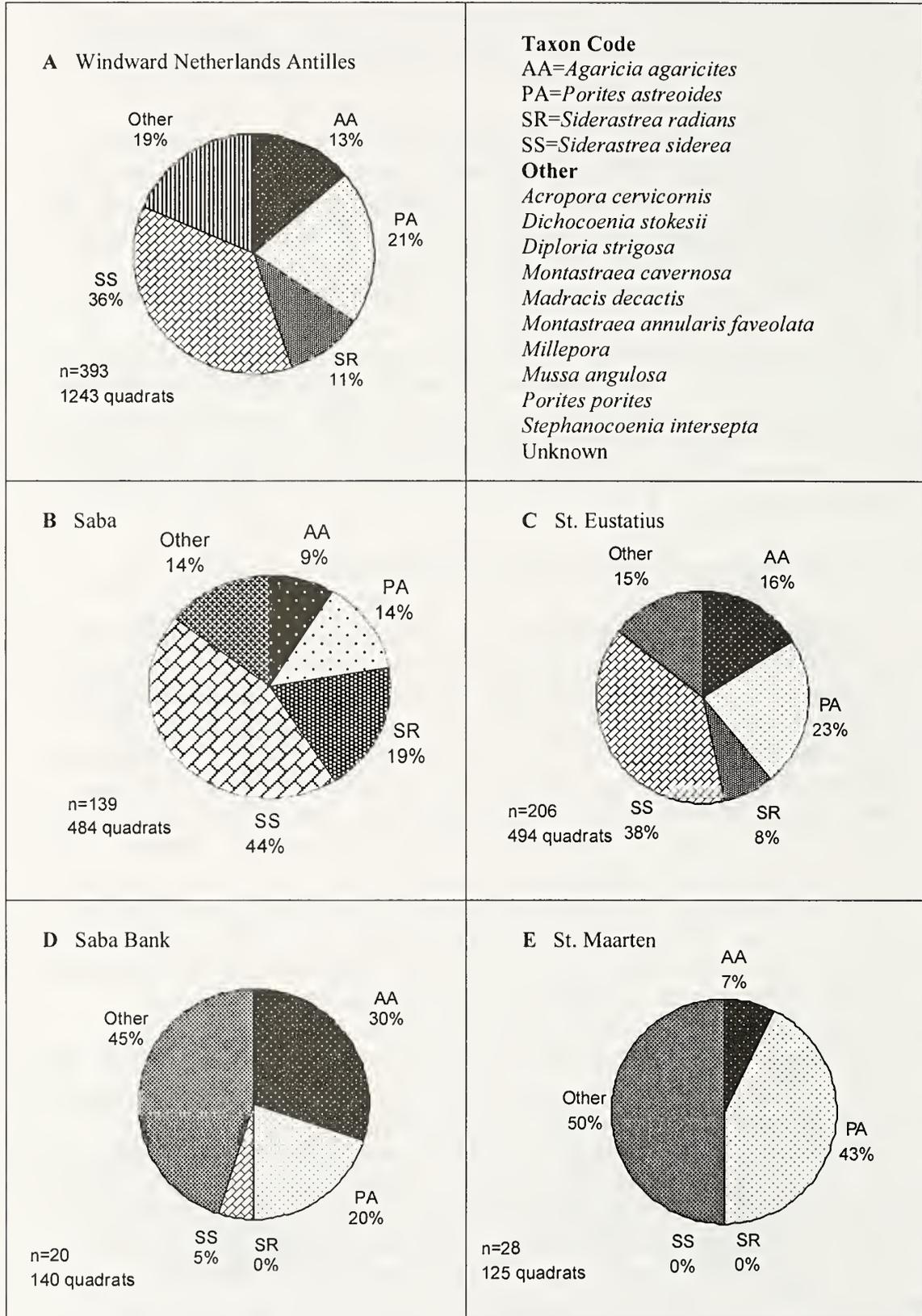
Coral recruits were observed at all sites with an overall mean density equivalent to 4.7 recruits/m<sup>2</sup> (Table 3). Recruitment levels were not significantly different between the four areas (ANOVA,  $p = 0.10$ ) and, although variable between transects, were not significantly different among sites at each of the four geographical areas (each ANOVA,  $p > 0.12$ ). Recruits were composed primarily of *Siderastrea* spp., *P. astreoides*, and *Agaricia agaricites* (Fig. 5).

#### Condition of Individually Surveyed Corals (at least 10 cm in Diameter)

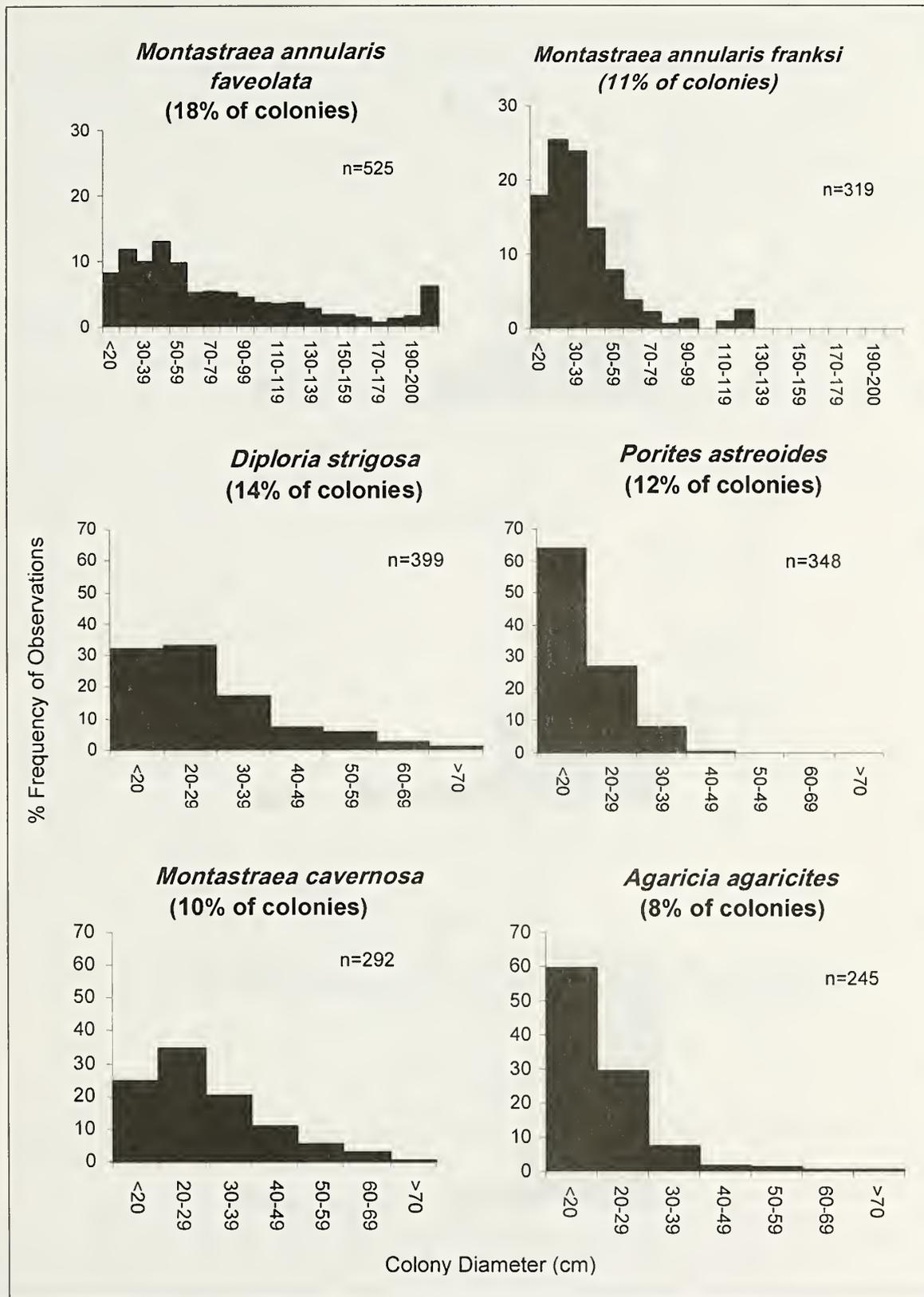
On average, the maximum diameter of the  $\geq 10$  cm corals in the windward N. A. was 37 cm and their height was 19 cm. In Saba, however, many colonies of *Diploria* spp. and *P. astreoides* form thin crusts (<10 cm high) as they spread over boulders that have eroded from the island. Mean colony size was notably greater (Table 2) at three sites (BNK01, EUX09, EUX10) where the species composition was more heavily dominated (44%, 51%, and 43 %, respectively) by *Montastraea annularis faveolata*. In addition, mean diameters of *Montastraea annularis faveolata* were greater at these three sites (105 cm, 122 cm, 122 cm) than was documented at other sites (*M. annularis faveolata*, mean diameter: BNK = 95 cm; EUX = 83 cm).

Size distributions for the five most common coral taxa (Fig. 6) reveal that *M. annularis faveolata* had the largest colonies (maximum diameter = 610 cm). The remaining corals were considerably smaller, especially the two smallest (*P. astreoides*, *A. agaricites*). Sample sizes were adequate to perform a chi-square analysis of size distribution among three geographical areas for four of the common coral taxa. Significant differences in size distribution were only noted in *M. annularis franksi* (Fig. 7), for which a higher proportion of smaller colonies (< 30 cm diameter) was found at St. Eustatius than in Saba and the Saba Bank ( $p < 0.001$ , both locations).

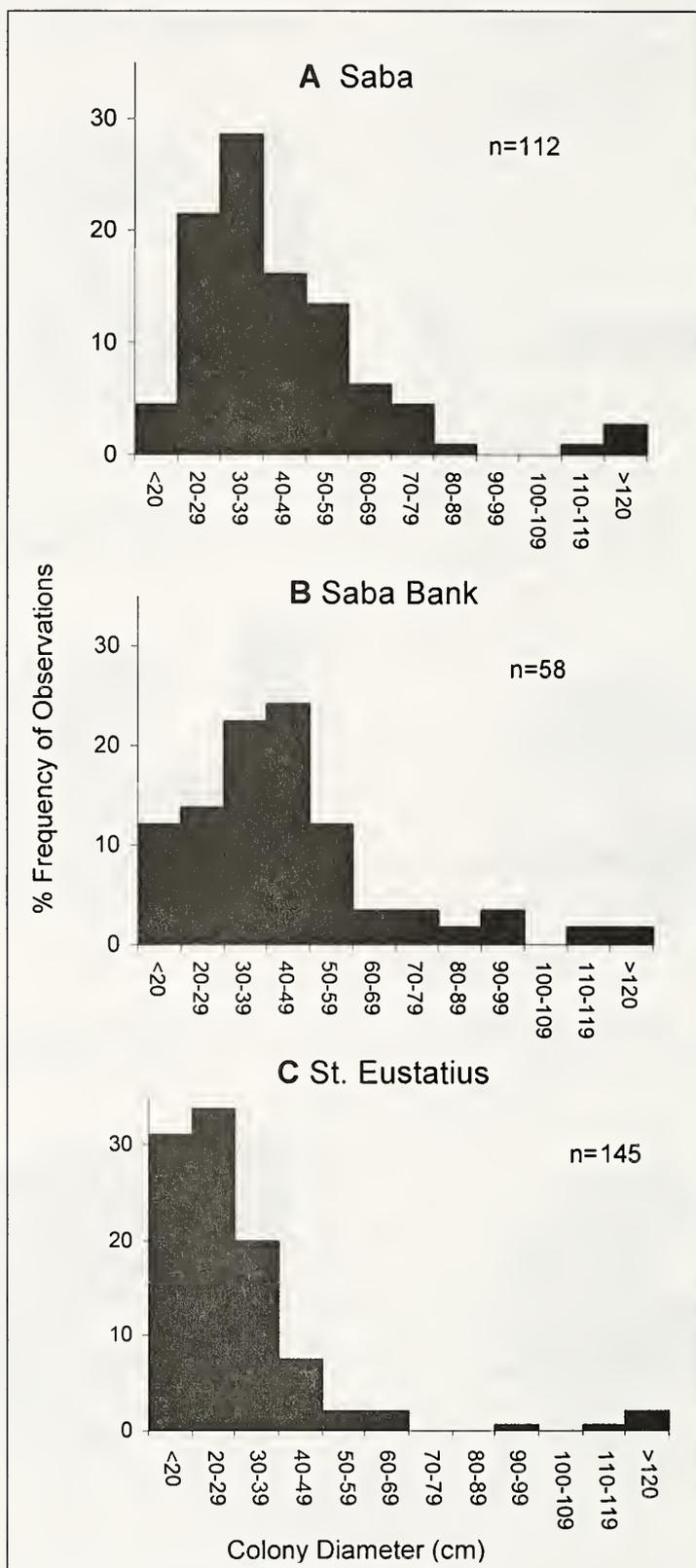
The impact of Hurricane Lenny was evident wherever coral colonies or fragments had been freshly dislodged. Recent storm damage was greatest at Saba, particularly at sites 4, 8 and 9 (where 4.5%, 6.5%, and 4.5%, respectively, of the individually surveyed stony corals were affected), but averaged about 1% for the windward N. A. as a whole (Table 2). It is noteworthy to mention that *Millepora complanata*, although it only represented  $\approx 2\%$  of the surveyed corals, constituted about 40% of the colonies that were damaged in the storm. While physical damage to corals was less severe in St. Maarten than Saba (Table 2), it was apparent that large quantities of sediment had been transported by currents here, burying some structures and exposing others.



**Figure 5.** Species composition and mean relative abundance of all stony coral recruits ( $\leq 2$  cm diameter) in (A) windward Netherlands Antilles, (B) Saba, (C) Saba Bank, (D) St. Eustatius, (E) St. Maarten.



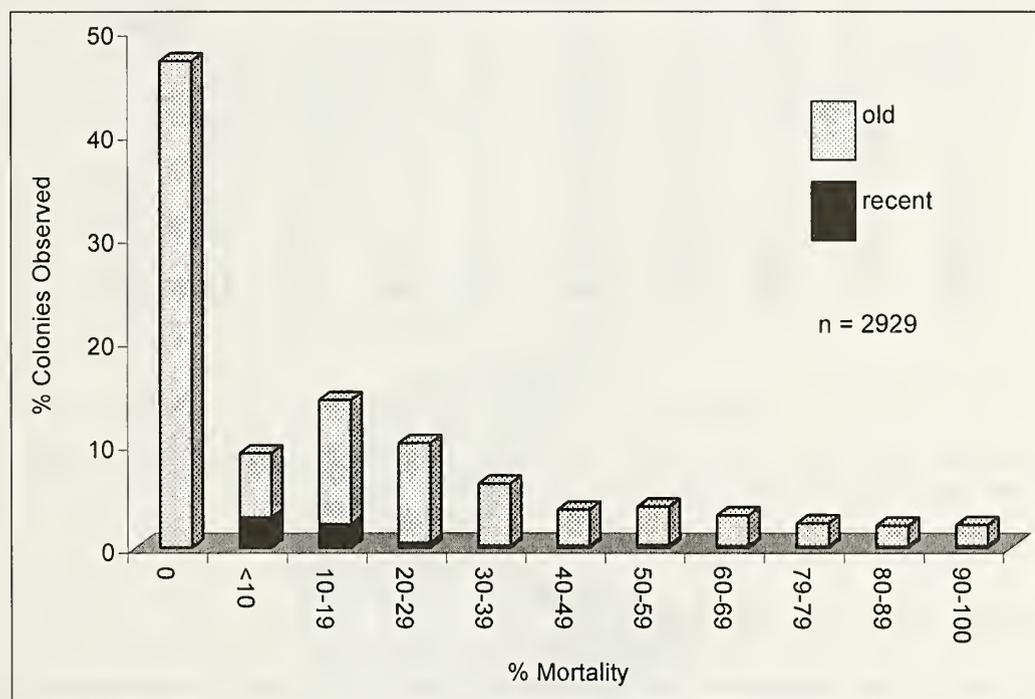
**Figure 6.** Size-frequency distribution of all colonies ( $\geq 10$  cm diameter) of *Montastraea annularis faveolata*, *M. annularis franksi*, *Diploria strigosa*, *Porites astreoides*, *Montastraea cavernosa*, *Agaricia agaricites* in the windward Netherlands Antilles.



**Figure 7.** Size-frequency distribution of all *Montastraea annularis faveolata* ( $\geq 10$  cm diameter) for (A) Saba, (B) Saba Bank, (C) St. Eustatius. St. Maarten was not included in the analysis due to insufficient sample size.

Over 23% of the > 10 cm stony corals exhibited some degree of bleaching (Table 2) but only 1% of these colonies were totally bleached. Of the bleached colonies throughout the N. A., more colonies were partly bleached (12%) than uniformly pale (9%). Bleaching was most prevalent in St. Maarten (44% of all colonies) where the percentage of pale colonies exceeded that found in the other three geographical areas (SXM = 25%; SAB = 10%; BNK = 2%; EUX = 9%). The Saba Bank sites were the least bleached (9% of all colonies). Intermediate levels of coral bleaching were found at Saba and St. Eustatius (22% and 24%, respectively).

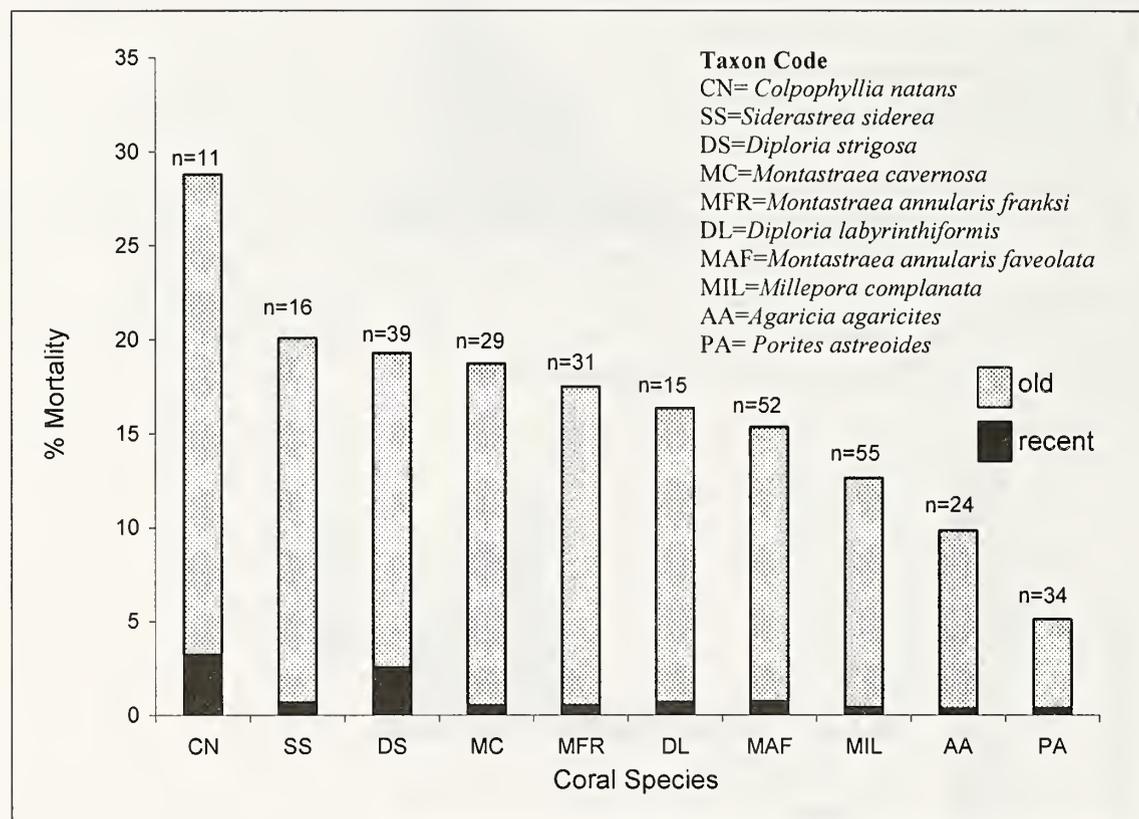
Diseases were noted in about 0.7% of the individually surveyed corals and showed relatively little geographic variation in abundance (Table 2). Yellow-blotch (yellow-band) disease, primarily in the *Montastraea annularis* complex, was the most prevalent disease affecting about 0.4% of the colonies overall. Black-band disease, occurring in 0.1% of the stony corals, affected *Diploria strigosa*, *Agaricia agaricites*, *M. annularis faveolata*, and *M. cavernosa*. Darkspots disease occurred in 0.1% of the colonies and was only seen in *Siderastrea siderea*. White-band disease was seen in *Acropora palmata* and affected 0.1% of the stony corals. White-plague disease was rare and seen in < 0.1% of the colonies of *Colpophyllia natans*.



**Figure 8.** Frequency distribution of total (recent and old) partial colony mortality of all stony corals ( $\geq 10$  cm diameter) in the windward Netherlands Antilles.

Partial-colony mortality of the individually surveyed corals averaged 1.0% for recent mortality and 16% for old mortality (Table 2). Forty-seven percent of these colonies showed no evidence of any kind of mortality and most of the remainder had less than 2% recent and 30% total (recent + old) partial mortality (Fig. 8). Although there were no significant differences among the four areas in recent, old, or total partial mortality (K-W ANOVA,  $p \geq 0.35$ ), partial mortality varied by taxon (Fig. 9). *Porites*

*astreoides* exhibited the lowest levels of recent and old partial mortality (0.5% and 5.0%, respectively). Recent mortality was highest in *C. natans* (3.5%) and *Diploria strigosa* (3.0%). Old (and total) mortality was also substantially higher in *C. natans* (26%). Overall, 0.7% of all surveyed corals were standing dead (Table 2). However, 45% of the *A. palmata* colonies we surveyed were determined to be standing dead and their skeletons represented 56% of the total standing dead in the windward N. A. survey.



**Figure 9.** Mean percent total (recent and old) partial colony mortality of common stony corals ( $\geq 10$  cm diameter) in the windward Netherlands Antilles.

### Algal Groups, *Diadema*, and Damselfishes

Relative algal coverage was assessed for a total of 1,242 quadrats in the windward N. A. Turf algae were the predominant functional group at 13 sites, and were codominants with crustose coralline algae at a further seven sites. Turf relative abundance overall averaged 53%, while the crustose corallines and macroalgae averaged 42% and 7%, respectively. Where macroalgae were present, they displayed a low vertical profile ( $< 2$  cm at 21/24 sites) so that macroalgal indices (% relative abundance of macroalgae  $\times$  macroalgal height—a proxy for macroalgal biomass) everywhere were  $< 40$  (Table 3). Most of the macroalgae we encountered were fleshy species, mainly *Dictyota*, but some calcareous *Halimeda* were also present. No *Diadema antillarum* were found in the benthic belt transects and *Diadema* were only sighted twice during any of our dives.

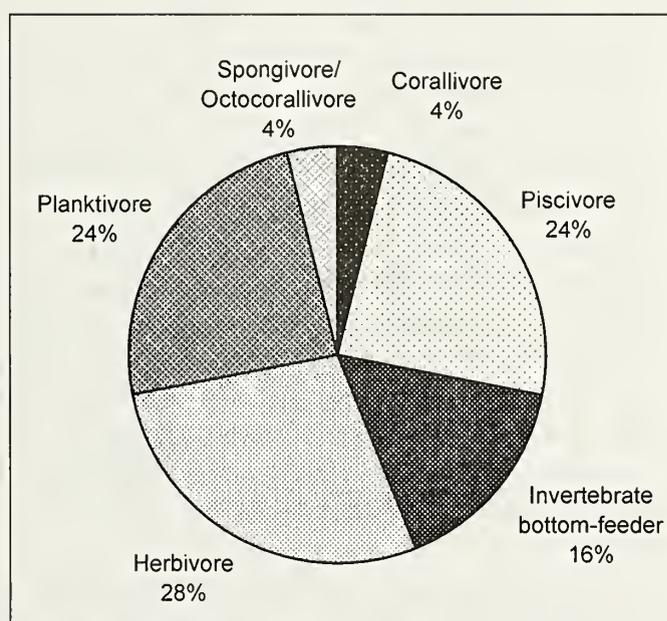
Benthic damselfish were detected in quantities of  $\approx 1.5$  individuals per 10 m transect (Table 2) and were represented primarily by the yellowtail damselfish, *Microspathodon chrysurus*. The threespot damselfish, *Stegastes partitus*, was rare as were its algal gardens in the surveyed corals.

### Fish Diversity

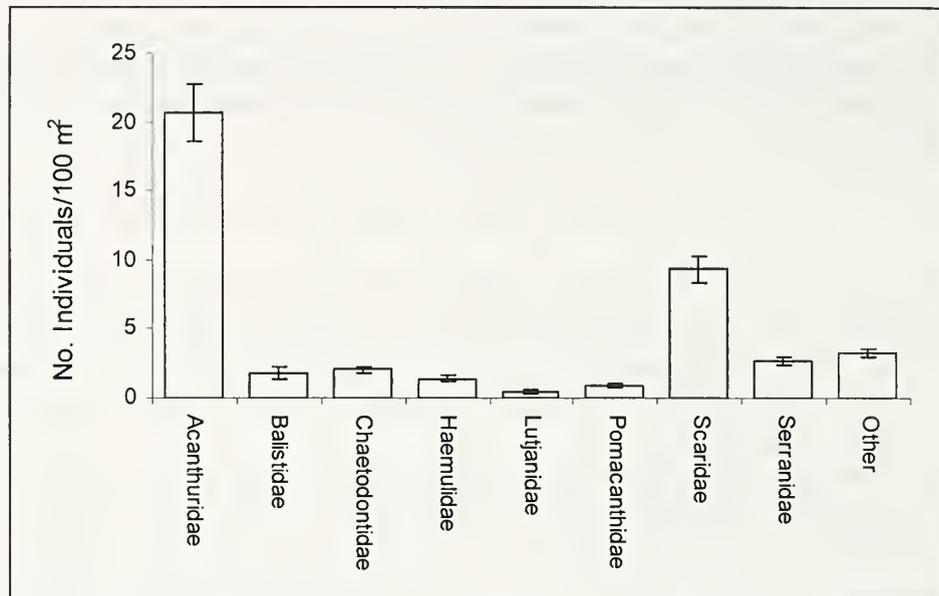
A total of 142 species of fish were recorded during 25.5 hours of roving diver surveys conducted at the 24 sites of the windward N. A. The fish assemblages were represented by six major trophic guilds with herbivores being dominant (28% of species sighted) and species from planktivorous and piscivorous guilds each representing 24% of the sightings (Table 4; Fig. 10).

Sightings of serranids included *Epinephelus fulvus* (coney, 100% of dives), *E. cruentatus* (graysby, 86% of dives), *E. guttatus* (red hind, 52% of dives), *Mycteroperca venenosa* (yellowfin grouper, 24% of dives), *M. tigris* (tiger grouper, 21% of dives), *E. striatus* (Nassau grouper, 7% of dives), *M. interstitialis* (yellowmouth grouper, 3% of dives), and *E. morio* (red grouper, 3% of dives).

Although no fish kills were noted, many of the fish, especially in St. Maarten and Saba, bore bruises and lacerations presumably resulting from Hurricane Lenny. These were particularly apparent on the pale, smooth skin of acanthurids but also noted on many scarids. Additionally, it appeared that a greater number of fish than normal were spending time at cleaning stations, seemingly having their injuries tended.



**Figure 10.** Trophic guild composition of the 25 most commonly sighted fish species during roving diver surveys in the windward Netherlands Antilles.

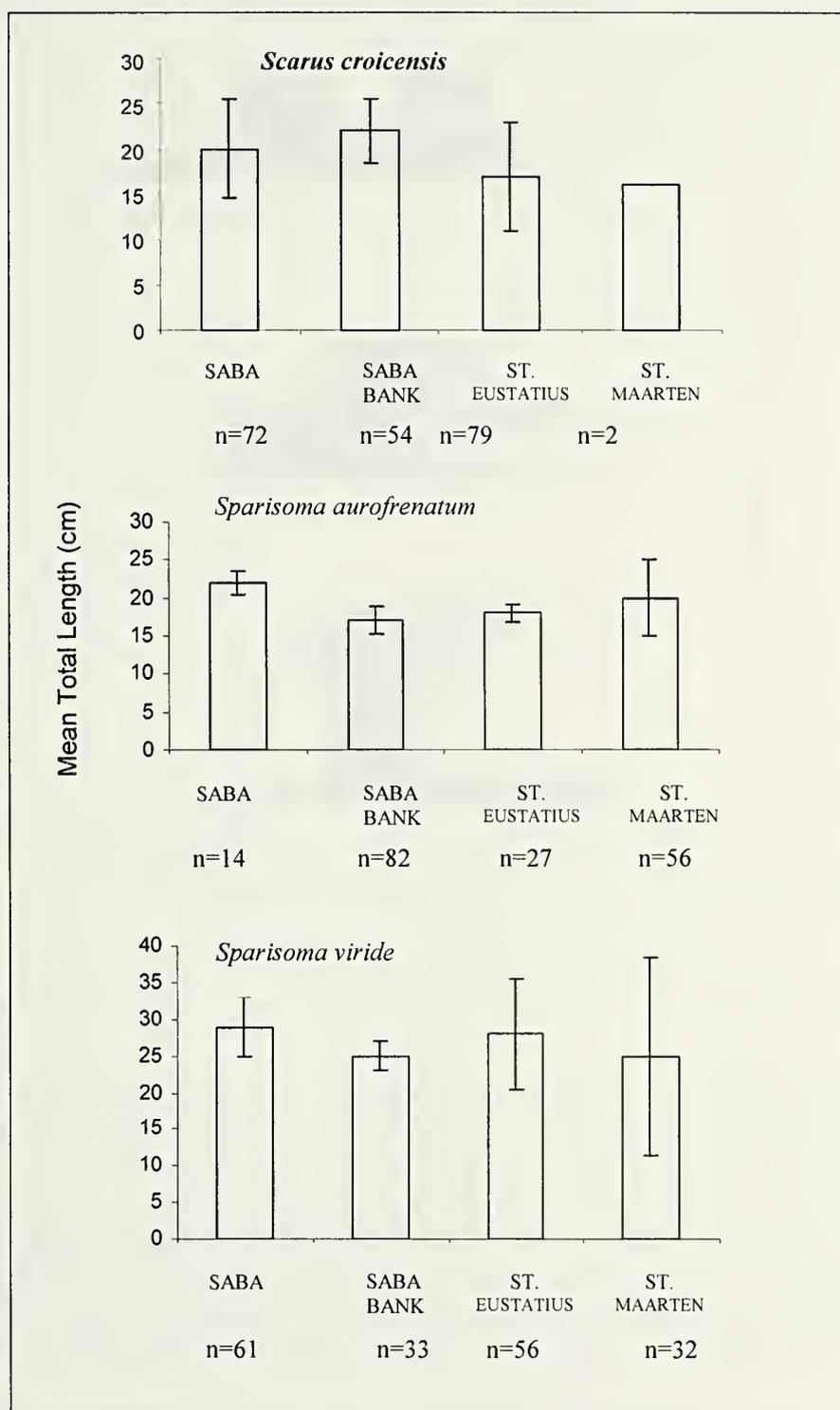


**Figure 11.** Mean fish density (no. individuals/100m<sup>2</sup> ± se) for AGRRA fishes in the windward Netherlands Antilles. Other = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Microspathodon chrysurus* and *Sphyrna barracuda*.

### Ecologically and Commercially Significant Fishes

A total of 6,116 individual fish were counted in 240 belt transects. Acanthurids (surgeonfish) and scarids dominated the selected species surveyed in the belt transects (Fig. 11). The most common species were *Acanthurus bahianus*, *A. coeruleus*, *Sparisoma aurofrenatum*, *S. viride*, *Scarus croicensis* and *S. taeniopterus*. Although variable among sites and areas, acanthurids had the greatest densities (mean = 19 individuals/100 m<sup>2</sup>) and highest biomass (mean = 3,150 grams/100 m<sup>2</sup>) overall (Table 5). Total lengths for four common species of parrotfish (counts restricted to ≥5 cm) averaged 19 cm for *Scarus croicensis*, 23 cm for *S. taeniopterus*, 20 cm for *Sparisoma aurofrenatum* and 28 cm for *S. viride*. No significant differences were detected in mean lengths among the four geographical areas for *S. croicensis* (ANOVA,  $p = 0.20$ ) and for *S. viride* (ANOVA,  $p = 0.51$ ). However, *S. aurofrenatum* were significantly larger in Saba (ANOVA,  $p = 0.01$ ) than on the Saba Bank (Tukey,  $p = 0.05$ ) and in St. Eustatius (Tukey,  $p = 0.01$ ), but overlapped with those in St. Maarten where mean lengths were highly variable (Fig. 12). Statistical comparisons for *S. taeniopterus* were not computed since this species was abundant only in St. Eustatius.

The density of herbivorous fishes (i.e., acanthurids, scarids ≥5 cm, *M. chrysurus*) averaged 30.8 individuals/100 m<sup>2</sup>. Acanthurids were predominant with an overall mean density of 19.3 individuals/100 m<sup>2</sup> followed by scarids and *M. chrysurus* with respective means of 9.3 and 2.2 individuals/100 m<sup>2</sup>. This pattern of dominance was consistent throughout the four geographical areas surveyed with the exception of Saba Bank where densities of acanthurids and scarids were nearly equal. However, mean density of the



**Figure 12.** Mean total length ( $\pm$  se) for three common scarids (all  $\geq 5$  cm) for each geographical area in the windward Netherlands Antilles.

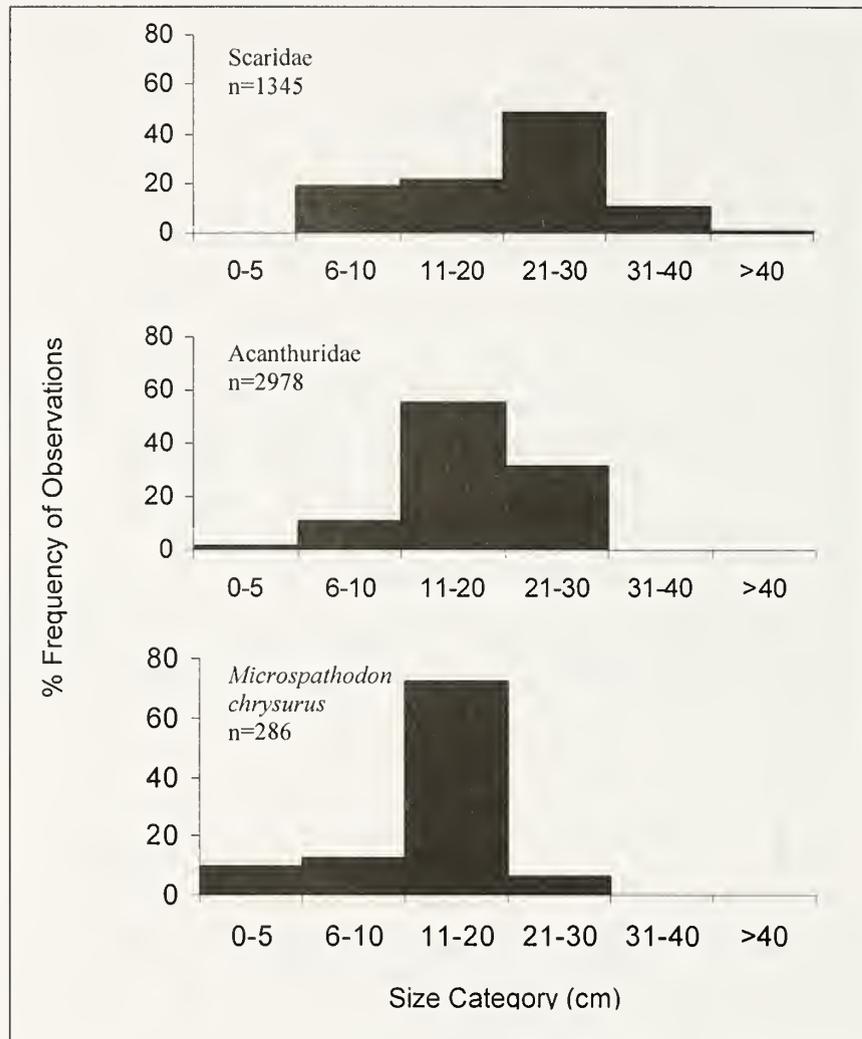
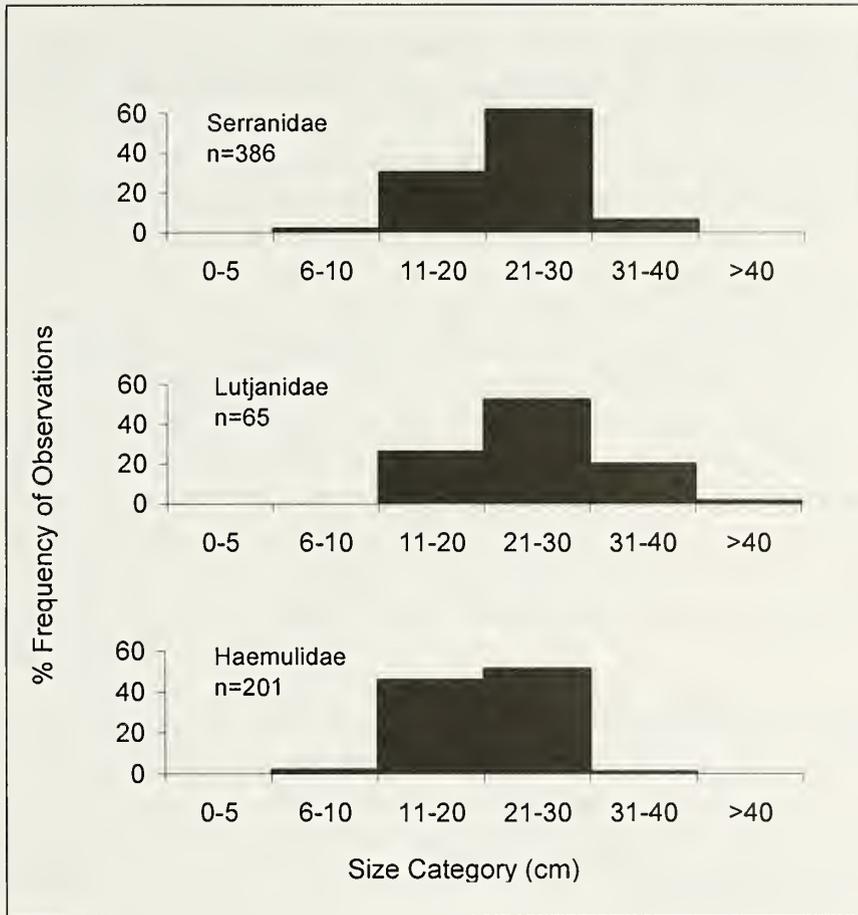


Figure 13. Size frequency distribution of dominant herbivores in the windward Netherlands Antilles.

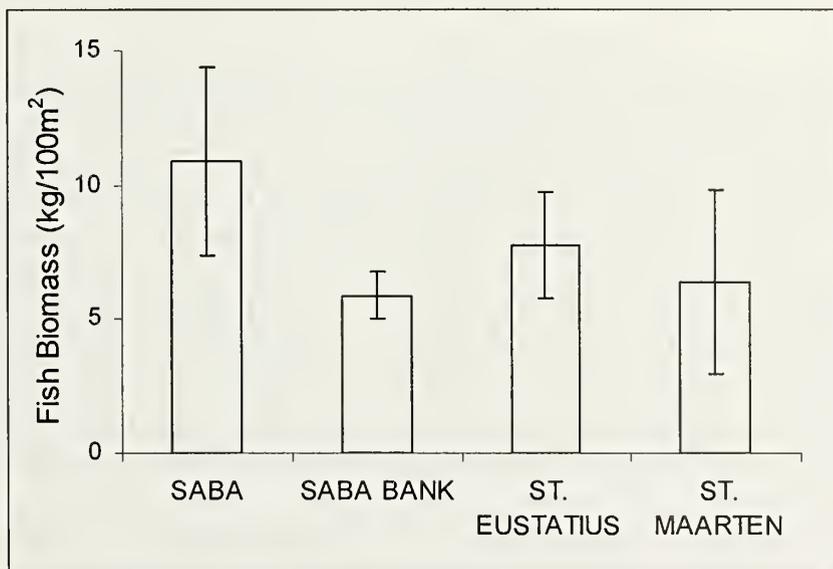
Acanthuridae was notably greater in Saba than in the other three areas (Table 5). Most of the herbivorous fish we recorded were between 11 and 30 cm in length (Fig. 13).

The mean density of fish species of high commercial value (i.e., large serranids, lutjanids, haemulids  $\geq 5$  cm) was 4.5 individuals/100 m<sup>2</sup>. The Serranidae, of which the numerical dominant was *Epinephelus fulvus* (with 85% of individuals), was the predominant family with an overall mean of 2.4 individuals/100 m<sup>2</sup> (Table 5). The most common size class for individual serranids and lutjanids in the windward N. A. was 21-30 cm whereas haemulids were evenly distributed between 11-20 cm and 21-30 cm (Fig. 14).

Total community biomass for the eight fish families included in the AGRRA belt transects averaged 16 kg/100 m<sup>2</sup> ( $\pm 5.1$  se). Saba (Fig. 15) had a significantly greater biomass (mean = 11 kg/100 m<sup>2</sup>  $\pm 3.5$  se;  $p=0.0003$ ) than the other geographical areas (combined mean = 7 kg/100 m<sup>2</sup>  $\pm 2.0$  se).



**Figure 14.** Size frequency distribution of commercially valuable carnivores in the windward Netherlands Antilles.



**Figure 15.** Total mean biomass (kg/100 m<sup>2</sup> ± se) for the AGRRA fishes in eight families in the windward Netherlands Antilles.

A linear regression revealed a significant inverse relationship ( $p = 0.03$ ;  $r^2 = 0.19$ ) between herbivorous fish biomass and macroalgal index at the windward N. A. survey sites (Fig. 16).

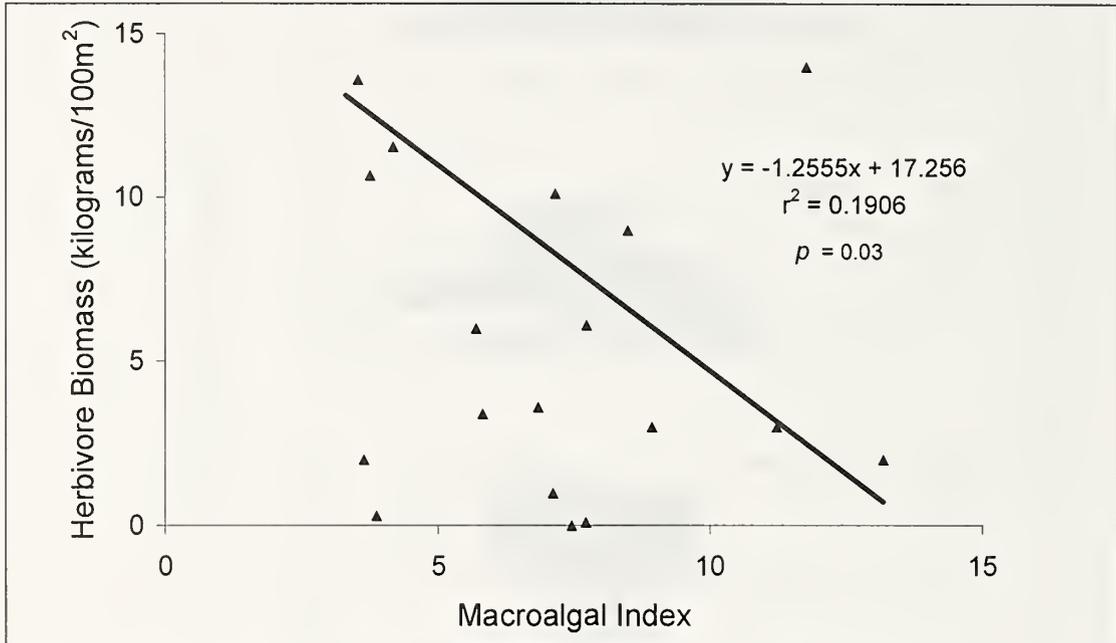


Figure 16. Regression plot between mean herbivore biomass (kg/100m<sup>2</sup>) and mean macroalgal index, by site in the windward Netherlands Antilles.

## DISCUSSION

The condition of reefs in the windward N. A. is largely influenced by their position within the hurricane belt of the Atlantic Ocean. The vulnerability of this region to the impacts of storm forces is evidenced by the disappearance of mature stands of *Acropora palmata* in recent years (Acroporids are particularly susceptible to damaging storm forces due to their fragile, branching nature.). Before Hurricane Luis (1995) *A. palmata* dominated sites SAB1 and SAB2 on Saba's windswept east coast, along with sites SAB8 and SAB9 on the north coast which also frequently experience high waves and strong surges (ENCAMP, 1980a). Although some recovery had been observed since Luis (Kooistra, personal observations), Hurricanes Georges (1998) and Lenny (1999) caused still more damage. We were unable to detect most of the colonies of *A. palmata* which died during earlier storms as the remaining skeletons have been eroded beyond taxon recognition.

The overall small size of the individually surveyed corals at most sites in the windward N. A. (Table 2; Fig. 6) reflects the high abundance of small colonies of *Diploria* spp., *Porites astreoides* and *Agaricia agaricites*. The flattened growth patterns of *Diploria* spp. and *P. astreoides* on many Saban reefs may reflect an adaptation to high surge and frequent hurricanes.

Hurricanes normally develop over the Atlantic and approach the windward N. A. islands from the east. Hurricane Lenny was unusual in that it developed in the southern Caribbean and advanced from west to east. While situated between St. Maarten and Saba, the hurricane stalled in position for nearly 22 hours which may explain why the greatest physical damage to stony corals was experienced in the Saban sites with western exposure (SAB4, SAB8, SAB9).

During our dives, visibility was somewhat restricted (ranging from 4–23 meters) due to increased turbidity from suspended sediments following Hurricane Lenny's passage. The high levels of partial bleaching that we recorded (nearly 25% of the individually surveyed colonies) was likely a response to conditions of reduced light due to the increased turbidity. Bleaching conditions may have been compounded by circumstances (i.e., suspended sediment and turbidity) introduced by Hurricane José in October 1999, although we have no evidence to support this claim. Although some settling of sediment had occurred at the time of our assessment, the full extent of the hurricane's impact on the reefs of the windward N. A. might not be recognizable since delayed mortality due to sedimentation will be difficult to attribute to this storm with any degree of certainty.

Additional damage was noted in large (up to 1 m high) barrel sponges. Some of the *Xestospongia muta*, which were previously abundant on the reefs of Saba and St. Eustatius, disappeared during Hurricane Lenny while others were severely damaged. Regrowth rates of approximately 2.5 cm per month were initially recorded in the surviving sponges (Kooistra, personal observation) although recovery rates slowed after several months.

It remains unclear why *C. natans* displayed a higher incidence of partial mortality (particularly old mortality) than these other species. Incidence of disease in *C. natans* and in *Diploria strigosa* (for which recent partial mortality was also relatively elevated) were no higher than in other taxa. Other potentially aggravating agents such as *Corallophilia abbreviata*, *Palythoa* spp., and burrowing serpulids were noted but *C. natans* and *D. strigosa* were not disproportionately plagued by these factors.

Partial mortality in the  $\geq 10$  cm corals was low throughout the windward N. A. in late 1999 (Fig. 8). It should be recognized, however, that all of our sites were situated in deeper water (by AGRRA standards) where mortality is normally less prominent. In all likelihood, mortality estimates would have been greater had we been able to document the demise of the *Acropora* that once existed in these waters (see above).

The *Montastraea annularis* complex consists of slow-growing massive corals and, like the acroporids, has been an important primary reef-builder in the Caribbean. If future climate conditions resemble contemporary patterns, it is unlikely that the reefs of the windward N. A. will sustain substantial populations of *Acropora*. Hence, the importance of the massive *Montastraea annularis* complex as a reef-building component is reinforced as it is more resilient to destructive storm forces. *Montastraea* has been a persistent genus on reefs over geologic time (Birkeland, 1997). Westerman and Kiel (1961) reported "plate-like" colonies of fossilized *M. annularis* embedded in St. Eustatius' coastal cliffs (Sugar Loaf) near our sites EUX09 and EUX10. Its presence in this region 20,000 to 70,000 years ago, and as mature living colonies today, suggests that local environmental conditions have been suitable and recruitment successful for this species throughout glacial sea-level oscillations and disturbances during the Recent. The

potential exists for reef development throughout this area if environmental conditions remain favorable and if *M. annularis* continues to proliferate and withstand disturbances.

Notwithstanding its dominance throughout the windward N. A. (the *Montastraea annularis* complex collectively constituted 30% of the individually surveyed corals), none of the various morphs had been a major contributor (< 1%) to coral recruitment (Fig. 5) in late 1999. Our findings are consistent, however, with previous studies of coral recruitment within the Caribbean. Rylaarsdam (1983) reported that *Montastraea annularis* composed  $\leq 1\%$  of juveniles recorded at Discovery Bay, Jamaica where its colonies made up 7-32% of the coral cover. Hughes (1985) noted complete failure to detect *M. annularis* recruitment on reefs at Rio Bueno, Jamaica where larger colonies were locally abundant. Similarly, a paucity of juveniles of reef-building taxa from Curacao and Bonaire led Bak and Engel (1979) to conclude that the composition of the adult coral community is not a direct function of juvenile abundance. The low rate of recruitment of *M. annularis* on Caribbean reefs has been explained as a characteristic of its life history strategy. This is in contrast to species which commonly dominate the juvenile community and that display relatively high rates of recruitment followed by high mortality (eg., *Agaricia agaricites* and *Porites astreoides*). *M. annularis* has a relatively high survival rate aided by its moderate ability to clear sediment from its surface, its aggressive nature toward other species, and its ability to regenerate lesions (Bak and Engel, 1979; Wittenberg and Hunte, 1992). However, since it is unknown what threshold level of parental spawning stock is necessary to maintain coral populations at sustainable levels, a decrease in adult spawners, as evidenced by a decline in live coral cover, may indicate a reduced capacity to replenish populations. Long-term monitoring studies in the U.S. Virgin Islands where *M. annularis* is the dominant coral indicated that following a 35% decline in its cover, there had been no substantial recovery (Smith et al., 1997). Subsequent recruitment studies in this area revealed that broadcast spawning species (including the *M. annularis* complex and *Acropora* spp.) composed less than 4% of coral recruits (Kojis and Quinn, 2001). Even if relatively low densities of juvenile *M. annularis* are a natural phenomenon, these levels may not be adequate to offset reduced reproduction associated with high rates of decline in live coral cover. Hence, the importance of *M. annularis* as a major component to reef structure in the N. A. may warrant monitoring to detect future trends.

The rather high relative abundance of crustose coralline algae (Table 3) is suggestive of conditions that should be conducive for the settlement and recruitment of coral larvae. Indeed, while recruitment was highly variable, corals less than 2 cm in diameter were detected at all sites. A low standing stock of macroalgae and low macroalgal indices (Table 3) may indicate a seasonal response to reduced light and water temperatures (Littler et al., 1986). Alternatively, it may imply that algae are routinely nutrient-limited or that algal biomass is kept in check by grazing herbivores. There are no data to support nutrient limitation or enrichment and the grazing echinoid, *D. antillarum*, was not present in our transects. Herbivorous fishes, especially acanthurids, were present in high densities at most sites (Table 5), however, and macroalgal biomass was inversely related to the biomass of herbivorous fishes in late 1999 (Fig. 16). Unfortunately, we have no quantitative data by which to compare algal conditions prior to Hurricane Lenny with those at the time of our assessment. Fleishy macroalgae (e.g., *Dictyota*) were abundant at two sites in St. Eustatius (EUX03, EUX05) during reconnaissance dives in

October 1999 but were not conspicuous elsewhere at that time. Considering that large barrel sponges were displaced by Hurricane Lenny, it is reasonable to believe that many of the macroalgae would also have been dislodged by this storm.

The fish assemblage of the windward N. A. is fairly rich in species and well represented by trophic guilds (Fig. 10). A relatively high density of herbivorous acanthurids and scarids was present at the time of our assessment (Table 5; Fig. 11). That we found greater mean total lengths for *Sparisoma aurofrenatum* in Saba (and possibly St. Maarten) than at Saba Bank and St. Eustatius (Fig. 12) may be an indication of better growth conditions at the former or it could be related to variations in age distribution among the geographic areas. Since these species are not targeted by the fishing community, small deviations in size are not of great concern but may be a point of interest to the dive community, or for future growth or size-related studies.

A number of studies have demonstrated that macroalgae appear to be common only where they are not heavily grazed (Lewis, 1986; Morrison, 1988; Schmitt, 1997). The inverse relationship we detected between herbivorous fish biomass and macroalgal biomass (Fig. 16) may suggest that herbivorous fish biomass increased as a result of effective grazing although this relationship only explains 19% of the variation in herbivore biomass. Even though most reef fish are generally considered “nonmigratory,” foraging groups of scarids and acanthurids are known to cover distances of hundreds of meters to several kilometers from their “home” reef (Reeson, 1983; Risk, 1998; Williams, 1991). Since fish biomass is a consequence of preceding food conditions and may have been attained at a time and locale where algal conditions varied from what we sampled in 1999, caution must be exercised in drawing conclusions based on a “snapshot” algal/fish biomass relationship.

It is noteworthy to recognize that most of the serranids encountered in our belt transects were *Epinepheus fulvus* (coney), a smaller (reaching  $\approx 30$  cm) less-targeted species. Since many of the larger serranids are wary of divers, it is possible that the belt-transect method did not adequately quantify their abundance. We did not detect many individuals of commercial value less than 10 cm (Fig. 14) which is not unexpected since our assessment took place in relatively deep water and it is recognized that juvenile serranids, lutjanids, and haemulids occupy shallow-water habitats (Thompson and Munro, 1983a; Thompson and Munro, 1983b; Gaut and Munro, 1983). Although lutjanids and serranids are known to feed on a wide range of prey, they are primarily piscivorous (Parrish, 1987). Their relatively low density suggests the possibility of reduced predation pressure on the fish community and may account for the fairly high densities of acanthurids and scarids.

The total biomass for the eight families (Fig. 15) provides an assay of fish community production in the windward N. A. Sites within the N. A. were relatively productive in fish biomass. A comparison of fish biomass from the same eight families from 60 sites along Jamaica's north coast (Klomp et al. in press), where decades of intense fishing activity has reduced fish abundance and size, indicates that fish biomass is over seven times greater in the windward islands of the N. A. Although not conclusive from this study, the greater biomass in Saba in part may be a result of reduced fishing pressure associated with the long-term protection within the Saba Marine Park since 1987.

The trophic guild classification (Fig. 10) illustrates the functional structure of the fish community as it existed in late 1999. It is important to remember that the forces responsible for shaping this community are a result of past events (e.g., larval recruitment and settlement, predation, environmental conditions) and cannot be identified from a single survey. This assessment does serve, however, as a baseline against which future data can be compared and, collectively with other AGRRA assessments, a point from which current trends in community structure can be detected (see Kramer, this volume).

Being able to detect between-site differences in live coral cover within the four geographical areas allows us to partition sites according to their potential for reef development. For example, *Acropora* has been virtually eliminated from Saban sites SAB01, SAB08, and SAB09 by recent hurricanes (see above). SAB03 is downstream from the only major coastal development on Saba (a man-made harbor and a recently closed rock-crushing facility). Increased sedimentation may be a limiting factor for coral development at SAB03 (Buchan, 1998). The remaining Saban sites offer the greatest potential for reef development and stand to benefit the most by maintaining marine-park protection. Indeed, while the coral assemblages along the west coast are not generally recognized as "true reefs" (Bak, 1977; Nagelkerken, 1981), they have accreted a contiguous, biogenic substratum (e.g., at SAB04, SAB05, and SAB06) and could, arguably, be classified as coral reefs (van't Hof, 1991). Not surprisingly, the *M. annularis* complex is the dominant coral (41% of all colonies) at these three sites. Our estimates for dominant coral cover (i.e., *M. annularis* complex = 46% of all colonies) and percent coral cover (mean = 23%) at SAB04 and SAB05, agree with estimates reported by Buchan (1998) for the CARICOMP coral reef site which lies between these two AGRRA sites (CARICOMP: *M. annularis* complex = 43% of all colonies; live coral cover = 24%).

Having an area greater than 2,200 km<sup>2</sup>, we did not sample a large enough sector on the Saba Bank to make definitive statements about the condition of its reefs. However, on the basis of our reconnaissance dives in October 1999 and previous reports (Hoetjes et al., 1999; Macintyre et al., 1975; Van Der Land, 1977), it is clear that live coral cover in some sectors exceeds that recorded during our December 1999 assessment and elsewhere approximates our lower estimate ( $\approx 11\%$ ). An extensive investigation, with the sole purpose of inventorying Saba Bank, would greatly increase our ability to speculate on the potential for reef development in the absence of coastal processes.

EUX09 and EUX10 had higher than average values for live coral cover (46%), and were dominated by colonies of *Montastraea annularis faveolata* having larger than average sizes. They may also have a positional advantage being the most southerly of our survey sites in St. Eustatius. Since prevailing coastal currents approach this area from the southeast (ENCAMP, 1980b), stony coral growth at EUX9 and EUX10 may be enhanced by currents introducing relatively nutrient-rich water and/or by receiving less coastal run-off.

The two sites surveyed on St. Maarten are routinely exposed to strong currents (SXM01) and high-energy waves (SXM02), their mean live coral cover was low (11%), and they are probably the most threatened reefs we encountered in the windward N. A. Even were hurricanes not a certain hazard, it will be a challenge to afford them enough protection to diffuse the impacts of St. Maarten's burgeoning coastal development and tourism.

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Table 1. Site information for AGRRA stony coral, algal and fish surveys in the windward Netherlands Antilles.

Site name	Site code	Reef type	Latitude (° N)	Longitude (° W)	Survey date	Depth (m)	Benthic transects (#)	≥25 cm stony corals (#/10 m)	≥10 cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)	30 m fish transects (#)	RDT fish species (#)
<b>Saba</b>												
Core Gut	SAB01	encrusted boulders; high-surge embayment	17 37.85	63 12.90	Dec 3 99	9	13	4	8	14 ± 6	10	58
Hole in the Corner	SAB02	encrusted boulders; gentle slope	17 36.90	63 13.50	Dec 3 99	11	13	6	12	22 ± 8	10	61
Tent Reef	SAB03	encrusted ledge w/ escarpment to sloping flat	17 36.98	63 15.51	Dec 6 99	10	15	3	8	9 ± 4	10	70
Hot Springs	SAB04	encrusted boulders and lava formations	17 37.49	63 15.57	Dec 7 99	11	12	7	11	22 ± 8	10	60
Ladder-Labyrinth	SAB05	encrusted rock buttresses & sloping lava flows	17 37.60	63 15.58	Dec 6 99	12	11	6	11	25 ± 12	10	53
Babylon	SAB06	encrusted ridges, sloping lava flows & sand channels	17 37.69	63 15.57	Dec 6 99	13	11	7	13	22 ± 4	10	54
Porites Point	SAB07	encrusted lava & boulders	17 37.76	63 15.53	Dec 9 99	15	13	6	11	19 ± 8	10	67
Torrrens Point	SAB08	exposed, encrusted boulders & high-relief buttresses	17 38.60	63 15.19	Dec 9 99	6	14	5	10	15 ± 6	10	40
Green Island	SAB09	encrusted boulder & rocky outcrops extend from islet	17 38.98	63 13.68	Dec 9 99	7	18	4	6	7 ± 4	10	46
<b>All Saba</b>										<b>17 ± 6</b>		<b>113</b>
<b>Saba Bank</b>												
BANK 1	BNK01	submerged atoll/bank rim	17 27.78	63 13.36	Dec 7 99	20	9	9	12	41 ± 12	10	54
BANK 2	BNK02	submerged atoll/bank rim	17 29.21	63 13.90	Dec 8 99	15	12	4	10	11 ± 4	10	47
BANK 3	BNK03	submerged atoll/bank rim	17 29.27	63 13.94	Dec 8 99	17	8	7	13	25 ± 12	10	54
<b>All Saba Bank</b>										<b>26 ± 15</b>		<b>81</b>
<b>St. Eustatius</b>												
Barracuda Reef	EUX01	low profile reef flat bordering sand flat	17 28.08	62 59.50	Dec 14 99	17	8	6	15	16 ± 6	10	52
Anchor Point	EUX02	low profile reef flat	17 27.86	62 59.27	Dec 13 99	14	8	6	15	18 ± 4	10	52
Hangover	EUX03	ridge abutment w. lava flow terminations	17 27.89	62 59.16	Dec 11 99	14	10	5	12	16 ± 6	10	60
The Blocks	EUX04	encrusted ledge; parallel to shoreline	17 27.86	62 59.10	Dec 12 99	15	9	7	13	17 ± 7	10	59
The Ledges	EUX05	encrusted ledge; parallel to shoreline	17 27.80	62 59.08	Dec 11 99	14	10	6	13	17 ± 6	10	66
Five Fingers	EUX06	sloping lava flows separated by sand channels	17 27.89	62 58.00	Dec 13 99	15	10	5	13	16 ± 12	10	64
Valley of the Sponges	EUX07	sloping lava flows separated by sand channels	17 27.84	62 58.95	Dec 12 99	14	11	4	10	12 ± 4	10	45
The Humps	EUX08	boulder corals on rocky outcrops	17 27.81	62 58.69	Dec 13 99	12	11	6	15	16 ± 6	10	61
Mushroom Gardens	EUX09	fringing reef. <i>M. annularis faveolata</i> dominant	17 27.75	62 58.66	Dec 11 99	16	8	10	14	47 ± 12	10	64
Mushroom South	EUX10	fringing reef. <i>M. annularis faveolata</i> dominant	17 27.70	62 58.66	Dec 14 99	16	9	9	14	45 ± 14	10	58
<b>All St. Eustatius</b>										<b>22 ± 13</b>		<b>106</b>
<b>St. Maarten</b>												
Mikes Maze	SXM01	reef flat; midshelf	17 59.74	63 03.60	Nov 30 99	9	20	3	5	10 ± 4	10	77
Hen & Chicks	SXM02	fringing reef off rocky islets	18 00.64	63 00.48	Dec 1 99	11	20	4	6	13 ± 8	10	69
<b>All St. Maarten</b>										<b>11 ± 2</b>		<b>92</b>

RDT = Roving Diver Technique

Table 2. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 10$  cm diameter) and mean damselfish density by site in the windward Netherlands Antilles.

Site name	Stony corals		Partial-colony mortality (%)			Standing dead	Stony corals (%)		Hurricane damaged	Damselfish <sup>1</sup> (#/10 m)	
	#	Diameter (cm)	Height (cm)	Recent	Old		Total	Bleached			Diseased
<b>Saba</b>											
Core Gut	106	41 $\pm$ 42	17 $\pm$ 24	2.0 $\pm$ 9.5	14 $\pm$ 22	16 $\pm$ 24	0.9	26	1.0	2.0	3.0
Hole in the Corner	151	38 $\pm$ 36	19 $\pm$ 24	<0.5 $\pm$ 1.0	11 $\pm$ 20	12 $\pm$ 20	0.0	30	0.5	0.5	1.5
Tent Reef	115	26 $\pm$ 16	14 $\pm$ 14	2.0 $\pm$ 10.0	16 $\pm$ 22	18 $\pm$ 24	0.9	20	1.0	2.5	0
Hot Springs	132	48 $\pm$ 43	32 $\pm$ 29	2.0 $\pm$ 7.5	18 $\pm$ 26	20 $\pm$ 26	0.0	27	1.0	4.5	5.5
Ladder Labyrinth	120	39 $\pm$ 40	22 $\pm$ 22	1.5 $\pm$ 9.0	17 $\pm$ 24	19 $\pm$ 27	0.8	18	0.0	0.0	0
Babylon	145	31 $\pm$ 25	17 $\pm$ 14	0.5 $\pm$ 2.0	15 $\pm$ 22	15 $\pm$ 23	0.0	28	1.5	2.0	.5
Porites Point	137	40 $\pm$ 44	22 $\pm$ 22	0.5 $\pm$ 2.0	15 $\pm$ 24	16 $\pm$ 24	0.7	25	0.5	0.5	4.5
Torrens Point	136	32 $\pm$ 30	16 $\pm$ 14	1.5 $\pm$ 6.5	18 $\pm$ 27	19 $\pm$ 28	0.0	21	0.0	6.5	0
Green Island	113	40 $\pm$ 47	23 $\pm$ 30	2.5 $\pm$ 13.5	25 $\pm$ 32	27 $\pm$ 33	7.1	3	2.5	4.5	0
<b>All Saba</b>	<b>1155</b>	<b>37 <math>\pm</math> 7</b>	<b>20 <math>\pm</math> 5</b>	<b>1.4 <math>\pm</math> 0.9</b>	<b>17 <math>\pm</math> 4</b>	<b>18 <math>\pm</math> 4</b>	<b>1.2</b>	<b>22</b>	<b>0.9</b>	<b>2.6</b>	<b>1.6 <math>\pm</math> 2.1</b>
<b>Saba Bank</b>											
BANK 1	105	73 $\pm$ 66	23 $\pm$ 24	<0.5 $\pm$ 1.5	15 $\pm$ 18	15 $\pm$ 18	0.0	6	1.0	0	4.0
BANK 2	114	25 $\pm$ 15	12 $\pm$ 10	1.0 $\pm$ 6.0	18 $\pm$ 25	19 $\pm$ 26	0.0	12	0	0	2.0
BANK 3	104	39 $\pm$ 37	26 $\pm$ 26	1.0 $\pm$ 6.5	18 $\pm$ 23	16 $\pm$ 21	1.0	9	0	0	0
<b>All Saba Bank</b>	<b>323</b>	<b>46 <math>\pm</math> 25</b>	<b>20 <math>\pm</math> 7</b>	<b>0.7 <math>\pm</math> 0.4</b>	<b>17 <math>\pm</math> 2</b>	<b>17 <math>\pm</math> 2</b>	<b>0.3</b>	<b>9</b>	<b>0.3</b>	<b>0</b>	<b>1.9 <math>\pm</math> 1.9</b>
<b>St. Eustatius</b>											
Barracuda Reef	119	24 $\pm$ 16	12 $\pm$ 8	0.5 $\pm$ 5.5	17 $\pm$ 24	18 $\pm$ 24	0.0	17	1	0	1.0
Anchor Point	116	23 $\pm$ 12	13 $\pm$ 12	0.5 $\pm$ 4.0	18 $\pm$ 26	19 $\pm$ 28	0.9	12	0	0	0
Hangover	121	27 $\pm$ 19	14 $\pm$ 12	2.5 $\pm$ 10.5	14 $\pm$ 21	17 $\pm$ 24	0.8	26	1	0	0
The Blocks	113	28 $\pm$ 16	15 $\pm$ 13	1.5 $\pm$ 4.0	15 $\pm$ 24	16 $\pm$ 24	0.0	31	1	0	0
The Ledges	126	29 $\pm$ 31	14 $\pm$ 11	<0.5 $\pm$ 1.0	12 $\pm$ 22	12 $\pm$ 22	0.8	18	0	0	0
Five Fingers	125	29 $\pm$ 21	16 $\pm$ 16	1.5 $\pm$ 8.0	16 $\pm$ 24	17 $\pm$ 25	0.0	26	0	0	0
Valley of the Sponges	109	24 $\pm$ 14	15 $\pm$ 14	0.5 $\pm$ 2.0	22 $\pm$ 26	22 $\pm$ 26	0.0	28	0	1	0
The Humps	165	25 $\pm$ 15	14 $\pm$ 12	0.5 $\pm$ 2.0	15 $\pm$ 22	15 $\pm$ 22	0.0	22	0	0	0
Mushroom Gardens	113	80 $\pm$ 90	37 $\pm$ 38	0.5 $\pm$ 1.5	12 $\pm$ 18	12 $\pm$ 18	0.0	28	2.0	0	1.0
Mushroom South	123	73 $\pm$ 88	33 $\pm$ 38	1.0 $\pm$ 7.5	15 $\pm$ 22	16 $\pm$ 22	0.8	34	1.5	1	9.0
<b>All S. Eustatius</b>	<b>1230</b>	<b>36 <math>\pm</math> 21</b>	<b>18 <math>\pm</math> 9</b>	<b>0.9 <math>\pm</math> 0.7</b>	<b>16 <math>\pm</math> 3</b>	<b>17 <math>\pm</math> 3</b>	<b>0.3</b>	<b>24</b>	<b>0.6</b>	<b>0.2</b>	<b>1.1 <math>\pm</math> 2.8</b>
<b>St. Maarten</b>											
Mikes Maze	105	27 $\pm$ 16	12 $\pm$ 12	0.5 $\pm$ 1.5	8 $\pm$ 20	9 $\pm$ 20	1.9	45	0	0	0
Hen & Chicks	117	47 $\pm$ 55	19 $\pm$ 21	2.0 $\pm$ 1.5	18 $\pm$ 27	20 $\pm$ 30	1.7	43	1.0	1.0	1.5
<b>All St. Maarten</b>	<b>222</b>	<b>37 <math>\pm</math> 14</b>	<b>15 <math>\pm</math> 5</b>	<b>1.3 <math>\pm</math> 1.2</b>	<b>13 <math>\pm</math> 7</b>	<b>14 <math>\pm</math> 8</b>	<b>1.8</b>	<b>44</b>	<b>0.5</b>	<b>0.5</b>	<b>0.9 <math>\pm</math> 1.2</b>

<sup>1</sup>Primarily *Microspathodon chrysurus*.

Table 3. Algal characteristics, density of stony coral recruits and *Diadema antillarum* (mean  $\pm$  standard deviation) by site in the windward Netherlands Antilles.

Site name	Site code	Quadrats (#)	Relative abundance (%)			Macroalgal		Recruits (#/0.0625 m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )
			Macroalgae	Turf algae	Crustose coralline algae	Height (cm)	Index <sup>1</sup>		
<b>Saba</b>									
Core Gut	SAB01	60	13 $\pm$ 7	61 $\pm$ 30	26 $\pm$ 30	1.0 $\pm$ 0.5	14	.35	0
Hole in the Corner	SAB02	46	3 $\pm$ 5	49 $\pm$ 31	48 $\pm$ 30	2.0 $\pm$ 1.5	6	.13	0
Tent Reef	SAB03	68	1 $\pm$ 1	75 $\pm$ 30	25 $\pm$ 30	1.0 $\pm$ 0.0	0	.50	0
Hot Springs	SAB04	50	2 $\pm$ 5	57 $\pm$ 32	41 $\pm$ 33	1.0 $\pm$ 0.5	2	.23	0
Ladder Labyrinth	SAB05	50	7 $\pm$ 8	49 $\pm$ 25	44 $\pm$ 27	2.5 $\pm$ 5.5	18	.14	0
Babylon	SAB06	45	4 $\pm$ 7	49 $\pm$ 28	47 $\pm$ 29	1.0 $\pm$ 0.5	3	.18	0
Porites Point	SAB07	55	8 $\pm$ 7	49 $\pm$ 26	43 $\pm$ 27	1.0 $\pm$ 0.5	9	.11	0
Torrens Point	SAB08	55	2 $\pm$ 5	52 $\pm$ 30	46 $\pm$ 30	1.5 $\pm$ 0.5	3	.28	0
Green Island	SAB09	55	1 $\pm$ 2	67 $\pm$ 28	32 $\pm$ 28	1.0 $\pm$ 0.5	1	.16	0
All Saba		<b>484</b>	<b>5 <math>\pm</math> 4</b>	<b>56 <math>\pm</math> 9</b>	<b>39 <math>\pm</math> 9</b>	<b>1.4 <math>\pm</math> 0.6</b>	<b>6 <math>\pm</math> 6</b>	<b>.24 <math>\pm</math> .18</b>	<b>0</b>
<b>Saba Bank</b>									
BANK 1	BNK01	50	14 $\pm$ 15	30 $\pm$ 26	57 $\pm$ 32	1.0 $\pm$ 1.0	18	.18	0
BANK 2	BNK02	50	<1.0 $\pm$ 1	56 $\pm$ 31	44 $\pm$ 31	2.0 $\pm$ 1.0	1	.14	0
BANK 3	BNK03	40	<1.0 $\pm$ 1	34 $\pm$ 28	65 $\pm$ 29	1.0 $\pm$ 0.0	<1	.10	0
All Saba Bank		<b>140</b>	<b>5 <math>\pm</math> 8</b>	<b>40 <math>\pm</math> 14</b>	<b>55 <math>\pm</math> 11</b>	<b>1.4 <math>\pm</math> 0.5</b>	<b>6 <math>\pm</math> 10</b>	<b>.14 <math>\pm</math> .04</b>	<b>0</b>
<b>St. Eustatius</b>									
Barracuda Reef	EUX01	50	17 $\pm$ 18	44 $\pm$ 26	39 $\pm$ 24	1.0 $\pm$ 0.5	19	.34	0
Anchor Point	EUX02	50	15 $\pm$ 15	44 $\pm$ 25	41 $\pm$ 24	1.0 $\pm$ 0.5	18	.46	0
Hangover	EUX03	50	6 $\pm$ 8	35 $\pm$ 23	58 $\pm$ 22	1.0 $\pm$ 0.5	6	.22	0
The Blocks	EUX04	50	9 $\pm$ 10	47 $\pm$ 28	44 $\pm$ 27	1.0 $\pm$ 0.5	10	.68	0
The Ledges	EUX05	50	15 $\pm$ 13	34 $\pm$ 26	51 $\pm$ 29	1.0 $\pm$ 0.5	17	.34	0
Five Fingers	EUX06	50	9 $\pm$ 13	63 $\pm$ 30	28 $\pm$ 26	1.0 $\pm$ 0.5	11	.34	0
Valley of the Sponges	EUX07	50	3 $\pm$ 5	64 $\pm$ 29	32 $\pm$ 27	1.0 $\pm$ 0.0	3	11.2	0
The Humps	EUX08	50	4 $\pm$ 5	64 $\pm$ 24	33 $\pm$ 24	1.0 $\pm$ 0.0	4	.26	0
Mushroom Gardens	EUX09	45	11 $\pm$ 11	59 $\pm$ 31	30 $\pm$ 30	1.0 $\pm$ 0.5	12	.13	0
Mushroom South	EUX10	50	14 $\pm$ 15	63 $\pm$ 25	24 $\pm$ 23	1.0 $\pm$ 0.0	14	.18	0
All St. Eustatius		<b>495</b>	<b>10 <math>\pm</math> 5</b>	<b>52 <math>\pm</math> 12</b>	<b>38 <math>\pm</math> 11</b>	<b>1.1 <math>\pm</math> 0.1</b>	<b>11 <math>\pm</math> 6</b>	<b>.42 <math>\pm</math> .31</b>	<b>0</b>
<b>St. Maarten</b>									
Mikes Maze	SXM01	58	<1.0 $\pm$ 1	62 $\pm$ 28	38 $\pm$ 28	0.5 $\pm$ 0.0	<1	.29	0
Hen & Chicks	SXM02	67	15 $\pm$ 21	51 $\pm$ 31	34 $\pm$ 27	2.0 $\pm$ 1.0	33	.18	0
All St. Maarten		<b>125</b>	<b>8 <math>\pm</math> 10</b>	<b>57 <math>\pm</math> 8</b>	<b>36 <math>\pm</math> 3</b>	<b>1.4 <math>\pm</math> 1.2</b>	<b>16 <math>\pm</math> 23</b>	<b>.24 <math>\pm</math> .08</b>	<b>0</b>

<sup>1</sup>Macroalgal index = % relative abundance of macroalgae x macroalgal height

Table 4. Twenty-five most frequently sighted fish species during roving diver surveys in the windward Netherlands Antilles with mean densities for species in belt transects.

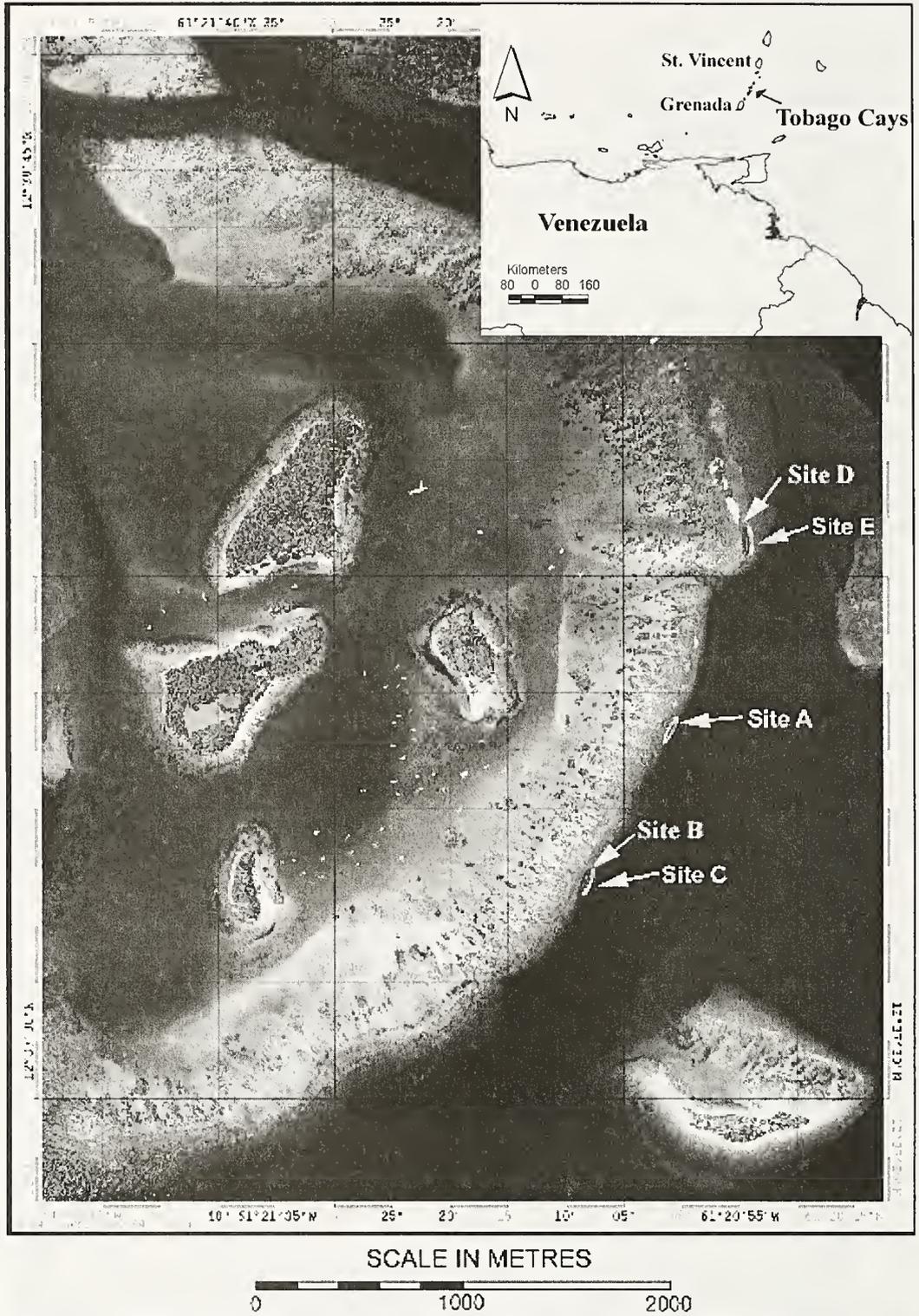
Scientific name	Common name	Sighting frequency (%) <sup>1</sup>	Density (# individuals/100 m <sup>2</sup> )				
			Saba	Saba Bank	St. Eustatius	St. Maarten	N. A. area
<i>Acanthurus coeruleus</i>	Blue Tang	100.0	12.8	2.4	7.6	7.3	8.9
<i>Thalassoma bifasciatum</i>	Bluehead Wrasse	100.0					
<i>Acanthurus bahianus</i>	Ocean Surgeonfish	100.0	14.5	7.5	9.4	14.5	11.5
<i>Epinephelus fulvus</i>	Coney	100.0	2.3	3.0	2.4	0.3	2.3
<i>Sparisoma viride</i>	Stoplight Parrotfish	100.0	1.1	1.8	1.3	2.7	1.4
<i>Microspathodon chrysurus</i>	Yellowtail Damselfish	100.0	2.4	1.1	1.7	3.0	2.0
<i>Holocentrus rufus</i>	Longspine Squirrelfish	100.0					
<i>Caranx ruber</i>	Bar Jack	100.0	0.6	0.4	0.6	0.1	0.5
<i>Stegastes partitus</i>	Bicolor Damselfish	96.5					
<i>Chromis cyanea</i>	Blue Chromis	96.5					
<i>Halichoeres garnoti</i>	Yellowhead Wrasse	96.5					
<i>Chaetodon capistratus</i>	Four-eye Butterflyfish	96.5	2.0	0.8	0.9	1.6	1.3
<i>Bodianus rufus</i>	Spanish Hogfish	96.5	1.1	0.2	0.6	0.3	0.7
<i>Mulloidichthys martinicus</i>	Yellow Goatfish	93.1					
<i>Melichthys niger</i>	Black Durgon	93.1	2.5	0.7	1.1	0.6	1.5
<i>Holacanthus tricolor</i>	Rock Beauty	93.1	0.4	1.0	0.8	0.3	0.7
<i>Serranus tigrinus</i>	Harlequin Bass	93.1					
<i>Aulostomus maculatus</i>	Trumpetfish	93.1					
<i>Chromis multilineata</i>	Brown Chromis	89.6					
<i>Sparisoma aurofrenatum</i>	Redband Parrotfish	89.6	2.7	4.6	4.7	4.7	3.9
<i>Scarus taeniopterus</i>	Princess Parrotfish	89.6	0.7	1.5	4.9	0.3	2.5
<i>Canthigaster rostrata</i>	Sharpnose Puffer	89.6					
<i>Epinephelus cruentatus</i>	Graysby	86.2	0.1	0.4	0.6	0.1	0.3
<i>Haemulon carbonarium</i>	Caesar Grunt	86.2	0.4	0.8	0.2	0.0	0.5
<i>Myripristis jacobus</i>	Blackbar Soldierfish	82.7					

<sup>1</sup>Sighting frequency for the AGRRA surveys from the REEF database at <http://www.reef.org/data/twa/surveys/index.shtml>

Table 5. Mean density and biomass of AGRRA fishes by site in the windward Netherlands Antilles.

Site code <sup>1</sup>	Commercially important carnivores											
	Acanthuridae			Herbivores			Microspathodon chrysurus			Serranidae <sup>2</sup>		
	Density (#/100 m <sup>2</sup> )	Biomass (g/100 m <sup>2</sup> )	Scaridae (>5 cm) Density (#/100 m <sup>2</sup> )	Biomass (g/100 m <sup>2</sup> )	Density (#/100 m <sup>2</sup> )	Biomass (g/100 m <sup>2</sup> )	Density (#/100 m <sup>2</sup> )	Biomass (g/100 m <sup>2</sup> )	Density (#/100 m <sup>2</sup> )	Biomass (g/100 m <sup>2</sup> )	Density (#/100 m <sup>2</sup> )	Biomass (g/100 m <sup>2</sup> )
<b>Saba</b>												
SAB01	30.0	9481	3.5	1332	3.5	973	0.0	0	1.0	378	2.0	475
SAB02	27.0	4105	2.5	921	4.5	674	<0.5	59	0.5	155	2.5	415
SAB03	20.0	3272	10.0	3346	1.5	839	2.0	678	0.5	95	4.5	936
SAB04	43.5	10146	7.5	2297	4.0	734	4.5	1138	0.0	130	2.0	505
SAB05	7.0	1154	6.5	2141	1.5	273	1.0	349	0.0	0	2.5	394
SAB06	26.0	5708	8.5	2843	1.5	379	0.5	132	2.0	1294	3.0	579
SAB07	15.5	4623	7.5	3328	0.5	132	1.0	132	1.5	502	2.0	668
SAB08	49.0	10438	1.0	261	3.5	527	1.0	213	0	0	1.0	215
SAB09	31.0	4777	6.0	1393	6.5	945	0.0	0	0	0	2.5	699
All Saba (mean ± standard error)	<b>27.7 ± 13.0</b>	<b>5967 ± 3967</b>	<b>5.9 ± 2.9</b>	<b>1985 ± 1083</b>	<b>3.0 ± 1.8</b>	<b>654 ± 244</b>	<b>1.2 ± 1.4</b>	<b>300 ± 379</b>	<b>0.6 ± 0.7</b>	<b>284 ± 417</b>	<b>2.4 ± 0.9</b>	<b>543 ± 208</b>
<b>Saba Bank</b>												
BNK01	7.5	1181	14.5	2169	1.5	287	<0.5	12	0.5	57	2.5	511
BNK02	13.0	1167	11.0	2299	<0.5	171	0.5	117	0.0	0	5.0	651
BNK03	10.0	1327	7.5	1732	1.5	812	3.0	783	2.0	321	3.0	639
All Saba Bank (mean ± standard error)	<b>10.1 ± 2.8</b>	<b>1225 ± 89</b>	<b>11.0 ± 3.3</b>	<b>2067 ± 297</b>	<b>1.1 ± 0.8</b>	<b>423 ± 341</b>	<b>1.2 ± 1.6</b>	<b>304 ± 418</b>	<b>0.8 ± 1.1</b>	<b>126 ± 171</b>	<b>3.6 ± 1.3</b>	<b>600 ± 78</b>
<b>St. Eustatius</b>												
EUX01	17.0	1663	10.0	1925	1.5	445	3.5	1019	0.5	143	1.5	480
EUX02	14.5	1548	9.5	2125	1.5	217	1.5	329	0	0	3.0	928
EUX03	18.5	2765	22.0	4463	3.0	511	1.0	200	0	0	2.5	763
EUX04	17.3	2793	16.5	3882	1.5	488	1.0	305	0	0	4.0	1577
EUX05	20.5	2466	16.0	4016	2.5	633	0.5	224	0	0	3.5	750
EUX06	14.5	1389	10.5	2089	1.0	276	3.5	466	0.5	85	3.5	868
EUX07	17.5	3341	9.5	2155	1.5	327	0.5	119	0	0	3.5	1160
EUX08	20.5	2823	13.5	3169	3.0	850	1.5	214	<0.5	45	4.0	1005
EUX09	10.0	1794	7.5	2010	1.0	375	1.0	106	0	0	3.0	767
EUX10	24.0	2188	7.0	1169	1.5	177	1.0	144	0.5	229	1.5	407
All St. Eustatius (mean ± standard error)	<b>17.4 ± 3.9</b>	<b>2277 ± 659</b>	<b>12.3 ± 4.7</b>	<b>2700 ± 1099</b>	<b>1.8 ± 0.6</b>	<b>430 ± 204</b>	<b>1.5 ± 1.2</b>	<b>313 ± 271</b>	<b>0.2 ± 0.3</b>	<b>50 ± 80</b>	<b>3.1 ± 0.8</b>	<b>871 ± 335</b>
<b>St. Maarten</b>												
SXM01	23.0	4237	7.0	3255	1.5	224	1.0	279	0.5	180	1.0	278
SXM02	20.5	2021	8.5	886	4.5	401	3.5	464	0.5	23	0	0
All St. Maarten (mean ± standard error)	<b>21.8 ± 1.8</b>	<b>3129 ± 1567</b>	<b>7.8 ± 1.2</b>	<b>2071 ± 1675</b>	<b>3.0 ± 2.4</b>	<b>313 ± 126</b>	<b>2.3 ± 1.5</b>	<b>371 ± 131</b>	<b>0.5 ± 0.2</b>	<b>101 ± 111</b>	<b>0.5 ± 0.7</b>	<b>139 ± 197</b>
<b>N.A. Region (mean ± standard error)</b>	<b>19.3 ± 7.4</b>	<b>3150 ± 2033</b>	<b>9.3 ± 2.9</b>	<b>2206 ± 332</b>	<b>2.2 ± 0.9</b>	<b>455 ± 143</b>	<b>1.6 ± 0.5</b>	<b>322 ± 33</b>	<b>0.5 ± 0.3</b>	<b>140 ± 101</b>	<b>2.4 ± 1.4</b>	<b>538 ± 302</b>

<sup>1</sup>See Table 1 for site names corresponding to site codes.<sup>2</sup>*Epinephetus* spp. and *Mycteroperca* spp.



**Figure 1.** Photo mosaic of the AGRRA survey sites on Horseshoe Reef, Tobago Cays, in St. Vincent and the Grenadines. See Table 1 for site codes.

**A RAPID ASSESSMENT OF THE HORSESHOE REEF,  
TOBAGO CAYS MARINE PARK, ST. VINCENT, WEST INDIES  
(STONY CORALS, ALGAE AND FISHES)**

BY

ALICE DESCHAMPS,<sup>1</sup> ANDRÉ DESROCHERS,<sup>1</sup> and KRISTI D. KLOMP<sup>2</sup>

**ABSTRACT**

Fore reefs at Horseshoe Reef, Tobago Cays, had an average live stony-coral cover of 30% at 3-4 m and nearly 40% at 9-11 m in June 1999. "Large corals" ( $\geq 25$  cm maximum diameter) were dominated by *Montastraea annularis*, *Porites astreoides* and *P. porites*. However, live colonies of *Acropora palmata*, which once flourished in the high-energy shallow-reef zones, had virtually disappeared. The maximum diameter of large reef corals averaged 58 cm which may be indicative of steady juvenile replenishment. The low values of recent partial-colony mortality ( $\leq 3\%$ ) and minor disease occurrences ( $\leq 6\%$  of colonies) at all sites indicated that the large corals on Horseshoe Reef had experienced no major recent disturbance events. Pale, or partially bleached, colonies on the deeper reefs (about 10% of large corals) were probably still recovering from the 1998 mass bleaching event. Algal communities in the shallower reefs were dominated by crustose coralline algae ( $>50\%$  relative abundance) whereas macroalgae (mainly *Halimeda* and *Dictyota*) were slightly more abundant than crustose corallines at 9-11 m. *Diadema antillarum* was uncommon in the deeper reefs but moderately abundant at 3-4 m. Eighty-one species of fish were recorded at Horseshoe Reef. The assemblage of censused fishes was dominated by herbivorous scarids (parrotfishes) and acanthurids (surgeonfishes). Herbivores, scarids in particular, also accounted for most of the censused fish biomass on the reef. Commercially valuable serranids, lutjanids and haemulids (groupers, snappers, grunts) were present in low densities ( $<1$  individual/100m<sup>2</sup>), indicative of overfishing.

**INTRODUCTION**

This study was located in the Tobago Cays, four small, uninhabited islands in St. Vincent-Grenadines (Fig.1) that are partially surrounded by fringing reefs. The Tobago

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Cays are protected on their windward sides by Horseshoe Reef, a semicircular, well-developed Holocene bank-barrier reef that is one of the longest ( $\approx 4$  km) in the southern Grenadine Islands (Dey and Smith, 1989). Seaward of its reef flat, the Horseshoe fore reef slopes steeply eastward to depths of about 20 meters.

The Grenadines are eroded remnant peaks and ridges of older volcanic islands that were partially or completely drowned during the post-Pleistocene rise in sea level. They arise from the submarine Grenadine Bank which runs NE-SW for about 180 km between St. Vincent and Grenada. The Bank is 15- to 20- km wide and 20-30 meters deep with local shallows at 3-6 m (Dey, 1985). The absence of rivers on these small islands prevents terrigenous sediments from being carried offshore and permits reef growth (Dey, 1985).

The Grenadines Bank lies within the Trade Wind belt where strong northeast winds develop during late autumn and winter and easterly winds during spring, summer and early autumn (Clack, 1977). The Equatorial Current moves over the Grenadine Bank, passes through the channels and reefs, and is diverted around the islands giving considerable local variation in current direction and strength (0.3-1.5 knots) (Dey and Smith, 1989). Tides in the Grenadines have a small range (0.6 m) and are mixed with a semidiurnal component dominating in the Tobago Cays area (Clack, 1977). Their strength, duration and general direction are influenced by local topographic variations present on the sea bottom of the shallow Bank. Temperature and salinity are both relatively constant over Grenadine Bank (Dey, 1985). The Bank is south of the region usually affected by hurricanes. The most violent recent storms to strike the Grenadine islands were Hurricanes Janet in 1955 and Allen in 1980 (Dey and Smith, 1989). No other hurricanes during the past 50 years have seriously affected the Tobago Cays' reefs (Kurt Cordice, personal communication).

The Tobago Cays are 3 km from Mayreau, the closest inhabited Grenadine Island with a population of approximately 250. Union Island, the closest major population center, with approximately 3500, is about 5 km away. The Cays are frequently visited by fishermen harvesting conch and lobster and are now increasingly used by tourists for sailing, snorkeling and diving.

Over the past 15 years a number of informal reports have indicated that reefs in the Tobago Cays have deteriorated as a result of physical damage from storms, anchors, and fishing gear, as well as from white-band disease, other diseases, and localized nutrient pollution from yachts (Wells, 1988; Smith et al., 1997). The Tobago Cays Marine Park (TCMP) protected area was established in 1998 and is gradually developing its management plans. Horseshoe Reef lies within the TCMP and some fishing regulations have been established. Conversations with local residents revealed, however, that illegal fishing with spearguns is a common practice in the local artisanal fishing community and that any sizable fish, including parrotfish, are targeted. Indeed, we witnessed fishing activity within the marine park on a daily basis during the course of our survey.

Except for preliminary descriptions by Lewis (1975), the Tobago Cay reefs have received little scientific attention (Wells, 1988; Smith et al., 1997). Hence, we aimed to characterize the present condition of Horseshoe Reef using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocol.

## METHODS

Field data were collected from June 7-16, 1999. Dive sites (Fig. 1) that were representative of areas of maximum reef development were selected with the help of true color aerial photographs (scale 1:10 000, March 1991), nautical charts (Hydrographic Office of the United Kingdom, 1999), reconnaissance dives, and the local knowledge of experienced divers and dive-shop operators. Site selection was limited, however, by accessibility to boats and by strong tidal currents. Two shallow reefs (B, D), located between 1 and 5 m in the shallow fore reef, consisted of dead *A. palmata* pavement, largely encrusted by crustose coralline algae and colonized by *Millepora* and scattered *Porites*. Three deeper reefs (A, C, E), located between 8 and 15 m on the fore-reef slope, consisted of dead coral pavement, largely colonized various coral species dominated by *Montastraea* and *Porites* with scattered patches of *Halimeda*.

Stony corals, algal groups, and *Diadema antillarum* were censused by one or two divers/survey. The following modifications were made to the AGRRA Version 2.2 protocols (see Appendix One, this volume). The size of individual corals was recorded to the closest 5 cm. Damselfish that were defending territories on censused corals were counted. Sediment in the algal quadrats was lightly brushed away before scoring the abundance of crustose coralline algae. Macroalgal heights were measured to the nearest 0.5 cm. Small (<2 cm diameter) corals of species that are tiny as adults (e.g., *Favia fragum*) were not counted as "recruits." Before starting the surveys, pilot transects were conducted in the back reef and results compared to ensure consistency between divers.

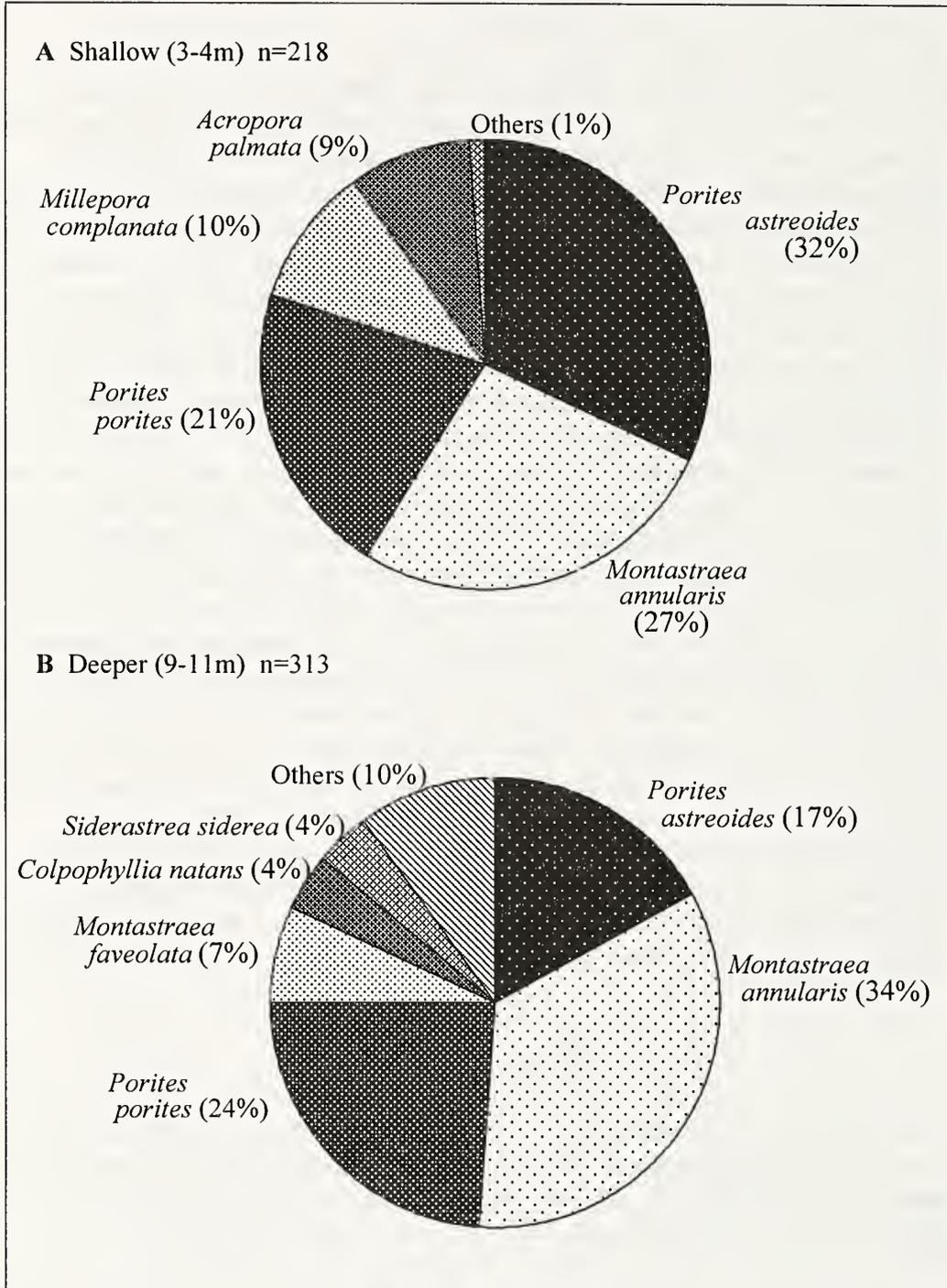
All fish surveys were made by one diver between 9:00 a.m. and 4:00 p.m. Counts of serranids (groupers) were restricted to species of *Epinephelus* and *Mycteroperca*; scarids (parrotfishes) and haemulids (grunts) less than 5 cm in length were not tallied. Roving Diver Transect (RDT) surveys averaged 30 minutes each. Fish biomass was estimated using the length-weight relationships given in Appendix Two (this volume). Field guides included Humann (1994, 1996).

## RESULTS

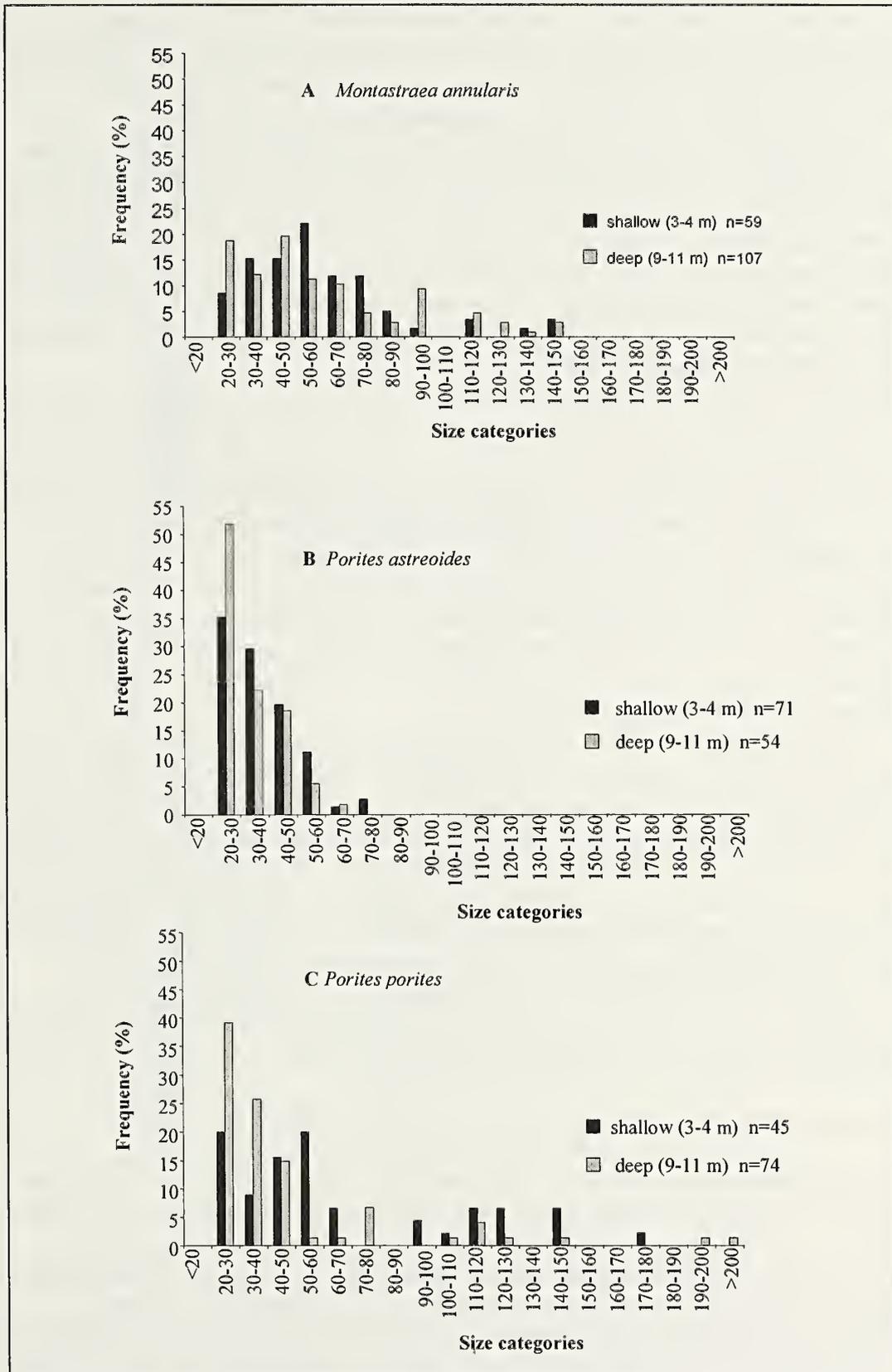
A total of 60 benthic line transects with 531 corals, 268 algal quadrats, 10 roving diver fish counts and 50 fish-belt transects were conducted on the Horseshoe Reef (Fig. 1, Table 1). Weather conditions were good during most dives with horizontal visibility estimated at approximately 20-25 meters.

### Stony Corals

The assemblage of "large" stony corals ( $\geq 25$  cm maximum diameter) was represented by 16 species and dominated overall by *Montastraea annularis* (31%), *Porites astreoides* (23%), *P. porites* (23%), *Montastraea faveolata* (5%), *Millepora complanata* (4%), *Colpophyllia natans* (2%) and *Siderastrea siderastrea* (2%). Nine large coral species predominated at the two shallow reefs (Fig. 2A) with *P. astreoides* > *M. annularis*



**Figure 2.** Species composition and mean relative abundance of all stony corals ( $\geq 25$  cm diameter) on (A) shallow, (B) deeper fore reefs at Horseshoe Reef.



**Figure 3.** Size-frequency distribution in cm of colonies ( $\geq 25$  cm diameter) of (A) *Montastraea annularis*, (B) *Porites astreoides*, (C) *Porites porites* at Horseshoe Reef.

> *P. porites* > *M. complanata* ~ standing dead colonies of *Acropora palmata* (recognizable by their characteristic colony shape, and only present at site B). The deeper reefs, which had a total of 15 large coral species, were dominated by *M. annularis* > *P. porites* > *P. astreoides* > *M. faveolata* > *C. natans* ~ *S. siderea* (Fig. 2B).

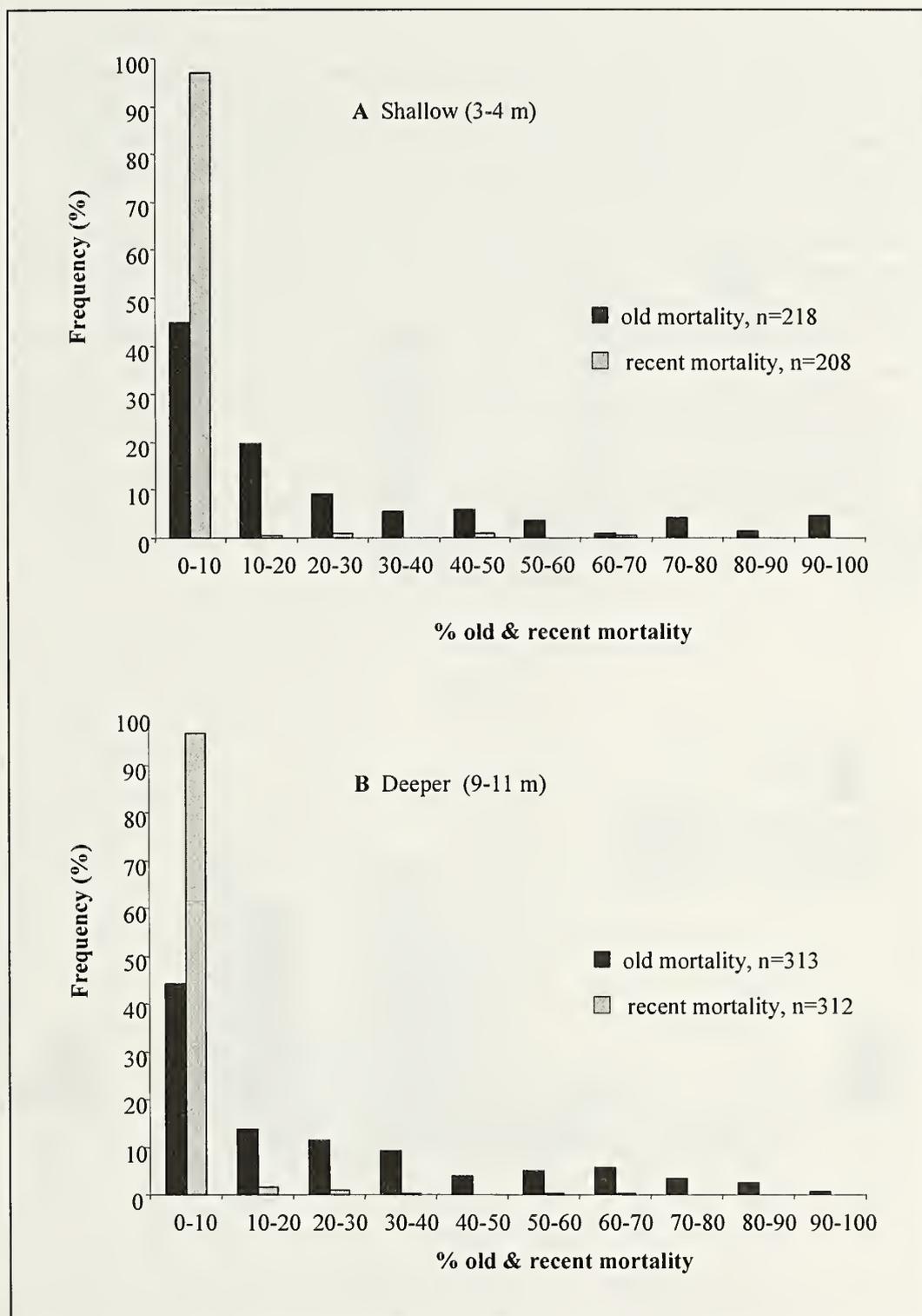
Live stony coral cover averaged about 30% at the shallow reefs and 38% at the deeper reefs (Table 1). The mean diameter of the large colonies among sites ranged from 50 to 65 cm (Table 2). Size-frequency distributions for the three major reef builders (*M. annularis*, *P. astreoides*, *P. porites*) were skewed towards smaller colonies (Fig. 3). Larger clumps of *P. porites* (to 300 cm) and mounds of *Montastraea annularis* (to 150 cm) were also present but relatively uncommon. Coral recruitment averaged 0.25/0.625 m<sup>2</sup> (~4/m<sup>2</sup>) the shallow reefs and 0.16/0.625 m<sup>2</sup> (~2.5/m<sup>2</sup>) in the deeper reefs (Table 3). *Agaricia* and *Porites* were the most commonly seen recruits.

Fewer than 4% of the large corals at the shallow and deep sites of Horseshoe Reef showed signs of disease (Table 2). Four percent of the colonies of *Montastraea* were affected by yellow-blotch disease (YBD) while 2% of all massive corals had black-band disease (BBD). Whereas BBD was only present at one deeper reef (E), YBD occurred at all sites but C. No large corals were completely bleached and the percentage that were either pale or partially bleached varied from about 2.5% to 10.5% in the shallow and deeper reefs, respectively (Table 2). *Montastraea* accounted for 45% of the bleached corals in shallow reefs and 84% of bleached corals in deeper reefs. However, the overgrowing mat tunicate, *Trididemnum solidum*, which was present at all five sites, had partially encrusted about 10% (54/531) of the surveyed corals, primarily *Porites*, *Montastraea* and, occasionally, *Millepora*.

Partial-colony mortality on the upward-facing surfaces of surveyed corals averaged 2% for recent mortality (hereafter recent mortality) and 25% for old mortality (hereafter old mortality) (Table 2). Frequency distributions (Fig. 4 A, B) reveal that recent mortality for over 95% of these corals was less than 10% at both shallow and deeper sites, whereas fewer than 50% had old mortality values within this range. The most commonly observed sources of recent mortality were predation by *Coralliophila abbreviata*, parrotfish bites, and incorporation in damselfish algal gardens. Over 60% of the 143 damselfish counted were associated with the *M. annularis* species complex, with about a quarter of the colonies having at least one resident algal gardener. At the deeper reefs, localized bleaching was associated with dense stands of the calcareous macroalga *Halimeda* that were observed overgrowing the margins of some stony corals.

#### Algae and *Diadema antillarum*

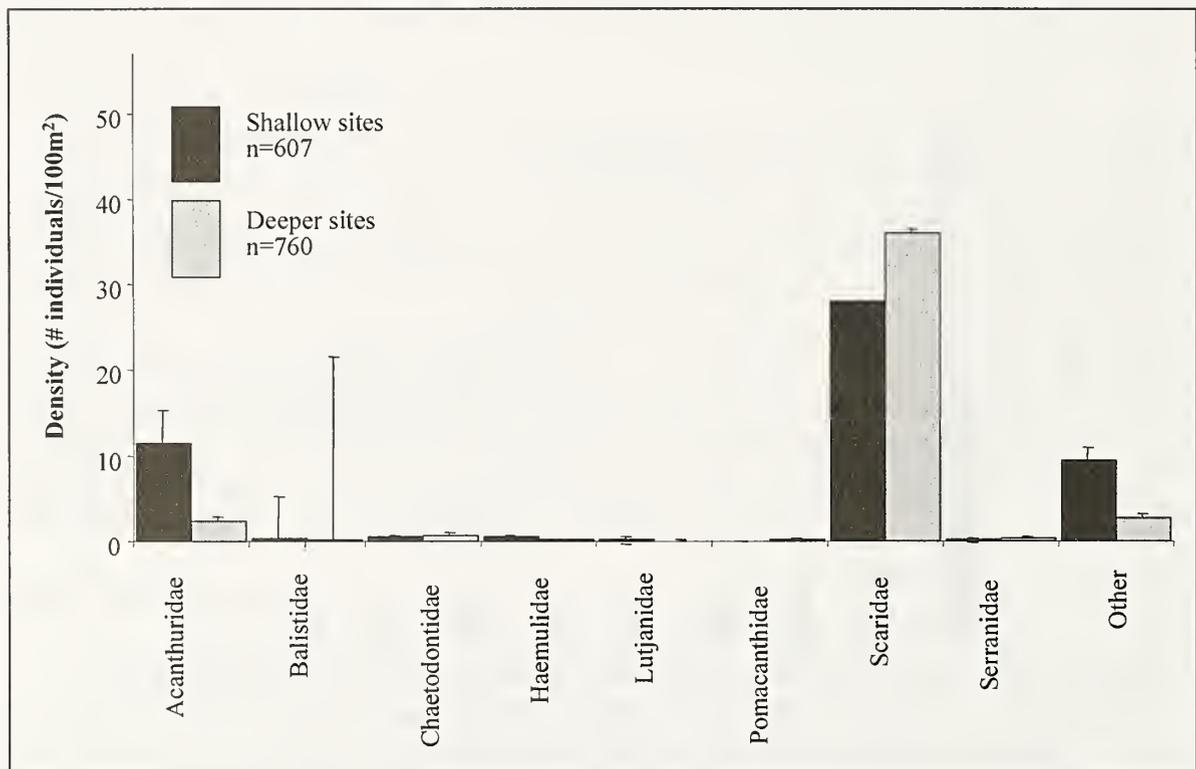
Algal communities at the two shallow and one of the deeper sites were dominated by crustose coralline algae, whereas macroalgae (particularly the calcareous *Halimeda* spp., and, to a lesser degree, the fleshy *Dictyota* spp.) were the predominant algal functional group at the remaining two deeper reefs (Table 3). Average macroalgal canopy height was 1.5 cm (Table 3). Macroalgal indices (macroalgal relative abundance x macroalgal height) were considerably higher in the deeper reefs (~26 at 3-4 m versus ~64 at 9-12 m respectively). *Diadema antillarum* was present with mean densities of about five individuals/100 m<sup>2</sup> in the shallower reefs and about 0.5/100 m<sup>2</sup> in the deeper reefs.



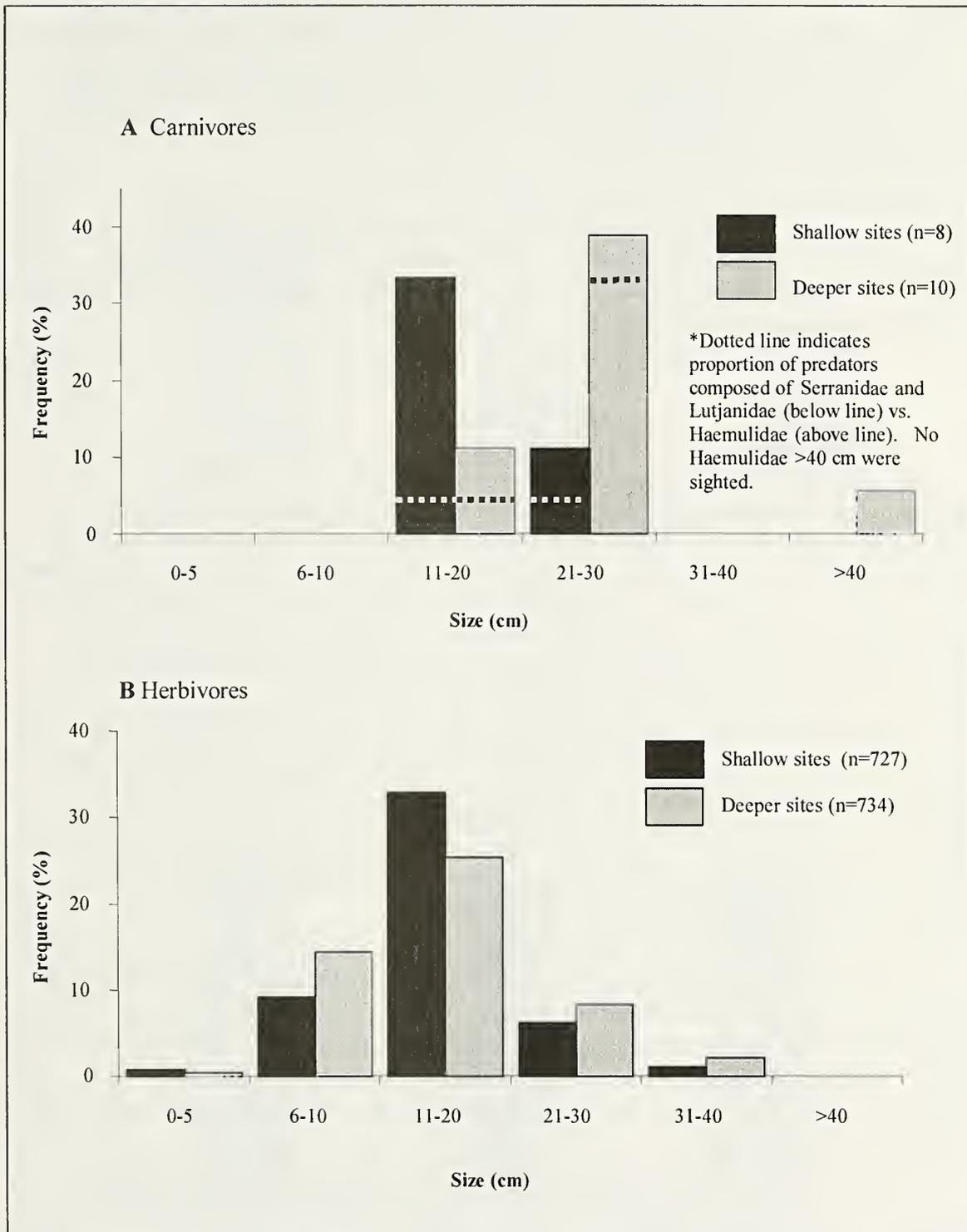
**Figure 4.** Frequency distribution for all stony corals ( $\geq 25$  cm diameter) of old partial colony mortality and of recent partial colony mortality at (A) shallow and (B) deeper fore reefs at Horseshoe Reef.

## Fishes

A total of 81 species of fish were recorded during the 10 roving diver surveys ( $\approx 7$  hours). The most commonly sighted were planktivorous pomacentrids and herbivorous scarids, pomacentrids and acanthurids, followed by planktivorous labrids (Table 4). *Aulostomus maculatus* (trumpetfish), present in all surveys, was the most common predator. The fish assemblage quantified in the belt transects was dominated by scarids with 28 and 36 individuals/100 m<sup>2</sup> having body lengths of at least 5 cm in shallow and deeper sites, respectively (Fig. 5). Acanthurids were present with higher average densities at 3-4 m than at 9-12 m (12 versus 2 individuals/100 m<sup>2</sup>, respectively). Similarly, *Microspathodon chrysurus* (yellowtail damselfish) were more abundant in shallow reefs (9 individuals/100 m<sup>2</sup>) than in the deeper reefs (3 individuals/100 m<sup>2</sup>). Commercially valuable species of select serranids, lutjanids (snappers), and haemulids ( $\geq 5$  cm only) were collectively present at densities of less than 1 individual/100 m<sup>2</sup> ( $\sum n = < 20$  individuals). It is notable, however, that although lutjanids were less commonly encountered in the belt transects than serranids or haemulids, *Lutjanus mahogoni* (mahogany snapper) were sighted more frequently (90 percent of dives) than serranids or haemulids during the roving diver surveys (Table 4).



**Figure 5.** Mean fish abundance (no. individuals/100 m<sup>2</sup> ± se) for AGRRA fishes at Horseshoe Reef. Other = *Microspathodon chrysurus* > *Bodianthus rufus*.



**Figure 6.** Size frequency distribution of (A) carnivores (lutjanids, select serranids, haemulids  $\geq 5$  cm), (B) herbivores (all acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) at Horseshoe Reef

Although commercially significant predators were rare in our belt transects, half (shallow) to most (deeper) of the groupers and snappers encountered ranged in size from 21-30 cm (Fig. 6A). Serranids (mean = 29 cm) were about twice as long as lutjanids (mean = 16 cm), which overlapped in size with the haemulids (all of which were  $\geq 5$  cm) (Fig. 7A). The lengths of half or more of each of the herbivorous groups (scarids  $\geq 5$  cm, acanthurids and *Microspathodon chrysurus*) were in the 11-20 cm length class and averaged about 15 cm (Figs. 6B, Fig. 7B).

Of the AGRRA fishes surveyed in the belt transects, the biomass of the herbivorous guild was 90 percent greater than that of the commercially valuable carnivores (Fig. 8). The scarid biomass was evenly distributed between shallow and deeper sites. Biomass estimates of acanthurids and *M. chrysurus* were considerably greater in the shallow reefs than in the deeper fore reefs (Table 5).

## DISCUSSION

The coral assemblage and coral cover on Horseshoe Reef are typical of high-energy, windward Caribbean reefs and, except for the near disappearance of live *A. palmata* from the reef flat and shallow fore reef, very similar to the descriptions of Lewis (1975) (some juvenile *A. palmata* are present in the back-reef area immediately behind the reef crest). The presence of standing dead colonies of *A. palmata* in the shallow fore reefs is a good indication that they were not killed by hurricanes. Hence, we concur that their demise at Horseshoe Reef, as elsewhere in St. Vincent-Grenadines, was probably due to white-band disease (Antonius, 1981; Aronson and Precht, 1997).

The positively skewed size distribution of the three major reef-building corals may be an indication of a reef system with adequate juvenile input (Bak and Meesters, 1998) since large-scale fragmentation by hurricanes seems unlikely on Horseshoe Reef (see Introduction). The slightly lower density of coral recruits on the deeper reefs is probably related to the generally higher abundance of macroalgae as well as the relatively lower abundance of crustose coralline algae and the near absence of *Diadema antillarum* (Birkeland 1977; Pearson, 1981; Hughes et al., 1987).

None of the factors contributing to recent mortality are responsible for major damage to the corals at Horseshoe Reef. The pale or partially bleached corals seen during our survey, particularly on the deeper reefs, were possibly still recovering from the 1998 Caribbean-wide bleaching event (Strong et al., 1998) that affected reefs in the Tobago Cays (Kurt Cordice, personal communication). *Halimeda*, engaged in marginal overgrowth of some stony corals, may also have contributed to the higher occurrence of bleaching on these deeper reefs.

Competitive overgrowth by organisms such as *Trididemnum solidum*, which was not noted in the Tobago Cays by Lewis (1975), may be influenced by environmental perturbations that reduce the efficiency of affected organisms to defend themselves. Thus, its relatively high abundance (on 10% of the large colonies) at Horseshoe Reef is cause for concern. Although the effects of *T. solidum* were scored as "old mortality" in accordance with the AGRRA protocol, there were many cases for which we suspected that loss of coral tissues had occurred within recent months. Had *T. solidum* been recorded under "recent" mortality, our estimates of the latter would have been substantially larger.

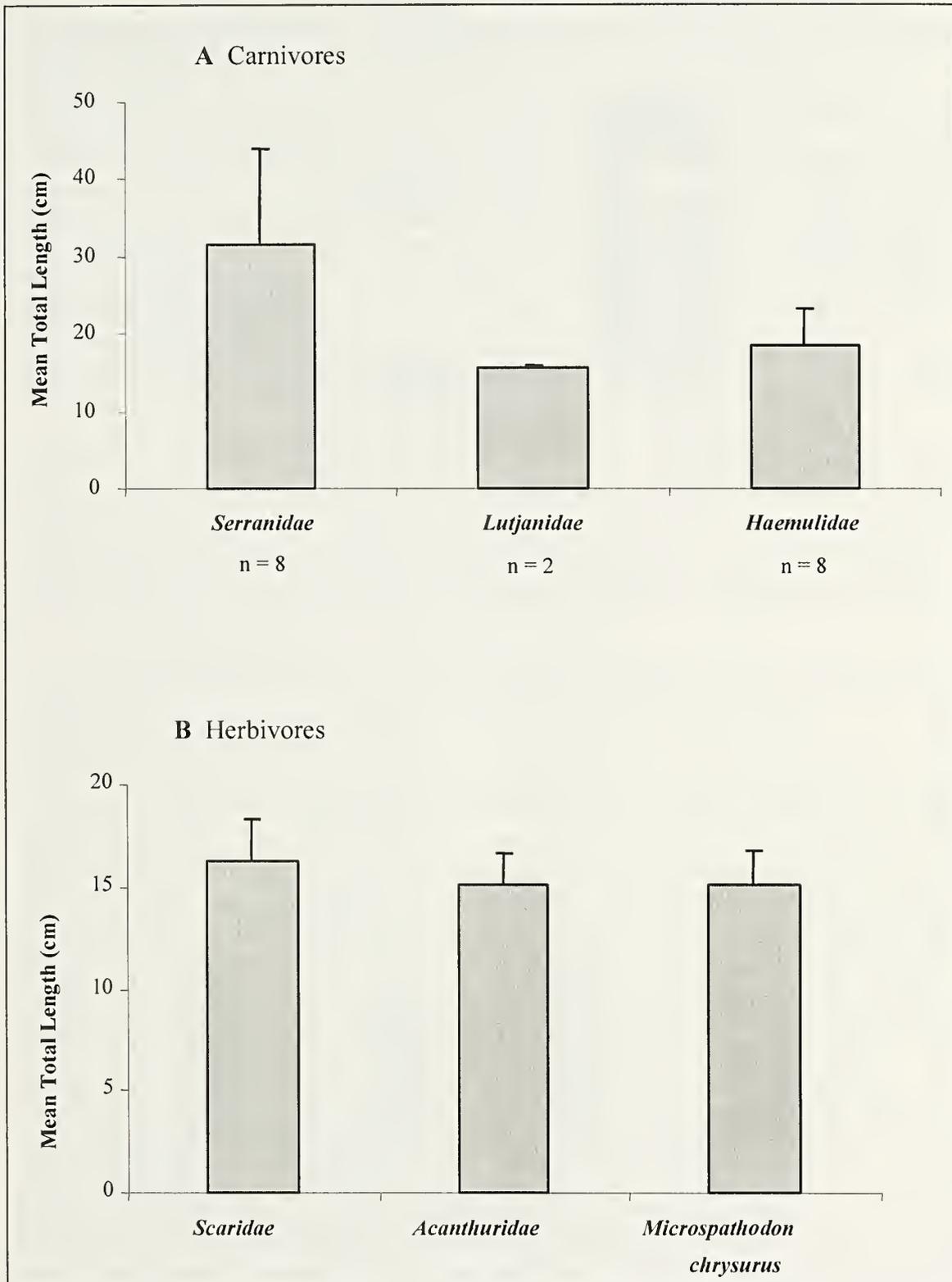


Figure 7. Mean total length ( $\pm$  se) of key (A) carnivores, (B) herbivores at Horseshoe Reef.

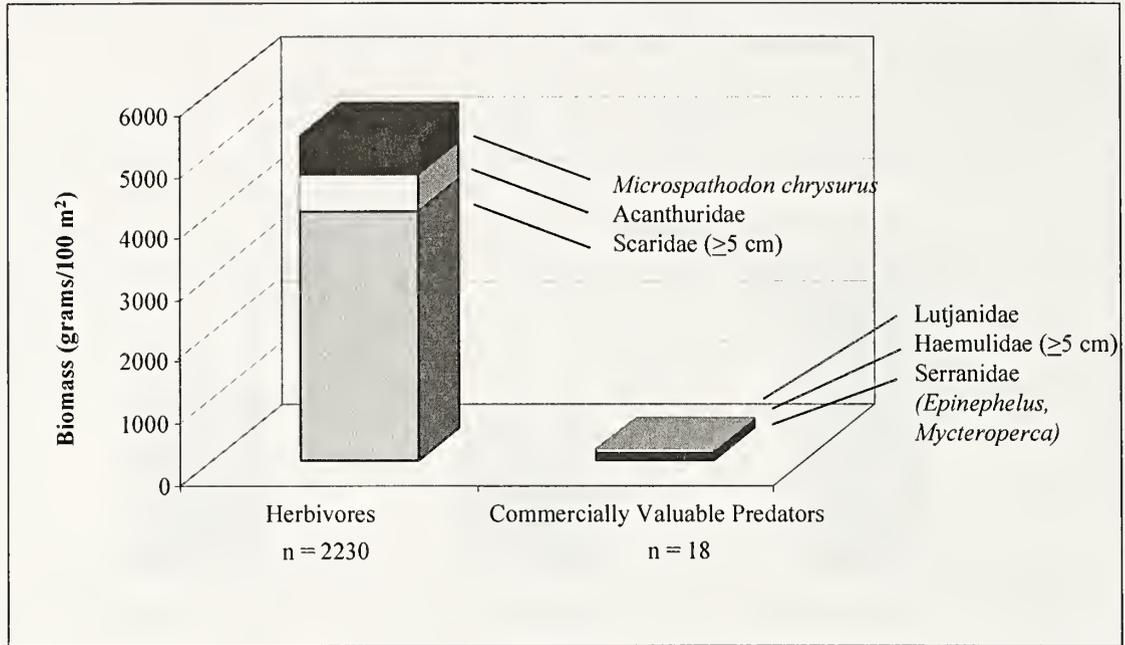


Figure 8. Mean biomass of key herbivores and carnivores at Horseshoe Reef.

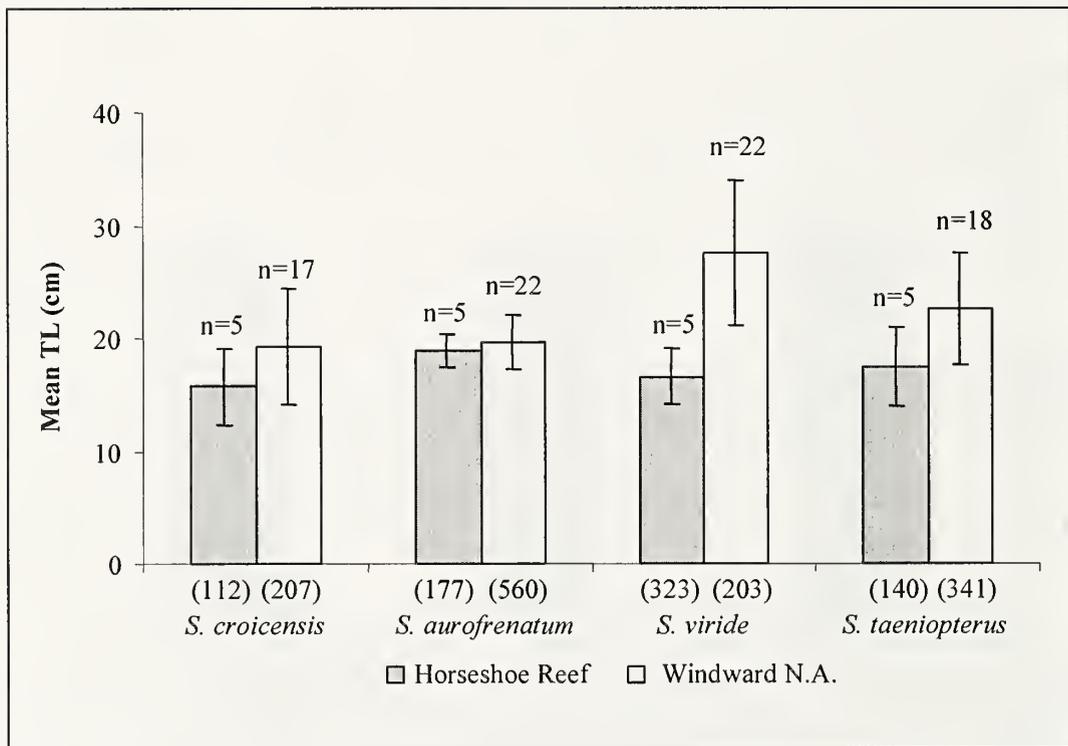


Figure 9. Mean total length ( $\pm$  se) of common scarids at Horseshoe Reef and from the windward Netherland Antilles (Kloomp and Kooistra, this volume). n = number of sites; number of individual fishes is in parentheses.

Algal community structure is greatly affected by the spectrum of grazers present on an individual reef and by fishing pressures that significantly alter these natural grazing patterns (McClanahan and Muthiga, 1998). Macroalgae can easily overgrow smaller algal turfs and crustose coralline algae (Lewis, 1986). We suspect that the herbivores (e.g., scarids, *D. antillarum*; to a lesser degree, acanthurids and *M. chrysurus*) on the shallow fore reefs at Horseshoe Reef are maintaining a benthic algal assemblage in which crustose corallines are spatial dominants. The somewhat greater proportion of turf algae at 3-4 m relative to 9-12 m may be a result of higher productivity rates which have been attributed to greater photosynthetic activity in shallower habitats, to the forces of herbivory that enhance growth by maintaining algal assemblages in an early successional stage, and to recycling of nutrients by herbivores back into the benthic community (Hatcher, 1997). Grazing intensity generally decreases with depth on fore reefs (Morrison, 1988) which may explain the relatively higher densities and biomass (Table 5) of herbivorous acanthurids and *M. chrysurus* at 3-4 m than at 9-11 m.

The macroalgae that predominated at two-thirds of the deeper reefs (*Halimeda*, *Dictyota*), where macroalgal grazers (particularly *Diadema*) are relatively less abundant, are of genera known to be avoided by herbivorous fishes (Schmitt, 1998). A reduction in herbivorous grazing pressure allows macroalgae to outcompete sessile reef invertebrates for space and can ultimately result in the degradation of coral communities (Hughes, 1989; Hughes et al., 1987). However, live stony coral cover was higher on the deeper fore reefs than in the shallow at the time of our survey.

As reported in Table 4, *L. mahogoni* was one of the 25 most frequently sighted species in the roving diver surveys, appearing in 9/10 surveys, yet was not seen in any of the belt transects. Its presence in the roving diver surveys was recorded as "few" (i.e., 2-10 individuals) in seven of these dives and as "many" (i.e., 11-100 individuals) in two dives. This may be an indication that the belt transect method is not adequately quantifying fish such as *L. mahogoni*, which are inherently wary of divers, or that a larger sample size is needed to adequately quantify species that are present but not necessarily abundant. Of additional interest from the roving diver surveys, *Chromis multilineata* (brown chromis) and *C. cyanea* (blue chromis) were recorded as being the most frequently sighted and "abundant" (>100 individuals) fishes on Horseshoe Reef. The prevalence of these planktivores may be an indication of reduced predation in the presence of an adequate food source. However, they are commonly found throughout the greater Caribbean strategically feeding above reefs where plankton is plentiful (Bohlke and Chaplin, 1993).

The Scaridae was the dominant fish family present in our belt transects on Horseshoe Reef (Fig. 5, Table 5). Scarids were represented primarily by *Sparisoma viride* (stoplight parrotfish) with lesser densities of *S. aurofrenatum* (redband parrotfish), *Scarus taeniopterus* (princess parrotfish), *S. vetula* (queen parrotfish) and *S. croicensis* (striped parrotfish) (Table 6). Average total lengths for *S. croicensis* and *S. aurofrenatum* were similar at Horseshoe Reef and in the windward Netherlands Antilles (N.A.) which have received a lesser degree of fishing pressure (Klomp and Kooistra, this volume). *S. viride* and *S. taeniopterus*, however, were both significantly smaller on average (F-test;  $p=0.001$  and  $p=0.044$ , respectively) than their conspecifics in the windward NA (Fig. 9). Although we cannot conclusively determine from this assessment the cause for the relatively small size of these parrotfishes in Horseshoe Reef, it may be further evidence of high harvesting levels.

Reduced abundance or biomass and a decreased size structure in targeted fishes as a result of intense fishing has been well-documented (Bohnsack, 1982; Munro, 1983). It was not common to see fish larger than 30 cm total length on Horseshoe Reef (Fig. 6) and we suspect that this is an effect of larger fish being targeted for harvest. Additionally, overfishing likely explains the low densities of grouper, snapper and grunts on Horseshoe Reef since these are favored food fishes. It is possible that these common carnivores have been replaced by other species, such as the "voracious predator" *Aulostomus maculatus* (trumpetfish), which is known to feed on young acanthurids and haemulids (Randall, 1967). *A. maculatus* are common residents of Caribbean-area reefs but ordinarily are not abundant (Bohlke and Chaplin, 1993). Randall (1967) found trumpetfish in the stomachs of snappers and groupers. Given the paucity of serranids and lutjanids, we suspect that the Horseshoe Reef population of *A. maculatus* is thriving due to reduced predation pressure and adequate prey supplies. This phenomenon may warrant further investigation if marine park managers seek to restore populations of snappers, groupers or grunts. The Vincentians rely on their artisanal fishery as a source of dietary protein. A restored population of commercially valuable carnivores within the marine park could serve as a source stock of fish and possibly create a spillover effect into legal fishing areas, providing harvestable resources for the fishing community.

In sum, strengthening and enforcing fishing regulations in the Tobago Cays Marine Park are critical steps to avoid further degradation of its fish assemblages. The AGRRA survey, being the first quantitative, assessment of Horseshoe Reef, provides baseline information against which to measure future changes in its condition.

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Table 1. Site information for AGRRA stony coral, algal and fish surveys on Horseshoe Reef, Tobago Cays, St. Vincent.

Site name	Latitude (°' " N)	Longitude (°' " W)	Survey date	Depth (m)	Benthic transects (#)	≥25 cm stony corals (#/10 m)	Live stony coral cover (mean ± se)	30 m fish transects (#)	RDT fish species (#) <sup>1</sup>
B	12 38 00.48	61 20 56.88	June 8-9 1999	3.5	14	8	29.0 ± 1.5	10	45
D	12 38 08.58	61 20 55.38	June 14, 1999	3	13	8	31.5 ± 1.5	10	44
<b>Shallow</b>				<b>3.5</b>	<b>27</b>	<b>8</b>	<b>30.5 ± 1.0</b>	<b>20</b>	<b>56</b>
A	12 38 00.48	61 20 56.88	June 7-8 1999	11.5	11	10	42.5 ± 2.5	10	49
C	12 37 48.06	61 21 04.02	June 10-11 1999	9	11	9	44.0 ± 2.5	10	41
E	12 38 08.58	61 20 54.46	June 10-11 1999	11	11	9	28.5 ± 2.5	10	50
<b>Deep</b>				<b>10.5</b>	<b>33</b>	<b>9</b>	<b>38.0 ± 1.0</b>	<b>30</b>	<b>77</b>

<sup>1</sup>RDT = Roving Diver Technique

Table 2. Size and condition (mean ± standard error) of all stony corals (≥25 cm diameter) by site on Horseshoe Reef.

Site name	Stony corals		Partial-colony surface mortality (%)				Stony corals (%)		
	(#)	Diameter (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased	
B	109	53.5 ± 3.0	1.0 ± 0.5	26.5 ± 3.0	27.5 ± 3.0	9	3	1	
D	109	65.0 ± 3.5	3.0 ± 1.0	24.0 ± 2.0	27.0 ± 2.0	0	2	2	
<b>Shallow</b>	<b>218</b>	<b>59.0 ± 2.5</b>	<b>2.0 ± 0.5</b>	<b>25.0 ± 2.0</b>	<b>27.5 ± 2.0</b>	<b>4.5</b>	<b>2.5</b>	<b>1.5</b>	
A	108	55.0 ± 3.0	1.0 ± 0.5	22.5 ± 2.5	24.0 ± 2.5	0	15	4.5	
C	104	64.0 ± 4.0	2.5 ± 0.5	25.5 ± 2.5	28.0 ± 2.5	1	6.5	1	
E	101	50.0 ± 3.0	2.0 ± 0.5	28.0 ± 2.5	30.0 ± 2.7	0	10	6	
<b>Deep</b>	<b>313</b>	<b>56.5 ± 2.0</b>	<b>2.0 ± 0.5</b>	<b>25.5 ± 1.5</b>	<b>27.0 ± 1.5</b>	<b>0.5</b>	<b>10.5</b>	<b>4</b>	

Table 3. Algal characteristics (mean  $\pm$  standard error), abundance of stony coral recruits and *Diadema antillarum*, by site on Horseshoe Reef.

Site name	Quadrats	Relative abundance (%)			Macroalgal Height (cm)	Macroalgal Index <sup>1</sup>	Recruits (#/0625 m <sup>2</sup> )	Diadema (#/100 m <sup>2</sup> )
		Macroalgae	Turf algae	Crustose coralline algae				
B	58	21.5 $\pm$ 3.5	23.0 $\pm$ 3.5	56.5 $\pm$ 4.0	1.5 $\pm$ 0.5	32	0.41 $\pm$ 0.14	7
D	51	12.0 $\pm$ 3.0	35.0 $\pm$ 3.5	35.0 $\pm$ 3.5	1.5 $\pm$ 0.5	18	0.09 $\pm$ 0.04	3
<b>Shallow</b>	<b>109</b>	<b>17.0 <math>\pm</math> 2.0</b>	<b>28.5 <math>\pm</math> 2.5</b>	<b>54.5 <math>\pm</math> 3.0</b>	<b>1.5 <math>\pm</math> 0.5</b>	<b>26</b>	<b>0.25 <math>\pm</math> 0.07</b>	<b>5</b>
A	53	55.0 $\pm$ 3.0	17.0 $\pm$ 2.0	28.0 $\pm$ 3.0	1.5 $\pm$ 0.5	82	0.03 $\pm$ 0.01	0
C	54	48.0 $\pm$ 4.0	14.0 $\pm$ 1.5	38.0 $\pm$ 3.5	1.5 $\pm$ 0.5	72	0.28 $\pm$ 0.08	2
E	52	23.5 $\pm$ 4.0	27.5 $\pm$ 4.0	48.5 $\pm$ 4.0	2.0 $\pm$ 0.5	47	0.20 $\pm$ 0.07	0
<b>Deep</b>	<b>159</b>	<b>42.5 <math>\pm</math> 2.0</b>	<b>19.5 <math>\pm</math> 1.5</b>	<b>38.0 <math>\pm</math> 2.0</b>	<b>1.5 <math>\pm</math> 0.5</b>	<b>64</b>	<b>0.16 <math>\pm</math> 0.03</b>	<b>0.5</b>

<sup>1</sup>Macroalgal index = macroalgal relative abundance x macroalgal height

Table 4. Twenty-five most frequently sighted fish species during rover diver surveys on Horseshoe Reef, with densities (mean  $\pm$  standard error) for species recorded in belt transects

Fish Species	Fish Species	Sighting frequency <sup>1</sup>	Density (#/100 m <sup>2</sup> )
<i>Chromis multilineata</i>	Brown Chromis	100	-
<i>Chromis cyanea</i>	Blue Chromis	100	-
<i>Sparisoma viride</i>	Stoplight Parrotfish	100	10.8 $\pm$ 3.2
<i>Sparisoma aurofrenatum</i>	Redband Parrotfish	100	5.9 $\pm$ 1.8
<i>Microspathodon chrysurus</i>	Yellowtail Damselfish	100	5.0 $\pm$ 3.2
<i>Halichoeres garnoti</i>	Yellowhead Wrasse	100	-
<i>Acanthurus coeruleus</i>	Blue Tang	100	3.6 $\pm$ 4.6
<i>Mulloidichthys martinicus</i>	Yellow Goatfish	100	-
<i>Aulostomus maculatus</i>	Trumpetfish	100	-
<i>Halichoeres maculipinna</i>	Clown Wrasse	100	-
<i>Thalassoma bifasciatum</i>	Bluehead Wrasse	90	-
<i>Clepticus parrai</i>	Creole Wrasse	90	-
<i>Stegastes partitus</i>	Bicolor Damselfish	90	-
<i>Stegastes planifrons</i>	Threespot Damselfish	90	-
<i>Lutjanus mahogoni</i>	Mahogany Snapper	90	0
<i>Holocentrus rufus</i>	Longspine Squirrelfish	90	-
<i>Canthigaster rostrata</i>	Sharpnose Puffer	90	-
<i>Scarus taeniopterus</i>	Princess Parrotfish	80	4.7 $\pm$ 3.3
<i>Scarus vetula</i>	Queen Parrotfish	80	5.2 $\pm$ 6.8
<i>Ophioblennius atlanticus</i>	Redlip Blennie	80	-
<i>Acanthurus bahianus</i>	Ocean Surgeonfish	80	2.1 $\pm$ 1.1
<i>Hypoplectrus chlorurus</i>	Yellowtail Hamlet	80	-
<i>Abudefduf saxatilis</i>	Sergeant Major	70	-
<i>Scarus croicensis</i>	Striped Parrotfish	70	3.7 $\pm$ 1.5
<i>Paranthias furcifer</i>	Creole-fish	70	-

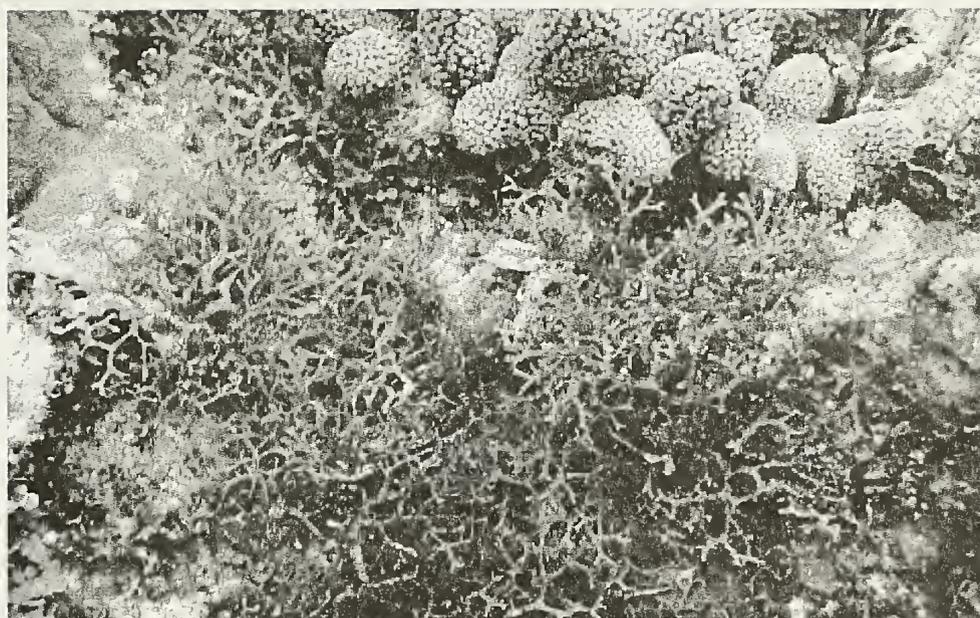
<sup>1</sup>Percent sighting frequency = percent of dives in which the species was recorded.

Table 5. Biomass of major fish families (as mean  $\pm$  standard error), by depth on Horseshoe Reef.

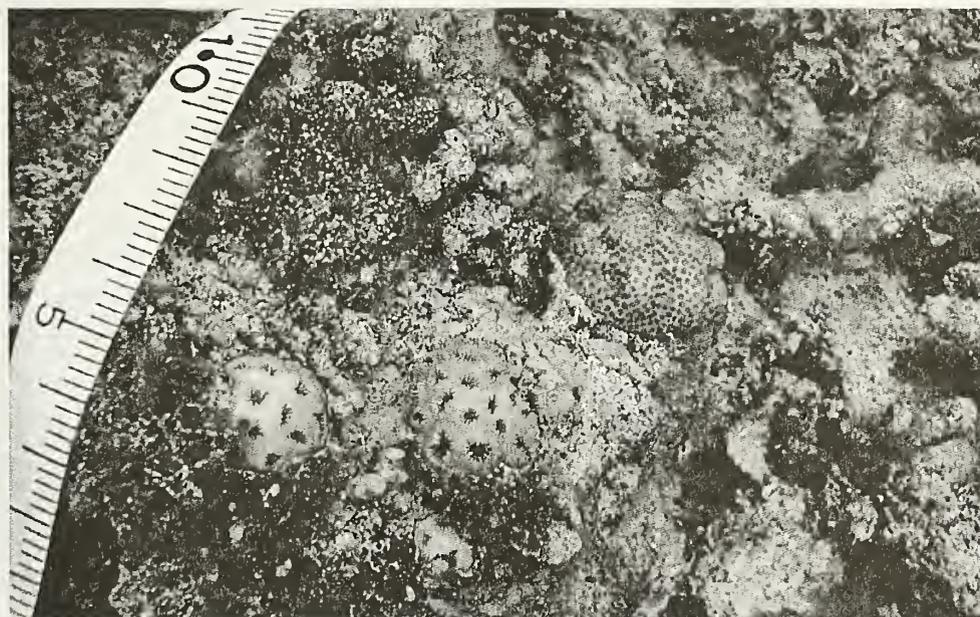
Sites	Herbivores (grams/100 m <sup>2</sup> )			Carnivores (grams/100 m <sup>2</sup> )		
	Acanthuridae	Scaridae ( $\geq 5$ cm)	<i>Microspathodon chrysurus</i>	Haemulidae ( $\geq 5$ cm)	Lutjanidae	Serranidae
All Shallow	1081 $\pm$ 364	3827 $\pm$ 487	1145 $\pm$ 354	84 $\pm$ 228	6 $\pm$ 7	22 $\pm$ 12
All Deep	269 $\pm$ 67	4217 $\pm$ 605	330 $\pm$ 231	28 $\pm$ 128	4 $\pm$ 6	198 $\pm$ 25

Table 6. Mean density of the  $\geq 5$  cm scarids, by site on Horseshoe Reef.

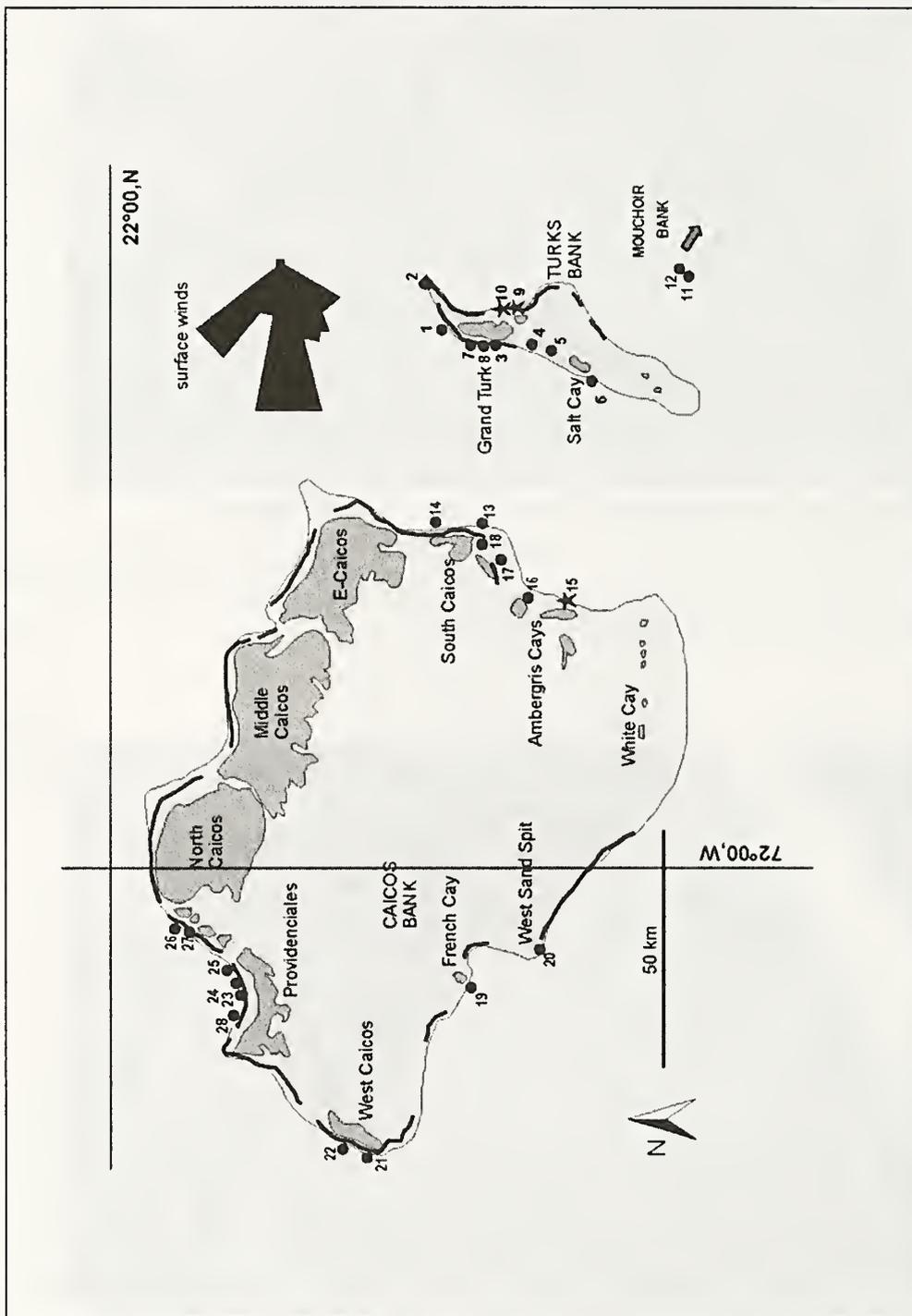
Site	Density (# individuals/100 m <sup>2</sup> )				
	<i>S. aurofrenatum</i> (redband) (n = 177)	<i>S. taeniopterus</i> (princess) (n = 140)	<i>S. viride</i> (stoplight) (n = 323)	<i>S. croicensis</i> (striped) (n = 112)	<i>S. vetula</i> (queen) (n = 156)
<i>Shallow</i> B	4.3	1.3	11.5	5.3	1.8
D	6.7	2.7	14.0	2.7	5.0
<i>Deep</i> A	5.3	3.0	5.3	2.3	1.2
C	8.7	8.0	12.0	5.5	17.0
E	4.5	8.3	11.0	2.8	1.0
All sites mean $\pm$ se	6.2 $\pm$ 2.2	6.4 $\pm$ 3.0	9.4 $\pm$ 3.6	3.6 $\pm$ 1.7	6.4 $\pm$ 9.2



**Plate 11A.** Reef condition is strongly dependent on the interplay of complex relationships involving stony corals, herbivores and benthic algae. Algae are assessed in the AGRRA benthos protocol as the relative abundance of several key functional algal groups, including macroalgae like the *Dictyota* shown here, in relation to coral condition and herbivorous fishes. (Photo Robert W. Steneck)



**Plate 11B.** Crustose coralline algae that grow around and between coral fragments may eventually immobilize loose pieces of coral rubble and may serve as recruitment sites for coral larvae. Recruitment is estimated as the number of small (<2 cm diameter) stony corals visible in the algal quadrats, like those shown here. (Photo Robert W. Steneck)



**Figure 1.** AGRRA survey sites in the Turks and Caicos Islands. See Table 1 for site codes. Wind rose for the southern Bahamas, from R.N. Ginsburg in P.A. Scholte, and N.P. James (1995).

# ASSESSMENT OF THE CORAL REEFS OF THE TURKS AND CAICOS ISLANDS (PART 1: STONY CORALS AND ALGAE)

BY

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## ABSTRACT

Major constituents of the benthic reef community (stony corals, algae) were assessed in 28 reefs on the Caicos, Turks and Mouchoir Banks. Living stony coral cover ranged from 8-28%, averaging 18% overall. *Montastraea annularis* and *M. faveolata* of “intermediate” sizes (<100 cm) dominated all examined reefs. Live *Acropora palmata* and *A. cervicornis* were scarce. The most frequently recruiting scleractinians were *Porites astreoides* and *Agaricia agaricites*; *Montastraea* recruits were uncommon. Old partial-colony mortality (overall mean=23%) was greater than recent partial-colony mortality (mean=3%). Crustose coralline algae and turf algae were generally more abundant than macroalgae. Mouchoir Bank, with the most isolated reefs, was in relatively poor condition, which suggests that remoteness alone does not protect coral reefs.

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## INTRODUCTION

The Turks and Caicos Islands (TCI), which lie at 21° to 22° N and 71° to 72° 30' W, consist of 8 islands (7 of which are inhabited) and approximately 40 low-lying cays distributed among two banks (Turks Bank, Caicos Bank) plus part of the entirely submerged Mouchoir Bank (Fig. 1). Over 300 km of coral reef surround the Turks and Caicos Islands (Wells, 1988). The prevailing easterly trade winds (see windrose for the TCI area in Fig. 1) create a clear differentiation on the banks into a windward eastern side with generally choppy conditions and a leeward western side that is usually calm. The banks have narrow, discontinuous, shelf-edge reef (SER) systems (*sensu* Blanchon and Jones, 1997) of variable depth, relief, and stony coral abundance (Chiappone et al., 1996). Along the western parts of the Caicos and Turks Banks, shallow fringing reefs are developed shoreward of the SERs. Shallow patch reefs also surround many of the islands and cays. Underwater visibility is considered good everywhere.

The reefs and banks of the Turks and Caicos Islands have been studied by Wanless and Dravis (1989), Sullivan et al. (1994), Gaudian (1995, unpublished report), Chiappone et al. (1996) and Steiner (1999). In the context of the ongoing general deterioration of reef health in the entire Caribbean basin (e.g., Ginsburg, 1994), these isolated islands, with relatively small human population pressures, are of particular interest as landmark study sites. Information to date indicates that the Turks and Caicos reefs are generally in good condition with some pollution impacts evident near the islands of Providenciales and Grand Turk (e.g., Sullivan et al., 1994; Lang et al., 1998; Steiner, 1999; Woodley et al., 2000). Hence, they can be used for comparison with other sites subjected either to direct continental influences or to higher impacts, both natural and anthropogenic.

This study presents: 1) the August 1999 Atlantic and Gulf Rapid Reef Assessment (AGRRA) results for benthic reef condition; 2) an evaluation of differences between shallow versus deep reefs and between windward versus leeward reefs; and 3) a qualitative comparison of reefs on the three banks (which are known to experience different levels of resource extraction). Our AGRRA fish surveys for the Turks and Caicos Islands are presented by Hoshino et al. (this volume).

## METHODS

Survey sites (Fig. 1) were selected with the assistance of locally available diving and sailing maps, charts (British Admiralty, U.S. Navy), maps in publications, and aerial photographs. We chose strategically accessible reefs (e.g., at established dive sites with mooring buoys) that were considered representative of special interests (i.e., reported to be heavily impacted, or of touristic, fisheries and/or conservation value). Although an effort was made to space sites as evenly as possible within all available exposures and reef types,

small boats and prevailing sea conditions restricted our surveys to areas of moderate exposure and/or short traveling distance. However, a mix of 11 moderately exposed and 17 sheltered reefs were obtained. The northern side of the Caicos Bank and much of Mouchoir Bank were not exhaustively investigated. On Turks Bank we sampled all available habitats within the appropriate depth intervals, but largely ignored the southern area south of Salt Cay. Since an *Acropora palmata* reef-crest zone was not encountered in any of the areas examined, we surveyed three shallow patch reefs at depths of 2.5-6.5 m (Table 1). The patch reefs had been constructed primarily of *A. palmata*, still had some live colonies of this species, and were considered representative of several other patch reefs that we also visited. Elsewhere we made qualitative notes of the abundance of *A. palmata*. The remaining surveys were located in depths of 9.5-22.5 m on the seaward margin of spurs in the SERs. Nine were high-relief (>5 m) and 16 had lower relief (<5 m) but all showed groove-and-spur morphology with sand-filled channels running between the reef lobes.

Three divers executed the AGRRA Version 2.1 benthos protocol (see Appendix One, this volume) using the following modifications: pockets of sand underlying the transect line were not measured; assessments were made for each stony coral of 10 cm or greater diameter beneath the transect line; colony height and diameter were measured to the nearest 5 cm or, when possible, the nearest cm. *Porites furcata* and *P. divaricata* were not separated from *P. porites*, and species of *Agaricia* were not determined but about 90% of the surveyed corals are thought to have been *A. agaricites* with most of the rest consisting primarily of *A. fragilis* and *A. humilis*. Diseases were characterized by criteria established by Antonius (1995), Santavy and Peters (1997), and Peters (1997). We looked for damselfish tending algal gardens on the individually surveyed corals but none were recorded. Species that are small as adults (e.g., *Favia fragum*) were not included in the counts of stony coral "recruits." Sediment was removed from the algal quadrats by fanning the substratum two or three times by hand after scoring the cover of algal turfs and macroalgae but prior to estimating the abundance of crustose coralline algae. Absolute algal abundance estimates frequently exceeded 100% since each layer was estimated separately.

Numerous consistency checks were performed. Prior to beginning the survey, all divers performed measurements on the same transect and the results were compared. This process was repeated until results were homogeneous within the group. In total, five training transects were necessary. During the surveys, divers repeatedly discussed coral identification and interpretation of mortality, disease, algal cover, etc. The field guide used for identification of marine organisms was Human (1993).

For statistical evaluations, the reefs were grouped into three ecological units: shallow *Acropora palmata*-built patch reefs; high-relief SERs; and low-relief SERs. All data were found to be normally distributed with Kolmogoroff-Smirnow one-sample tests for normality of distribution. Parametric testing statistics were used to compare groups by means of the student's t-test or one-way analysis of variance and Tukey's post-hoc test to

identify significant groupings. Differences between the three banks were not tested for significance since the sample size ( $n=2$ ) on Mouchoir Bank was not representative. The SERs were also grouped for testing according to expected exposure regime with all reefs on the eastern sides of the banks considered “moderately exposed” and those on the western side to be “sheltered.” The three patch reefs all were in exposed locations, hence “windward” versus “leeward” comparisons were not possible for this habitat type.

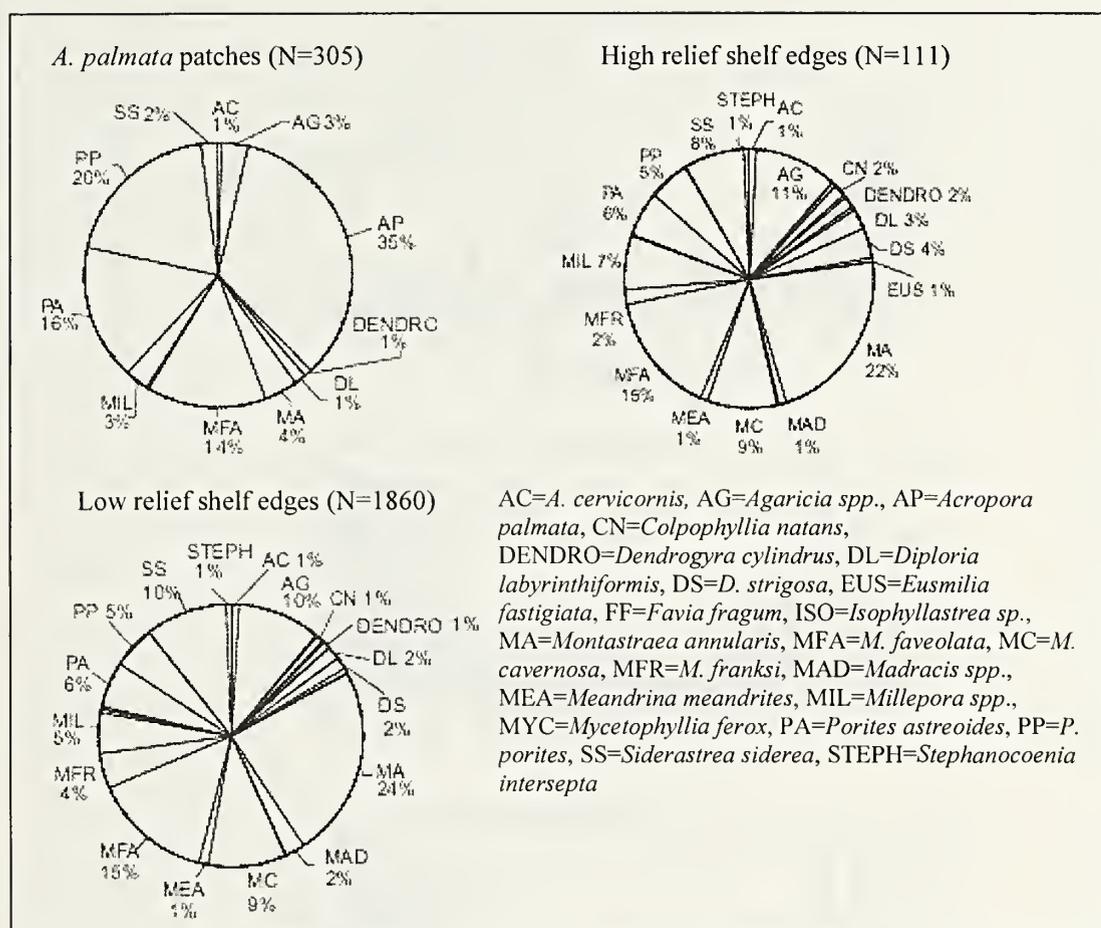
## RESULTS

### Stony Corals

A total of 3,270 corals were surveyed in 289 transects on three banks and around 11 islands in the Turks and Caicos Islands. Live stony coral cover averaged ~18% overall (Table 1). No significant differences were found between the patch reef and the high-relief or low-relief SERs (ANOVA,  $F=1.749$ ,  $p=0.195$ ), despite clear evidence of previous *A. palmata* mortality (numerous large skeletons in the framework) in the former. Differences of exposure were just significant (t-test,  $F=4.3$ ,  $p=0.05$ ), the percentage of live coral cover being higher in the moderately exposed SERs (mean=20.5,  $sd=6.8$ ,  $n=8$ ) than in the sheltered SERs (mean=14.6,  $sd=3.3$ ,  $n=17$ ). The very low coverage (7.5%) seen on one high-relief SER (TC5) was largely due to a local limitation in the amount of suitable habitat as the spurs were dissected into patches each  $\leq 10$  m across and separated from the others by pockets of sand.

Scleractinian growth, particularly on Turks Bank, was most profuse in the area immediately adjacent to the platform margin. In some instances two platform edges were found, a shallower rim at 10-15 m depth and a deeper edge seaward of a small (10-50 m wide) plateau at around 30-35 m (best developed near South Caicos). Stony coral growth was always densest on the outer edge of the shallower shelf. Steep slopes exhibited few scleractinians but in many areas dense populations of black corals, *Cirrhopathes* sp. and *Antipathes* spp., were observed (particularly in TC 7 on Turks Bank). On gentler ( $<40^\circ$  inclination) reef slopes, stony corals were abundant to depths of 50 m on Turks Bank (in TC8), the platy coral facies in places (e.g., TC19) being well over 50% at 25-30m depth.

Of the stony corals that were  $\geq 10$  cm in diameter, *A. palmata* was most common in the patch reefs, while poritids (*Porites porites*, *P. astreoides*) were more abundant here than in the deeper SERs (Fig. 2). Indeed, *P. astreoides* became increasingly common with decreasing water depth (and thus increasing hydrodynamic exposure). *Montastraea annularis* and *M. faveolata* were the most abundant stony corals in deeper water with the *M. annularis* complex constituting about 40% of all the colonies in the SERs (Fig. 2). *A. cervicornis* was present but rare ( $<1\%$  in SERs) on the Caicos and Turks Banks. It is interesting to note that *Dendrogyra cylindrus*, which is generally uncommon in most

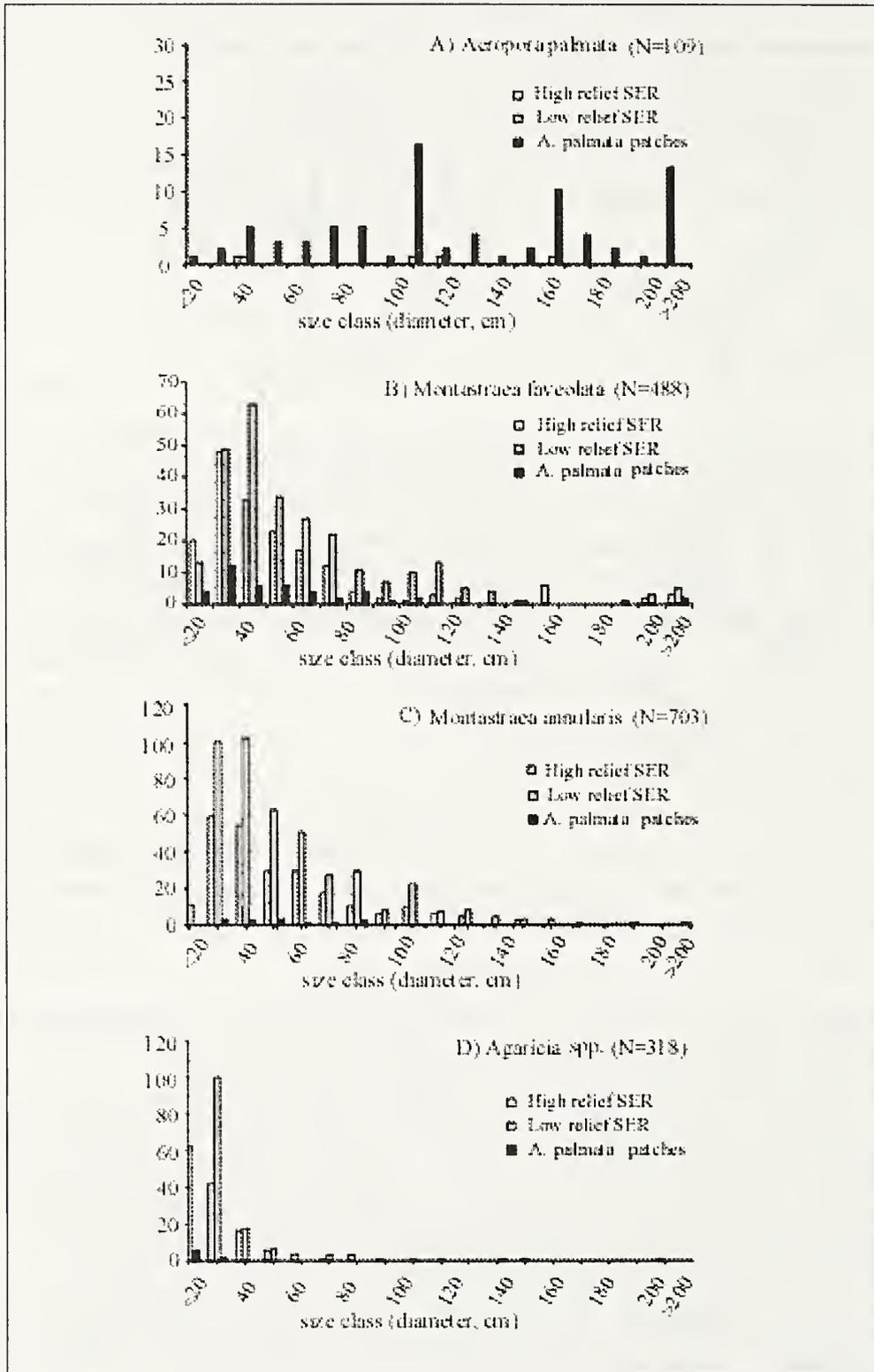


**Figure 2.** Species composition and mean relative abundance of the most abundant stony corals ( $\geq 10$  cm diameter) in *Acropora palmata* patch reefs, low-relief shelf-edge reefs and high-relief shelf-edge reefs in the Turks and Caicos Islands.

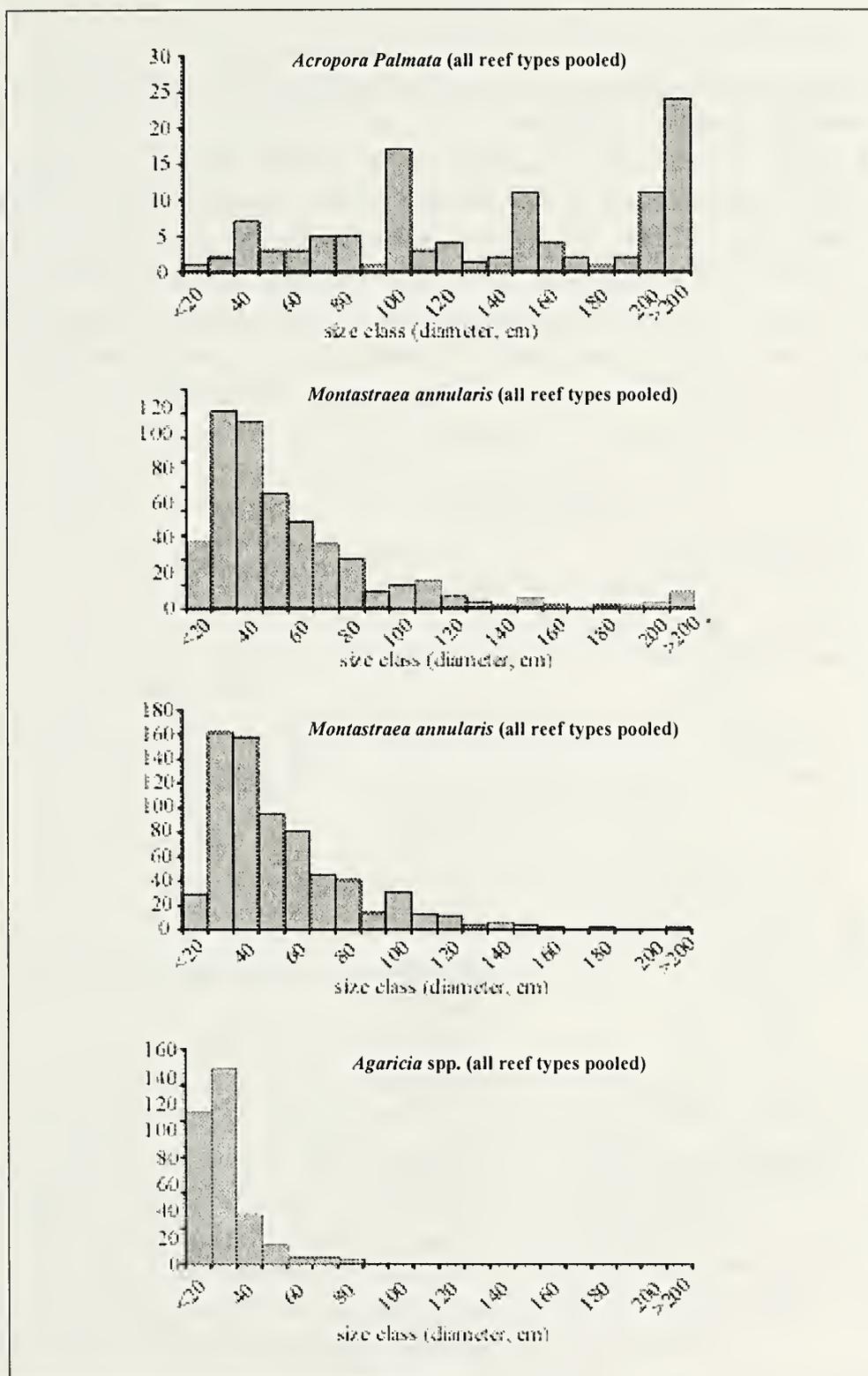
Caribbean areas, obtained counts of 1-2% overall with no apparent preference for any particular depth zone and was particularly conspicuous on Turks Bank.

For colonies  $\geq 10$  cm in diameter, the average maximum diameter (Table 2) ranged between 26 cm in a high-relief SER (TC5) and 103 cm in a patch reef (TC9); their average maximum height varied from 16.5 cm (in TC1, a high-relief SER) to 66 cm (in TC9). Average maximum diameter and height were both significantly higher in the patch reefs than in the deeper reefs (ANOVA for greatest diameter  $F=18.5$ ,  $p<0.001$ ; for greatest height  $F=14.7$ ,  $p<0.001$ ), but no significant differences in size were found between the high-relief and low-relief SERs. Nor were any differences in diameter or height found between moderately exposed and sheltered SERs (t-tests,  $f=0.71$ ,  $p=0.401$  for diameter;  $F=1.91$ ,  $p=0.181$  for height).

Amongst the more common corals, *Acropora palmata* showed a polymodal size distribution which was somewhat skewed towards the larger ( $>100$  cm) size classes (Fig. 3A,B). The size-frequency distributions of *Montastraea annularis* and *M. faveolata* were



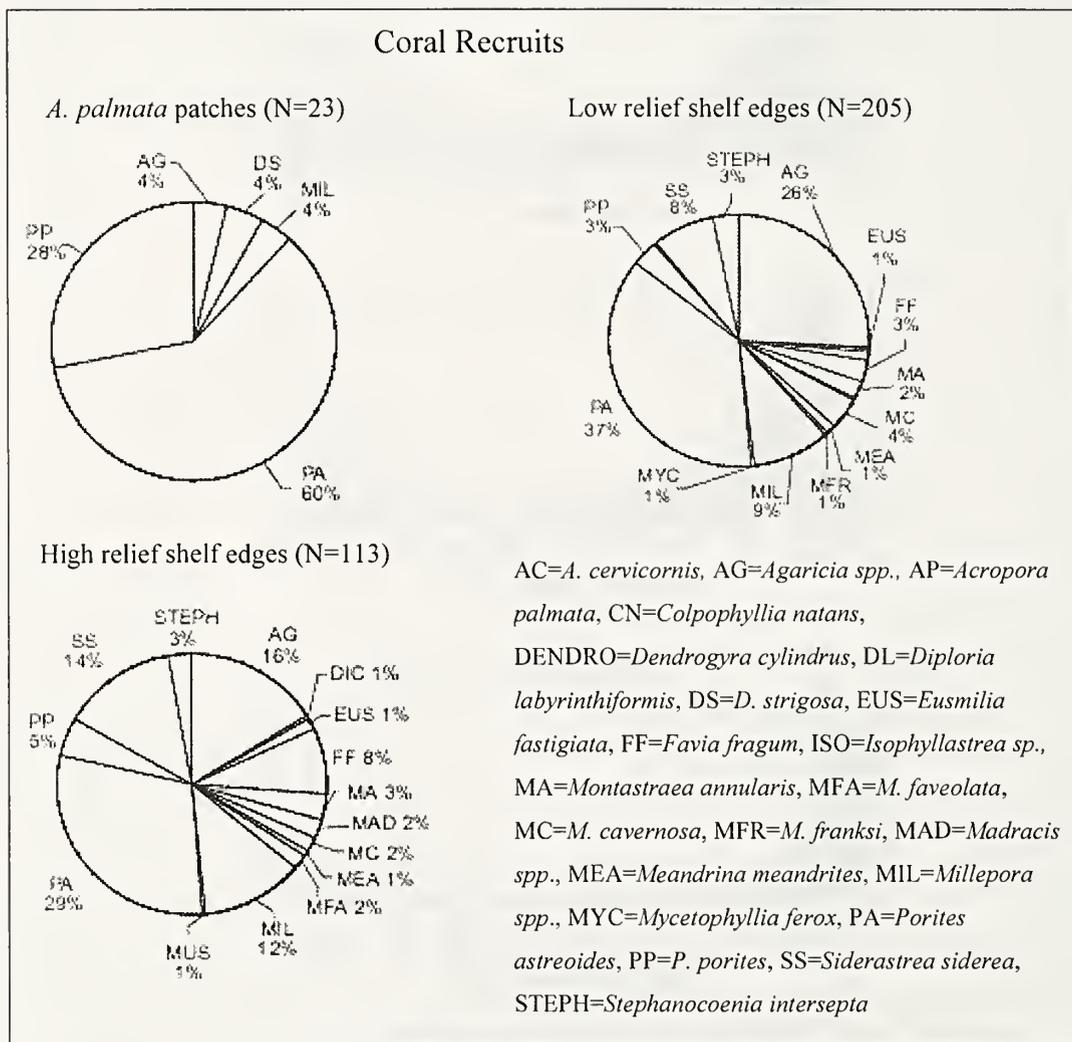
**Figure 3A.** Size-frequency distributions of  $\geq 10$  cm diameter colonies of (A) *Acropora palmata*, (B) *Montastraea faveolata*, (C) *M. annularis*, and (D) *Agaricia* spp. in *Acropora palmata* patch reefs, low-relief shelf-edge reefs and high-relief shelf-edge reefs in the Turks and Caicos Islands.



**Figure 3B.** Size-frequency distributions of  $\geq 10$  cm diameter colonies of (A) *Acropora palmata*, (B) *Montastraea faveolata*, (C) *M. annularis*, and (D) *Agaricia* spp. pooled for all reefs in the Turks and Caicos Islands.

clearly skewed to the “intermediate sizes” (20-70 cm for *M. annularis*, 20-90 cm for *M. faveolata*). Most of the *Agaricia* spp. (primarily *A. agaricites*) were less than 30 cm in maximum diameter. The colonies of *A. cervicornis* in the Caicos and Turks Banks were relatively small (rarely >1 m in diameter).

The density of stony coral recruits (Table 3) in the 1,156 algal quadrats ranged from 0.02/0.0625 m<sup>2</sup> in a patch reef (TC9) to 0.8/0.0625 m<sup>2</sup> in a low-relief SER (TC27). By far the most common (Fig. 4) were *Porites astreoides* followed by *Agaricia* spp. (mostly *A. agaricites*). *Montastraea annularis* and *M. faveolata* were present but in low abundance. No acroporid recruits were encountered during the surveys. Recruit density did not differ significantly between the patch reefs and either the high-relief or the low-relief SERs (ANOVA, F=2.92, p=0.072). There were no differences in recruitment between the moderately exposed and sheltered SERs (t-test, F=0.22, p=0.641).



**Figure 4.** Species composition and mean relative abundance of all stony coral recruits (≤2 cm diameter, excluding species that are small as adults) in *Acropora palmata* patch reefs, low-relief shelf-edge reefs and high-relief shelf-edge reefs in the Turks and Caicos Islands.

## Stony Coral Condition

On average, nearly 5% of the  $\geq 10$  cm diameter stony corals on each reef were diseased. The percentages of diseased stony corals were highest in one site on Mouchoir Bank (17% in TC 12) and in two sites on Caicos Bank (13% in each of TC26 and TC27), while the lowest values were found on Turks Bank (Table 2). The percent of affected colonies did not differ significantly between the patch reefs and either type of SER (ANOVA,  $F=0.223$ ,  $p=0.802$ ). Moreover, there were no differences between the moderately exposed and sheltered SERs (t-test,  $F=0.05$ ,  $p=0.998$ ).

Healthy patches of *Acropora palmata* and *A. cervicornis* were only encountered in Caicos Bank on the southwestern (between Providenciales and West Caicos) and southeastern (at TC15) sides, and in the two patch reefs on the eastern side of Turks Bank (TC9, TC10). Three of the small colonies of *A. cervicornis* in the SER reefs (one each in TC3, TC26 and TC28) had white-band disease, and three of the patch-reef *A. palmata* (two in TC9, one in TC10) exhibited similar characteristics. Only a few cases of black-band disease were encountered. White plague was common, and about 75% of the 130 diseased colonies belonged the *Montastraea annularis* complex (*M. faveolata* 43%, *M. annularis* 27%, *M. franksi* 5%). No bleaching at all was captured in the dataset; neither was any damage by damselfish observed.

Mortality patterns (as a percent of affected upper colony surfaces) were somewhat different between shallow and deeper water (Fig. 5). Values for recent partial-colony mortality (hereafter recent mortality) varied from  $<0.5\%$  in a patch reef (TC10) to  $7.5\%$  in a low-relief SER (TC18) and values of old partial-colony mortality (hereafter old mortality) from  $12.5\%$  in a high-relief SER (TC2) to  $47\%$  in a patch reef (TC9). Percentages of both old mortality and total (recent + old) mortality were slightly higher in the moderately exposed SERs (old mortality mean= $24.9$ ,  $sd=7.9$ ; total mortality= $36.1$ ,  $sd=3.9$ ) than in the sheltered SERs (old mortality= $20.4$ ,  $sd=3.4$ ; total mortality= $31.3$ ,  $sd=3.9$ ).

Recent mortality showed no significant differences between the patch reefs and the SERs (ANOVA,  $F=1.045$ ,  $p=0.367$ ). Both old mortality and total mortality differed significantly between the patch reefs and the high- and low-relief SERs, although the latter did not differ from each other (ANOVA for old mortality  $F=7.33$ ,  $p=0.03$ ; for total mortality  $F=10.2$ ,  $p<0.001$ ). Recent mortality did not differ between the moderately exposed and sheltered SERs (t-test,  $F=4.05$ ,  $p=0.056$ ) whereas significant differences were found in old mortality (t-test,  $F=10.4$ ,  $p=0.004$ ) and total mortality (t-test,  $F=8.5$ ,  $p=0.008$ ), being higher in the sheltered sites than in the moderately exposed SERs.

No examples of stony corals having experienced 100% recent mortality were encountered. Less than 5% were “standing dead” (100% mortality and still in original growth position) (Table 2), except in two of the patch reefs (TC9- $20.5\%$ , TC10- $15.5\%$ ) where much of the reef framework was made of large, long-dead skeletons of *Acropora*

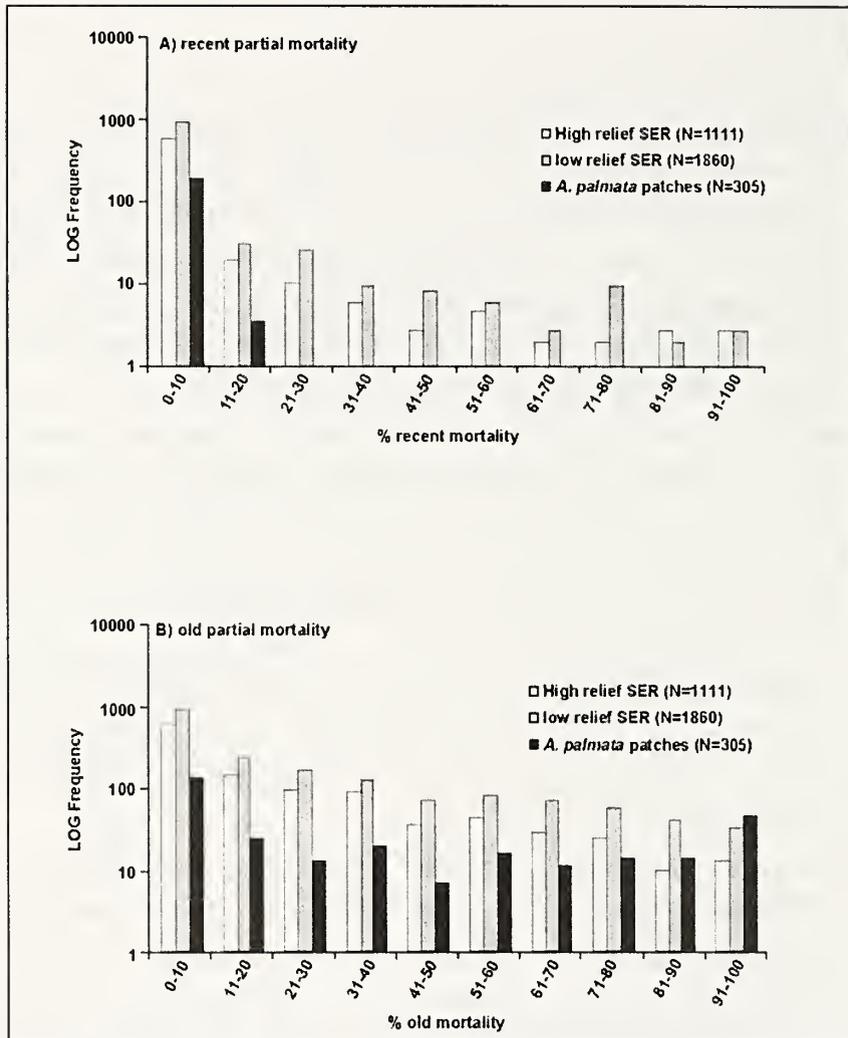


Figure 5. Log-frequency distributions of (A) recent partial colony mortality and (B) old partial colony mortality of all stony corals ( $\geq 10$  cm diameter) in *Acropora palmata* patch reefs, low-relief shelf-edge reefs and high-relief shelf-edge reefs in the Turks and Caicos Islands.

*palmata*. The differences in standing dead corals between the patch reefs and the SERs were significant (ANOVA,  $F=21.77$ ,  $p<0.001$ ). On the Mouchoir Bank, only isolated and badly damaged ridges of *A. palmata* were observed in shallow habitats and in many cases the coral skeletons were heaps of large rubble. Diseases (possibly including aspergillosis) in sea fans and other gorgonians were only observed in very rare instances and did not enter the dataset.

#### Algae and *Diadema antillarum*

Macroalgae constituted the most abundant algal functional group in the algal quadrats in one patch reef (TC15) and were codominant with crustose coralline algae in the other two (Table 3). In the SERs, turf algae predominated in nine, crustose coralline algae were predominant in eight, these two algal groups were approximately equally abundant in

six reefs and two had essentially equal abundances of all three algal groups. Thus, turfs and crustose corallines were about equally common in all but the shallow patch reefs, where macroalgae were comparatively abundant.

Macroalgal heights averaged less than 1 cm in 18 reefs, and from 1-2 cm in seven reefs (Table 3). By far the tallest algae (about 7 cm high) were found in a high-relief SER at Mouchoir Bank (TC11) where clumps of *Turbinaria* were seen to be overgrowing colonies of *Montastraea* spp. Macroalgal indices (absolute abundance of macroalgae x macroalgal height) were highest here and in the patch reefs (particularly TC15).

There were no significant differences in crustose coralline algal abundance between patch reefs and either type of SER (ANOVA,  $F=1.14$ ,  $p=0.335$ ), but macroalgal abundance, turf algal abundance, macroalgal height and macroalgal index differed significantly between the patch reefs and the SERs, which did not differ from each other (ANOVAs for macroalgae,  $F=16.6$ ,  $p<0.001$ ; for turf algae,  $F=5.6$ ,  $p=0.009$ ; for macroalgal height,  $F=3.633$ ,  $p=0.041$ ; for macroalgal index,  $F=8.07$ ,  $p=0.002$ ). The abundance of macroalgae, turf algae and crustose coralline algae did not differ significantly between sheltered and moderately exposed SERs (t-tests,  $F=0$ ,  $p=0.998$  for macroalgae;  $F=0.52$ ,  $p=0.477$  for algal turfs  $F=0.29$ ,  $p=0.594$  for crustose coralline algae). However, macroalgal height and macroalgal index (a proxy for biomass) were significantly greater in moderately exposed SERs (height:  $1.7 \pm 2.2$ cm, index:  $40.6 \pm 79.5$ ) than in the sheltered SERs (height:  $0.6 \pm 0.4$  cm; index:  $10.2 \pm 15.9$ ) (t-test,  $F=7.25$ ,  $p=0.013$  for macroalgal height;  $F=7.54$ ,  $p=0.011$  for macroalgal index). No relationship was noted between either macroalgal height or macroalgal index and the number of stony coral recruits in the quadrats.

No *Diadema antillarum* were found in any of the belt transects, nor elsewhere in the TCI reefs.

## DISCUSSION

Notwithstanding the moderately low total cover by live stony corals, the reef ecosystems in the Turks and Caicos Islands gave the overall impression of being in good condition. The large amounts of standing dead stony corals in two of the patch reefs were clear evidence that the presently low cover of live stony corals reflected at least one previous mortality event. This was not the case on the SERs where the stony corals were in good health with a low prevalence of standing dead colonies (range 0-4%,  $n=25$  reefs), hence their relatively low cover (usually  $<25\%$ ) may be a natural phenomenon. Since all the investigated reefs are within the influence of bank waters, it is possible that the latter exert a strong control over their scleractinian communities. Warmed or cooled bank waters spilling over the reefs may sufficiently stress scleractinians so as to preclude faster growth or higher recruitment. As the Turks and Caicos Islands are also situated within one of the

main hurricane paths, some control may also be exerted by high wave-energy events (e.g., Blanchon and Jones, 1997). The higher cover of live stony corals in the more exposed locations suggests greater influence by bank waters than by waves, which would be expected to produce the reverse pattern; however, it is alternatively possible that stony corals grow faster in windward reefs.

Few healthy patches of *A. palmata* were present either in the surveyed patch reefs or in patches that were visited but not surveyed. (It may be that more could be found on the northern reefs between North Caicos and East Caicos; however, we were not able to survey this area.) For example, patch reefs built by dead or partly dead colonies of *A. palmata* cover extensive areas on the windward (eastern) side of Turks Bank and near Ambergris Cay in southeastern Caicos Bank. These 'long-dead' colonies of *A. palmata* may have been caused by diseases since intact skeletons were common. No information on the timing of death is available; however, it appears that many *Acropora* were already dead when surveyed by Sullivan et al. (1994).

Some of the partially living colonies of *A. palmata* exhibited signs of what appeared to be infection by white-band disease. A measure of uncertainty as to the cause of the *present* die-back remains, however, since local fisheries and nature conservation authorities mentioned occurrences of fishing with chemicals (dish-washing liquid, possibly bleach or gasoline) in these patch reefs. Therefore, what we interpreted as white-band disease might rather have been recent mortality triggered by exposure to toxins. However, we saw no direct evidence of fishing with toxic substances.

*A. cervicornis* was not seen in Mouchoir Bank which is likely to be an artifact of incomplete sampling. On the Caicos and Turks Banks, the small colonies of *A. cervicornis* possibly represented a new generation of recruits or survivors from a previous mortality event. By selective removal of *A. cervicornis*, previous outbreaks of disease could have contributed to the overall low cover of live stony corals. In contrast to the Cayman Islands, where large reef areas are covered by skeletons of *A. cervicornis*, no such skeletal remains were observed in the TCI. Had *A. cervicornis* been more common previously and killed by disease, its skeletons must have completely disappeared due to in-situ erosion or down-slope transport into deepwater, but given the persistence of its skeletons in the fossil record this scenario seems unlikely. The presence of white-band disease in some colonies of *A. cervicornis* is, however, evidence that acroporid diseases were present in the TCI.

The absence of bleached stony corals and of 100% recently dead colonies are indications that no catastrophic mortality events had occurred shortly before our surveys were made. We thus presume that the mass bleaching event of 1998 had only minimal impact in the TCI. Similarly, the low count of standing dead colonies in the SERs indicates a similar absence of major mortality outbreaks in these deeper reef habitats for at least several previous years. The generally low rates of recent mortality (mean=3%) suggest that much of the reef system was in good condition overall. Nevertheless, in a quarter of the examined reefs the  $\geq 10$  cm stony corals exhibited moderately high rates of disease

(6.5-17%). Our surveys may have coincided with an outbreak of white plague that had disproportionately affected colonies of *Montastraea faveolata*.

The paucity of *Acropora palmata* recruits (none encountered in the 1,411 quadrats) and the skewed size distribution of the  $\geq 10$  cm sized corals are suggestive of a pulse-like population replenishment by rare, high-recruitment events. The general skewness of the *Montastraea* distributions towards intermediate sizes may indicate that most of the colonies were of similar ages (resulting from a strong recruitment pulse), and/or be an indication of strong asexual recruitment by fragmentation or, less likely, that they simply do not grow very large in the TCI. Recruitment by small planulating scleractinians like *Porites astreoides* and *Agaricia agaricites* was an order of magnitude higher than by the larger, spatially dominant brooders (e.g., *Acropora*, *Montastraea*) in accord with general experience elsewhere in the wider Caribbean (e.g., Smith, 1992).

That macroalgal height was greater in the windward reefs than in the leeward reefs, notwithstanding their comparatively low hydrodynamic resistance compared to crustose coralline algae and algal turfs, is surprising. The overall scarcity of macroalgae, which accounted for <20% of the benthic algae in 75% (21/28) of the surveyed reefs is encouraging. However, our qualitative impression on Mouchoir Bank was one of unusually strong macroalgal overgrowth over dead stony corals. Whether this is a sign of degradation or a transient temporal phenomenon could only be verified with time-series data.

High cover by macroalgae is generally seen as a sign of deteriorating reef quality (e.g., Steneck, 1994), in part because they restrict the recruitment of stony corals (Rogers et al., 1984). Coral planulae are thought to settle preferentially on crustose coralline algae (Johnson et al., 1991), hence rates of coral recruitment may be lower when crustose corallines are scarce. Although a high abundance of macroalgae did not always correlate with a low recruit count (Table 3), it was associated with some of the lowest recruitment observed on the Mouchoir Bank and near Ambergris Cay. However, it has to be noted that AGRRA sample sizes are not large enough to warrant detailed comparisons among sites or make a credible estimate of recruitment at any given site.

In general we found that the reefs on Mouchoir Bank, which lacks any human population, were in worse condition than in our survey sites on the Turks and Caicos Banks, which was an unexpected result. The part of the Mouchoir Bank within Turks and Caicos Islands territory is difficult to police and protect and is the target of an intense, mostly illegal, fishery. Fishing vessels (reputedly mostly from the Dominican Republic) were reported to sometimes use fishing methods that are destructive to corals. Also, overharvesting of herbivores facilitates the expansion of macroalgae over corals. Remoteness from human population need not necessarily translate into "pristine" and "healthy" reefs. Rather, from our survey it appears that controlled use of reef resources near a moderately dense population may be more sustainable than largely uncontrolled activities in remote locations.

## ACKNOWLEDGMENTS

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Table 1. Site information for AGRRA coral and algal surveys in the Turks and Caicos Islands.

Site Name	Site code	Latitude (° ' N)	Longitude (° ' W)	Survey date	Depth (m)	Benthic transects (#)	≥10 cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)
<i>A. palmata patch reefs</i>								
b/w Round and Gibbs Cay	TC9	21 26.307	71 06.628	Aug 17 99	3.5	13	8	10.5 ± 3.0
E. of S. end of Grand Turk	TC10	21 27.534	71 06.818	Aug 17 99	2.5	13	7	8.0 ± 4.0
Ambergris Cay 1	TC15	--	--	Aug 20 99	6.5	12	9	15.5 ± 6.5
<i>High relief shelf-edge reefs</i>								
Lighthouse Point (anchor)	TC1	21 31.139	71 08.037	Aug 14 99	17.5	10	11	16.0 ± 9.0
North Point (anchor)	TC2	21 32.220	71 06.553	Aug 14 99	16.0	13	10.5	16.5 ± 6.0
Coral Garden	TC3	21 27.493	71 09.301	Aug 14 99	11.5	11	14	25.5 ± 7.0
N. of Salt Cay (anchor)	TC5	21 22.408	71 12.078	Aug 15 99	12.0	10	8.5	7.5 ± 2.0
Mouchoir Bank	TC11	20 59.159	70 47.008	Aug 18 99	22.5	11	11.5	10.5 ± 4.0
The Arch	TC13	21 28.996	71 31.062	Aug 19 99	10.5	10	12.5	20.5 ± 5.0
Ambergris Cay 2	TC16	21 22.359	71 35.949	Aug 20 99	20.0	12	10	12.5 ± 8.5
The Pinnacles (Grace Bay)	TC23	21 48.841	72 11.219	Aug 26 99	10.0	10	12.5	31.0 ± 9.5
Football Field	TC26	21 54.381	72 06.916	Aug 27 99	19.0	8	16.5	28.5 ± 7.5
<i>Low-relief shelf-edge reefs</i>								
West of Little Sand Key	TC4	21 23.658	71 10.088	Aug 15 99	9.5	10	10.5	14.5 ± 6.0
Casey's Wall (anchor, W. of Salt Cay)	TC6	21 18.201	71 13.388	Aug 16 99	11.5	9	14.5	24.5 ± 6.5
Black Forest	TC7	21 28.754	71 09.211	Aug 16 99	11.5	9	13	21.0 ± 4.5
Chief Minister's House	TC8	21 26.443	71 09.294	Aug 16 99	16.5	9	12.5	24.0 ± 8.5
Mouchoir Bank	TC12	21 01.369	70 49.108	Aug 18 99	13.5	11	14	17.0 ± 6.0
Airplane	TC14	21 32.762	71 27.348	Aug 19 99	16.5	10	11.5	13.5 ± 3.5
(No Name)	TC17	21 28.029	71 33.125	Aug 21 99	15.5	11	11	14.5 ± 2.0
Fish Hole	TC18	21 29.072	71 30.629	Aug 21 99	13.0	11	10.5	12.0 ± 3.0
French Cay	TC19	21 29.357	72 13.456	Aug 23 99	14.5	10	0	22.0 ± 6.0
West Sand Spit	TC20	21 23.285	72 08.637	Aug 23 99	13.5	10	12	20.0 ± 10.0
Spanish Anchor (West Caicos wall)	TC21	21 38.739	72 28.473	Aug 24 99	15.5	11	8	12.5 ± 4.0
West Caicos Wall-middle	TC22	21 39.890	72 28.211	Aug 24 99	18.5	9	12	22.0 ± 4.5
Coral Gables (North side of Provo)	TC24	21 49.080	72 11.010	Aug 26 99	11.0	8	12.5	22.0 ± 8.5
Grace Bay (North of TC24)	TC25	21 49.809	72 10.395	Aug 26 99	10.0	10	10.5	14.0 ± 6.0
Grouper Hole	TC27	--	--	Aug 27 99	11.5	8	15	31.0 ± 7.0
Aquarium West	TC28	21 48.518	72 13.542	Aug 27 99	17.5	10	11	12.5 ± 3.5
<b>All sites (mean ± sd)</b>						<b>1.3</b>	<b>11.1</b>	<b>18.0 ± 5.5</b>

Table 2. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 10$  cm diameter) by site in the Turks and Caicos Islands.

Site name/Site code	Stony corals			Partial-colony mortality (%)			Stony corals (%)		
	#	Diameter (cm)	Height (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased
<i>A. palmata</i> patch reefs									
b/w Round and Gibbs Cay/TC9	107	103.0 $\pm$ 95.0	66.0 $\pm$ 74.0	1.0 $\pm$ 7.5	47.0 $\pm$ 40.5	61.5 $\pm$ 36.0	20.5	0	4.5
E. of S. end of Grand Turk/TC10	91	78.5 $\pm$ 77.5	54.0 $\pm$ 57.0	<0.5 $\pm$ 0.5	41.0 $\pm$ 40.0	56.5 $\pm$ 36.0	15.5	0	4.5
Ambergris Cay 1/TC15	107	55.5 $\pm$ 50.5	33.0 $\pm$ 27.5	3.0 $\pm$ 12.5	21.0 $\pm$ 28.0	34.0 $\pm$ 29.5	3	0	7
<i>High relief shelf-edge reefs</i>									
Lighthouse Point (anchor)/TC1	112	29.0 $\pm$ 17.5	16.5 $\pm$ 14.0	0.5 $\pm$ 2.0	18.5 $\pm$ 23.5	29.5 $\pm$ 23.5	0	0	2
North Point (anchor)/TC2	136	37.5 $\pm$ 26.5	20.0 $\pm$ 15.0	1.5 $\pm$ 7.0	12.5 $\pm$ 16.5	22.0 $\pm$ 18.0	0	0	2
Coral Garden/TC3	154	40.5 $\pm$ 24.0	26.0 $\pm$ 16.5	5.5 $\pm$ 18.5	21.5 $\pm$ 24.0	34.0 $\pm$ 27.5	2.5	0	3
N. of Salt Cay (anchor)5/TC5	84	26.0 $\pm$ 14.5	18.5 $\pm$ 16.5	1.0 $\pm$ 6.0	20.0 $\pm$ 27.0	31.5 $\pm$ 28.0	1	0	3
Mouchoir Bank/TC11	126	31.5 $\pm$ 14.0	26.5 $\pm$ 14.5	4.0 $\pm$ 9.5	31.5 $\pm$ 26.5	41.5 $\pm$ 26.0	4	0	7
The Arch/TC13	123	44.0 $\pm$ 28.0	29.0 $\pm$ 20.5	1.5 $\pm$ 4.5	25.0 $\pm$ 25.0	32.0 $\pm$ 24.0	1	0	4
Ambergris Cay 2/TC16	117	32.0 $\pm$ 20.0	21.5 $\pm$ 15.0	4.5 $\pm$ 13.5	15.0 $\pm$ 20.0	27.5 $\pm$ 23.5	0	0	1
The Pinnacles (Grace Bay)/TC23	127	50.0 $\pm$ 41.5	39.0 $\pm$ 38.5	2.5 $\pm$ 9.5	16.5 $\pm$ 19.0	24.5 $\pm$ 21.5	0	0	2
Football Field/TC26	132	48.0 $\pm$ 29.0	28.0 $\pm$ 17.0	3.5 $\pm$ 1.0	18.5 $\pm$ 23.5	30.0 $\pm$ 25.5	1.5	0	13
<i>Low-relief shelf-edge reefs</i>									
West of Little Sand Cay/TC4	105	37.5 $\pm$ 22.0	22.0 $\pm$ 17.5	3.0 $\pm$ 10.5	27.5 $\pm$ 29.0	38.5 $\pm$ 31.5	2	0	1
Casey's Wall (anchor, W. of Salt Cay)/TC6	132	40.5 $\pm$ 6.5	25.5 $\pm$ 19.0	1.0 $\pm$ 4.5	21.5 $\pm$ 26.0	32.0 $\pm$ 25.5	0	0	2
Black Forest/TC7	118	40.5 $\pm$ 9.0	27.5 $\pm$ 18.0	2.0 $\pm$ 11.0	27.0 $\pm$ 24.5	33.5 $\pm$ 25.0	0	0	2
Chief Minister's House/TC8	111	57.0 $\pm$ 44.0	37.0 $\pm$ 28.0	4.9 $\pm$ 13.0	22.5 $\pm$ 24.5	34.0 $\pm$ 26.5	2	0	1
Mouchoir Bank/TC12	153	53.0 $\pm$ 41.5	33.5 $\pm$ 23.5	6.0 $\pm$ 13.0	29.0 $\pm$ 29.5	43.5 $\pm$ 30.5	2.5	0	17
Airplane/TC14	116	38.0 $\pm$ 24.0	27.0 $\pm$ 18.5	2.5 $\pm$ 9.5	34.0 $\pm$ 31.0	45.0 $\pm$ 30.5	3.5	0	4
(No Name)/TC17	123	37.5 $\pm$ 19.0	25.0 $\pm$ 14.0	2.5 $\pm$ 10.0	31.5 $\pm$ 31.0	41.0 $\pm$ 30.0	1.5	0	4
Fish Hole/TC18	117	33.0 $\pm$ 18.5	21.5 $\pm$ 14.0	7.5 $\pm$ 17.5	21.5 $\pm$ 24.5	36.5 $\pm$ 29.0	1.5	0	2
French Cay/TC19	128	43.5 $\pm$ 23.5	30.5 $\pm$ 20.5	2.5 $\pm$ 10.5	18.5 $\pm$ 23.0	31.0 $\pm$ 25.5	0	0	2
West Sand Spit/TC20	118	42.0 $\pm$ 29.5	31.0 $\pm$ 21.5	1.0 $\pm$ 4.0	17.0 $\pm$ 22.0	25.5 $\pm$ 22.5	0	0	3
Spanish Anchor (West Caicos wall)/TC21	90	45.0 $\pm$ 33.0	32.5 $\pm$ 24.5	3.5 $\pm$ 12.5	22.0 $\pm$ 27.0	34.5 $\pm$ 29.0	2	0	9
West Caicos Wall-middle/TC22	110	59.0 $\pm$ 47.5	36.5 $\pm$ 33.5	3.0 $\pm$ 9.5	20.5 $\pm$ 26.0	36.0 $\pm$ 29.5	3.5	0	9
Coral Gables (North side of Provo)/TC24	101	49.0 $\pm$ 33.5	33.0 $\pm$ 28.5	3.5 $\pm$ 12.5	16.5 $\pm$ 22.0	26.0 $\pm$ 25.0	1	0	4
Grace Bay (North of TC24)/TC25	106	39.5 $\pm$ 21.0	30.5 $\pm$ 19.0	1.5 $\pm$ 10.0	22.5 $\pm$ 26.5	35.0 $\pm$ 27.0	0	0	6
Groupcr Hole/TC27	121	51.5 $\pm$ 39.0	37.5 $\pm$ 28.5	2.5 $\pm$ 10.0	19.5 $\pm$ 23.5	29.0 $\pm$ 26.5	1	0	13
Aquarium West/TC28	111	32.5 $\pm$ 15.0	22.0 $\pm$ 14.0	2.5 $\pm$ 10.0	17.0 $\pm$ 23.0	29.0 $\pm$ 26.0	2.5	0	6
<b>All sites (mean <math>\pm</math> sd)</b>	<b>119</b>	<b>45.2 <math>\pm</math> 38.7</b>	<b>30.1 <math>\pm</math> 28.2</b>	<b>2.8 <math>\pm</math> 10.7</b>	<b>23.2 <math>\pm</math> 27.1</b>	<b>34.8 <math>\pm</math> 28.5</b>	<b>2.5</b>	<b>0</b>	<b>5</b>

Table 3. Algal characteristics, density of stony coral recruits and *Diadema antillarum* (mean  $\pm$  standard deviation) by site in the Turks and Caicos Islands.

Site name	Quadrats (#)	Absolute abundance (%)			Macroalgal		Recruits (#/0.0625 m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )
		Macroalgae	Turf algae	Crustose coralline algae	Height (cm)	Index <sup>1</sup>		
<b><i>A. palmata</i> patch reefs</b>								
b/w Round and Gibbs Cay/TC9	55	45.0 $\pm$ 35.0	28.5 $\pm$ 32.0	51.0 $\pm$ 36.5	2.2 $\pm$ 1.2	92	0.02 $\pm$ 0.1	0
E.of S. end of Grand Turk/TC10	60	38.0 $\pm$ 30.5	26.5 $\pm$ 23.0	36.0 $\pm$ 24.5	1.8 $\pm$ 1.2	71	0.2 $\pm$ 0.5	0
Ambergris Cay 1/TC15	60	62.0 $\pm$ 25.5	14.0 $\pm$ 20.5	24.0 $\pm$ 16.0	3.8 $\pm$ 2.8	235	0.2 $\pm$ 0.6	0
<b><i>High relief shelf-edge reefs</i></b>								
Lighthouse Point (anchor)/TC1	45	16.5 $\pm$ 19.0	28.0 $\pm$ 25.5	55.5 $\pm$ 26.5	1.0 $\pm$ 1.0	16	0.3 $\pm$ 0.6	0
North Point (anchor)/TC2	65	26.0 $\pm$ 17.5	22.5 $\pm$ 25.0	51.5 $\pm$ 23.5	2.0 $\pm$ 3.2	49	0.1 $\pm$ 0.3	0
Coral Garden/TC3	55	4.0 $\pm$ 11.5	51.0 $\pm$ 28.5	45.5 $\pm$ 26.5	0.2 $\pm$ 0.6	<1	0.4 $\pm$ 0.6	0
N. of Salt Cay (anchor)/TC5	50	17.0 $\pm$ 19.5	34.0 $\pm$ 23.5	48.5 $\pm$ 27.0	0.8 $\pm$ 0.8	13	0.3 $\pm$ 0.6	0
Mouchoir Bank/TC11	55	33.5 $\pm$ 25.5	32.0 $\pm$ 23.0	34.0 $\pm$ 20.0	7.0 $\pm$ 5.4	233	0.2 $\pm$ 0.5	0
The Arch/TC13	50	2.5 $\pm$ 7.0	52.5 $\pm$ 25.5	45.0 $\pm$ 25.0	0.2 $\pm$ 0.6	<1	0.6 $\pm$ 1.3	0
Ambergris Cay 2/TC16	60	11.5 $\pm$ 16.5	24.0 $\pm$ 22.0	64.5 $\pm$ 27.5	1.6 $\pm$ 3.0	17	0.3 $\pm$ 0.7	0
The Pinnacles (Grace Bay/TC23)	50	0.5 $\pm$ 3.0	58.0 $\pm$ 21.0	41.5 $\pm$ 21.0	0.2 $\pm$ 0.2	<1	0.2 $\pm$ 0.5	0
Football Field/TC26	40	39.0 $\pm$ 35.0	28.5 $\pm$ 29.5	32.5 $\pm$ 27.0	1.2 $\pm$ 1.0	43	0.2 $\pm$ 0.5	0
<b><i>Low-relief shelf-edge reefs</i></b>								
West of Little Sand Key/TC4	50	8.5 $\pm$ 13.5	49.5 $\pm$ 24.5	42.5 $\pm$ 27.0	0.6 $\pm$ 0.8	5	0.5 $\pm$ 0.8	0
Casey's Wall (anchor, W. of Salt Cay)/TC6	45	4.0 $\pm$ 8.0	53.5 $\pm$ 22.0	43.0 $\pm$ 19.0	0.6 $\pm$ 1.0	2	0.2 $\pm$ 0.5	0
Black Forest/TC7	45	2.0 $\pm$ 6.5	79.5 $\pm$ 18.5	18.5 $\pm$ 18.5	0.2 $\pm$ 0.8	.5	0.4 $\pm$ 0.7	0
Chief Minister's House/TC8	41	1.0 $\pm$ 2.5	84.5 $\pm$ 19.5	34.5 $\pm$ 32.5	0.8 $\pm$ 2.4	<1	0.3 $\pm$ 0.5	0
Mouchoir Bank/TC12	55	11.0 $\pm$ 19.5	54.0 $\pm$ 19.0	35.0 $\pm$ 19.0	1.8 $\pm$ 3.6	20	0.1 $\pm$ 0.4	0
Airplane/TC14	47	6.5 $\pm$ 13.5	36.5 $\pm$ 22.0	57.0 $\pm$ 22.0	0.6 $\pm$ 1.0	4	0.5 $\pm$ 0.9	0
(No Name) /TC17	55	1.5 $\pm$ 6.5	46.5 $\pm$ 19.5	52.0 $\pm$ 20.5	0.2 $\pm$ 0.6	<1	0.3 $\pm$ 0.6	0
Fish Hole/TC18	54	2.5 $\pm$ 6.5	49.0 $\pm$ 27.5	48.5 $\pm$ 26.5	0.4 $\pm$ 0.8	<1	0.4 $\pm$ 0.8	0
French Cay/TC19	50	37.5 $\pm$ 25.0	15.5 $\pm$ 21.0	47.5 $\pm$ 28.5	1.6 $\pm$ 1.0	56	0.2 $\pm$ 0.5	0
West Sand Spit/TC20	50	13.5 $\pm$ 22.0	47.5 $\pm$ 30.5	39.0 $\pm$ 26.0	0.8 $\pm$ 1.0	9	0.3 $\pm$ 0.7	0
Spanish Anchor (West Caicos wall) /TC21	49	13.5 $\pm$ 22.5	35.5 $\pm$ 20.0	51.0 $\pm$ 19.5	0.8 $\pm$ 1.0	11	0.4 $\pm$ 0.6	0
West Caicos Wall-middle/TC22	45	14.5 $\pm$ 17.5	45.0 $\pm$ 24.0	40.0 $\pm$ 19.5	0.8 $\pm$ 0.8	12	0.4 $\pm$ 0.7	0
Coral Gables (North side of Provo) /TC24	40	1.0 $\pm$ 4.0	55.5 $\pm$ 23.5	43.0 $\pm$ 22.5	0.2 $\pm$ 0.4	<1	0.6 $\pm$ 0.9	0
Grace Bay (North of TC24)/TC25	50	<0.5 $\pm$ 0.5	62.0 $\pm$ 24.0	38.0 $\pm$ 24.0	<0.1 $\pm$ 0.2	<1	0.5 $\pm$ 0.6	0
Grouper Hole/TC27	40	1.5 $\pm$ 4.0	66.0 $\pm$ 23.5	33.0 $\pm$ 23.5	0.2 $\pm$ 0.6	<1	0.8 $\pm$ 1	0
Aquarium West/TC28	50	8.5 $\pm$ 11.5	39.5 $\pm$ 29.5	52.5 $\pm$ 25.0	0.6 $\pm$ 0.8	5	0.4 $\pm$ 0.7	0
<b>All sites (mean <math>\pm</math> sd)</b>	<b>50.4</b>	<b>18.4<math>\pm</math>28.3</b>	<b>52.2 <math>\pm</math>35.1</b>	<b>55.7 <math>\pm</math> 35.9</b>	<b>1.2<math>\pm</math> 2.4</b>	<b>21.82</b>	<b>0.3<math>\pm</math>0.7</b>	

<sup>1</sup>Macroalgal index = absolute macroalgal abundance x macroalgal height

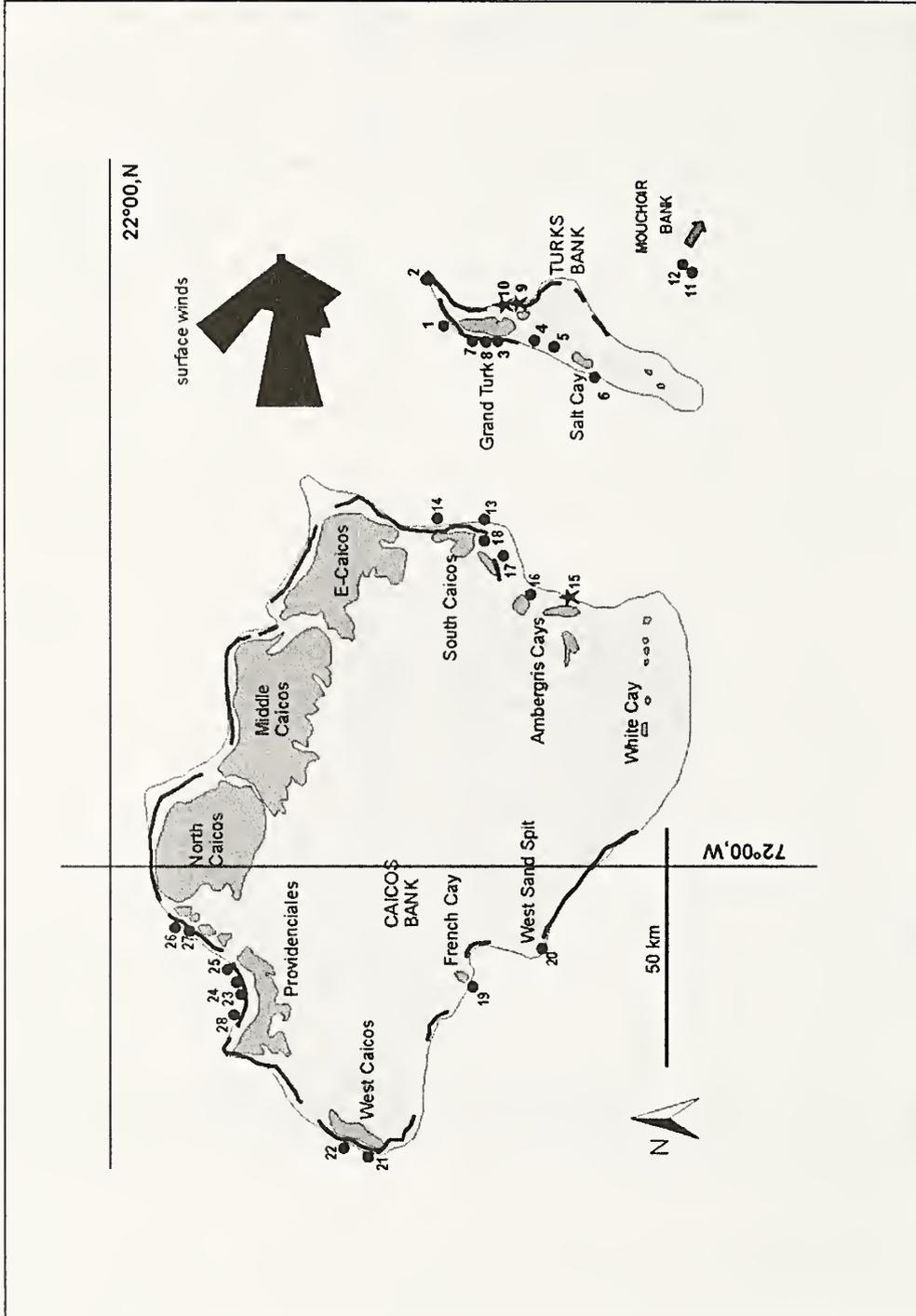


Figure 1. AGRRA survey sites in the Turks and Caicos Islands. See Table 1 for site codes. Wind rose for the southern Bahamas, from R.N. Ginsburg in P.A. Scholle, and N.P. James (1995).

# ASSESSMENT OF THE CORAL REEFS OF THE TURKS AND CAICOS ISLANDS (PART 2: FISH COMMUNITIES)

BY

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## ABSTRACT

Ecologically and commercially significant coral reef fishes were surveyed at 28 sites in the Turks and Caicos Islands during August 1999. Our results constitute the first quantitative census of these fishes and can serve as baseline information for subsequent studies. Their density and size generally were highest off West Caicos and lowest in Mouchoir Bank. Herbivore density overall showed no correlation with macroalgal index (a proxy for biomass) or live stony coral cover, but surgeonfish density was positively correlated with macroalgal index. Species richness of these select fishes was positively correlated with the species richness of stony corals that were  $\geq 10$  cm in diameter. Current fishing pressures overall were low, and the reef-fish communities appeared relatively intact on the Turks and Caicos Banks. However, overfishing and destructive fishing practices have negatively impacted the reef fish communities on Mouchoir Bank.

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## INTRODUCTION

Fish are known to occur at high species diversity and density in undisturbed coral reefs. Although their exact numbers are unknown (Sorokin, 1993), they play important roles in reef ecosystems as herbivores and as top predators (e.g., Hatcher and Larkum, 1983; Hay, 1997). Reef fishes can be sensitive indicators of general reef condition (Pattengill-Semmens and Semmens, this volume). Furthermore, certain large-sized predators such as groupers (serranids) have been historically targeted by commercial as well as sport fishers and they are known to be particularly susceptible to fishing pressure (Russ, 1991; Sluka et al., 1998). Therefore, their relative abundance should serve as a good indicator of fishing pressures.

The Turks and Caicos Islands (TCI), which comprise the southeastern extent of the Bahamian Archipelago, lie between 21° and 22° N latitude and 71° and 72° 30' W longitude (Fig. 1). There are seven inhabited islands, one uninhabited island, and approximately 40 low-lying coral limestone cays distributed on two banks (Turks and Caicos). Part of the entirely submerged Mouchoir Bank, which is located approximately 65 km southeast of Turk Bank between the Mouchoir Passage and the Dominican Republic, also belongs to the TCI. The prevailing easterly trade winds keep sea conditions on the eastern sides of the banks and islands choppy, while the western sides are generally calm. Currents generally run to the north. Over 300 km of coral reef are reported to exist in the TCI (Wells, 1988; Turks and Caicos National Parks Advisory Committee, 1998, report).

The TCI's coral reefs are essential resources for tourism and provide seafood for local consumption. Fisheries, often directly reef-related, are economically important with lobster and conch being the only national exports (Woodley et al., 2000). The tourism industry has developed rapidly during the past decade, particularly for recreational diving. Increased development, population increase, and illegal fishing may in future be challenges for the integrity of the TCI's coral reefs; however, at present they are still remote from intense human impacts.

In 1975, a National Parks Ordinance established 19 marine protected areas with three levels of protection (Woodley et al., 2000) which collectively cover about 700 km<sup>2</sup>. Active management is currently restricted to protected areas near Providenciales, where much of the TCI's populace resides (Woodley et al., 2000). Nevertheless, given its total land area of 505 km<sup>2</sup>, the TCI government has legislated proportionately more protected areas than any other country in the world. In addition, the government has established a self-financing system called the Conservation Fund to ensure long-term sustainability of the national park system.

The major objective of this study was to document fish community composition and structure in the TCI in a snapshot fashion with the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocols. Because of their remote locality, these islands afforded

an opportunity to study relatively “pristine” reef fish communities with few anthropogenic impacts. Relative to many other tropical western Atlantic ecosystems, the reef communities in the TCI are understudied (Chiappone et al., 1996). Our results represent the first systematic census of key fish species and provide baseline data for future monitoring and research in the TCI.

## METHODS

A total of 28 sites (Fig. 1) were studied: two on Mouchoir Bank; 10 on Turks Bank; and 16 in three major localities on the larger Caicos Bank (4 near West Caicos and French Cay, 6 near Providenciales, and 6 near South Caicos). We selected sites that were representative of special interests (i.e., reported to be heavily impacted or of touristic, fisheries and/or conservation value) and strategically accessible (at established dive sites with mooring buoys). An effort was made to space the sites as evenly as possible within all available exposures and reef types, but the small size of the dive boats and prevailing sea conditions restricted surveys to areas of moderate exposure and/or short traveling distance. Although the northern side of Caicos Bank and much of Mouchoir Bank were not exhaustively investigated, a relatively even mix of exposed and sheltered reefs was obtained. Three shallow (<8 m) patch reefs dominated by *Acropora palmata* were sampled (TC9, TC10, TC1); however, a continuous *Acropora palmata* reef crest was not observed in the surveyed areas. The remaining 25 reefs were located on the outer margin of the upper shelf-edge reefs in low-relief (<5 m, n=16) and high-relief (>5 m, n=9) habitats with relatively dense coral growth (Table 1).

The surveys were carried out following the AGRRA Version 2.2 fish protocol (see Appendix One, this volume) by a team of four divers who generally spent between one and two hours underwater at each site. Each belt transect was 30 m x 2 m. Counts of groupers were restricted to species of *Epinephelus* and *Mycteroperca*; scarids (parrotfishes) and haemulids (grunts) less than 5 cm in length were not tallied due to the difficulty in accurate identification. Each diver also performed one 20-minute-long Roving Diver Technique (RTD) survey at every site. One day of consistency training using plastic fish models of varying sizes (10 cm, 20 cm, 40 cm) was completed prior to initiating the surveys.

Field identifications were based on Humann (1994). While we attempted to identify all species as accurately as possible, some errors are likely to have occurred, particularly with the smaller parrotfishes and grunts. Fish in these size classes were not very common, however, and are believed to have introduced little bias into the dataset.

The belt-transect data were used to compare the mean density (as numbers/100 m<sup>2</sup>) and size distribution (in cm) of selected fishes among the five geographic areas using Analysis of Variance (ANOVA). The relationship between key herbivore density and

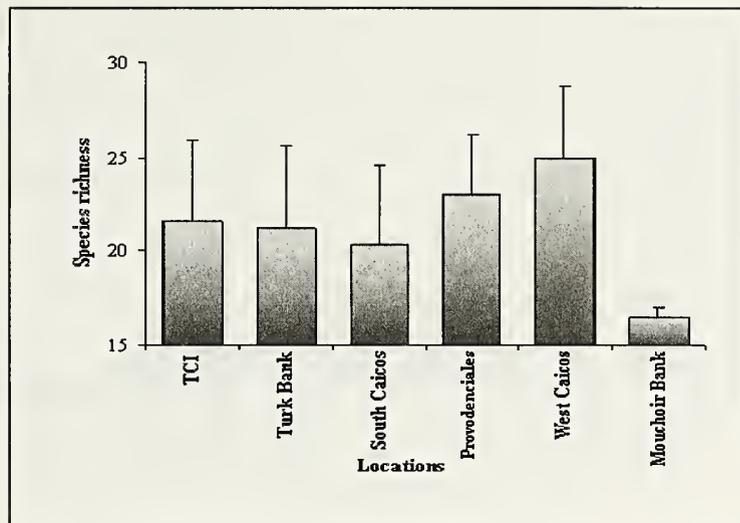
macroalgal index (absolute macroalgal abundance x macroalgal height) was analyzed using Kendall's rank correlation test, as was the relationship between total AGRRA fish diversity and the diversity of stony corals >10 cm in diameter (select stony corals). Relationships between fish density and other benthic habitat variables (live stony coral cover; height, diameter and mortality of select stony corals; depth) were investigated using linear and multiple regressions. (The AGRRA benthic survey data are summarized in Riegl et al., this volume.) Whenever necessary, the data were transformed prior to performing a parametric statistical analysis (e.g., log-transformation was used for fish size, square-root transformation for fish density, and arcsine transformation for percentile data).

## RESULTS

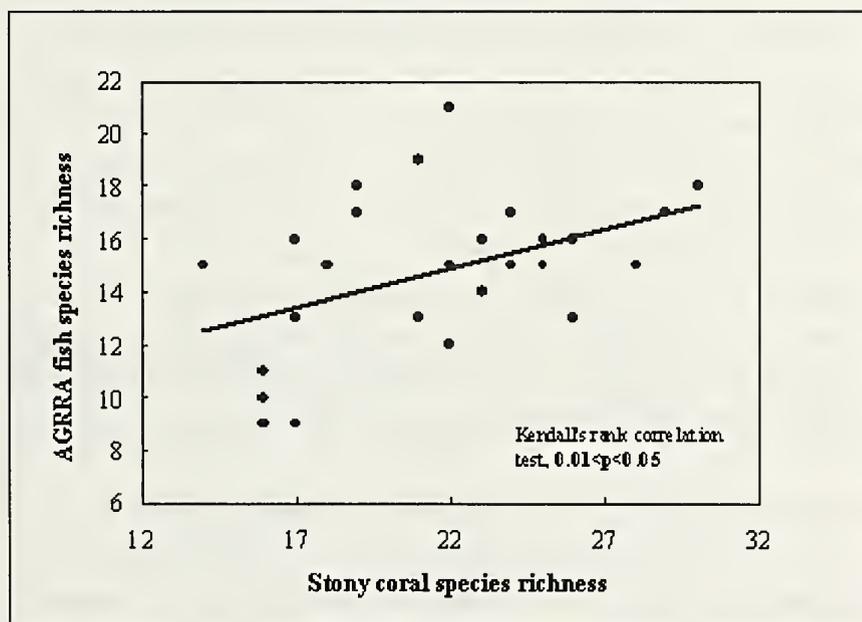
### Belt Transects

*Species richness.* A total of 4,597 fishes belonging to 46 of the species in the AGRRA fish list, including seven each of groupers and parrotfishes, were counted in 279 belt transects (Tables 1, 2). Mean AGRRA species richness/site in the five geographic locations varied from 17 in Mouchoir to 25 in West Caicos, averaging 22 for the TCI as a whole (Fig. 2). The West Caicos Wall (site TC22) had the highest record (30 species) for species richness (Table 1). Pairwise comparisons (Mann-Whitney U-test) revealed that the species richness of the AGRRA fishes in Mouchoir Bank was significantly lower than in Turks Bank ( $p=0.0417$ ), Providenciales ( $p=0.0318$ ), and West Caicos ( $p=0.0301$ ). Moreover, the AGRRA fish species richness in the belt transects was positively correlated (Fig. 3; Kendall's rank correlation,  $0.01 < p < 0.05$ ) with the species richness of the select stony corals assessed concurrently by Riegl et al. (this volume).

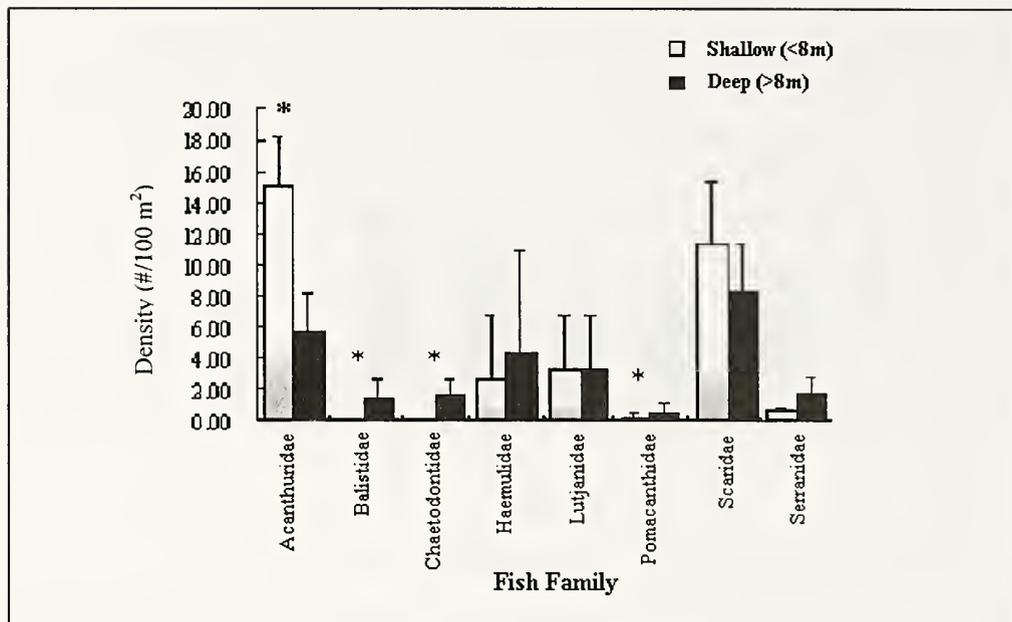
*Density.* Mean pooled fish densities (as numbers/100 m<sup>2</sup>) for select species in eight families are shown for all shallow ( $\leq 8$ m;  $n=3$ ) reefs and all deeper ( $> 8$ m;  $n=25$ ) reefs in Figure 4. Angelfish (pomacanthids), butterflyfish (chaetodontids) and triggerfish (balistids) were significantly more abundant in the deeper reefs (ANOVA,  $p < 0.05$ ), whereas surgeonfish (acanthurid) densities were significantly higher in the shallow reefs (ANOVA,  $p < 0.05$ ).



**Figure 2.** Species richness (mean/site  $\pm$  standard deviation) of AGRRA fishes for the Turks and Caicos Islands (TCI, 4,597 fishes, 28 sites) and five geographic areas: Turk Bank (1,524 fishes, 10 sites), South Caicos (972 fishes, 6 sites), Providenciales (1,198 fishes, 6 sites), West Caicos (691 fishes, 4 sites), Mouchoir Bank (275 fishes, 2 sites).



**Figure 3.** Correlation plot between species richness of all stony corals ( $\geq 10$  cm diameter) and all AGRRA (belt-transect) fishes by site in the Turks and Caicos Islands.

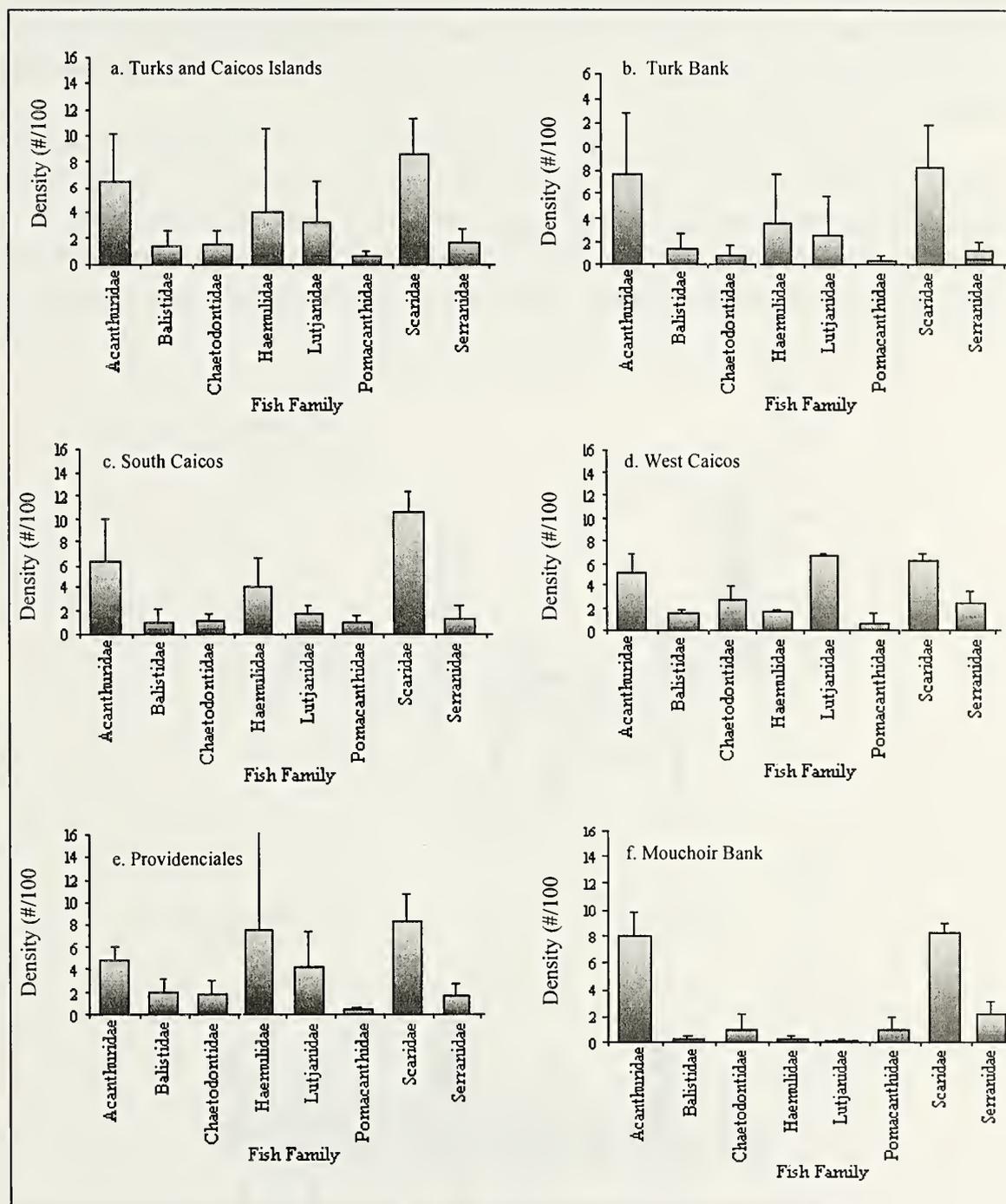


**Figure 4.** Mean fish abundance (no. individuals/100 m<sup>2</sup> ± sd) for AGRRA fishes at <8 m (629 fishes, 3 sites) and >8 m (n=3,968 fishes, 25 sites) in the Turks and Caicos Islands. \* = statistically significant difference (ANOVA, p<0.05) between shallow and deep sites.

The average density of the select fishes in each of these eight families in the TCI as a whole and in the five geographic locations (Table 3A) are shown in Figure 5. A statistically significant difference among geographic areas was found in mean snapper (lutjanid) density (ANOVA, P=0.0234), with the highest value occurring in the reefs around West Caicos. Mean densities of grunts, snappers and triggerfishes were significantly lower (Mann-Whitney U-test, p=0.0405, 0.0246, 0.0405 respectively) in Mouchoir Bank than means for the pooled data in the remaining TCI reefs (Fig. 6A).

The average density for five of these families in each reef is given in Table 4. There were no significant differences in the densities of parrotfish [Kruskal-Wallis (K-W) H Test p=0.16], snappers (K-W H-Test, p=0.91), and grunts (K-W H-Test, p=0.19) among the three types of investigated reefs, but significant differences were found in the densities of surgeonfishes (K-W H Test, p=0.02) and groupers (K-W H Test, p=0.028). Parrotfish and surgeonfish densities were higher in the shallow *Acropora palmata* patch reefs than in the deeper reefs. Groupers had higher densities in high-relief shelf-edge reefs.

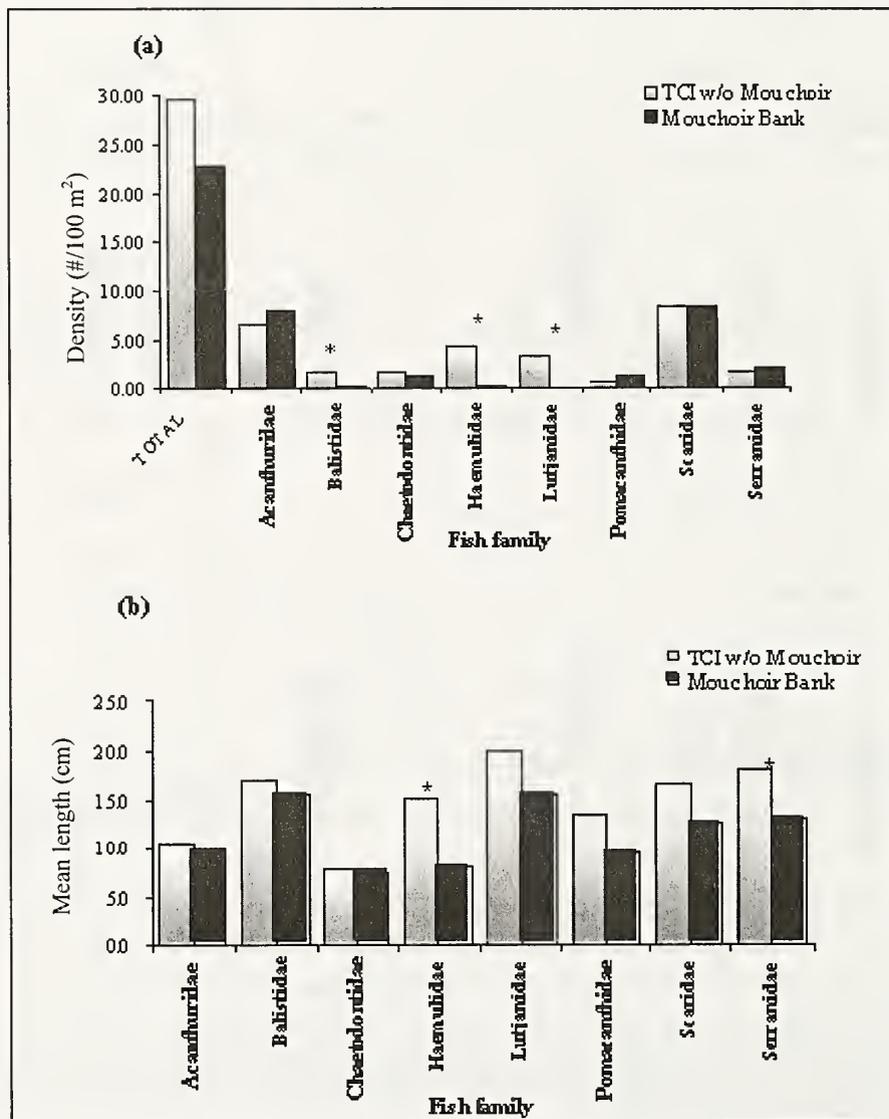
**Size.** The size (as total length) distributions of herbivores (parrotfish ≥5 cm, surgeonfish, the yellowtail damselfish *Microspathodon chrysurus*) and carnivores (snapper, select grouper) at shallow (<8 m) and deeper (>8 m) reefs are summarized in Figure 7. Shallow and deeper reefs differed significantly for both of these important trophic groups (G-test, both p<0.001) with deeper reefs generally having larger fishes. Average sizes were higher in carnivores than herbivores at both depths even though the modes for both groups were in the 11-20 cm-size class in the deeper reefs.



**Figure 5.** Mean fish density (no. individuals/100 m<sup>2</sup> ± sd) for AGRRA fishes in (A) the Turks and Caicos Islands, and five of its geographic areas: (B) Turk Bank, (C) South Caicos, (D) West Caicos, (E) Providenciales and (F) Mouchoir Bank.

The mean size of grunts was significantly different among the five locations (ANOVA,  $p=0.0312$ ) with the largest being found in West Caicos (Table 3B) where the largest fishes in three other families (angelfish, butterflyfish and triggerfish) were also found. The average sizes of grunts and groupers were significantly smaller on the Mouchoir Bank (Mann-Whitney U-test,  $p=0.0477$ ,  $0.0445$ , respectively) when compared with the pooled data from the other TCI locations (Fig. 6B).

*Relationships.* Neither key herbivore (parrotfish  $\geq 5$  cm and surgeonfish) nor surveyed parrotfish density was correlated with macroalgal index (Fig. 8 A, B; Kendall's rank correlation tests,  $p>0.1$ ). However, surgeonfish density was positively correlated with macroalgal index (Fig. 8C; Kendall's test,  $0.01 < P < 0.05$ ). A multiple regression analysis revealed that the size (diameter and height) of the select stony corals had the

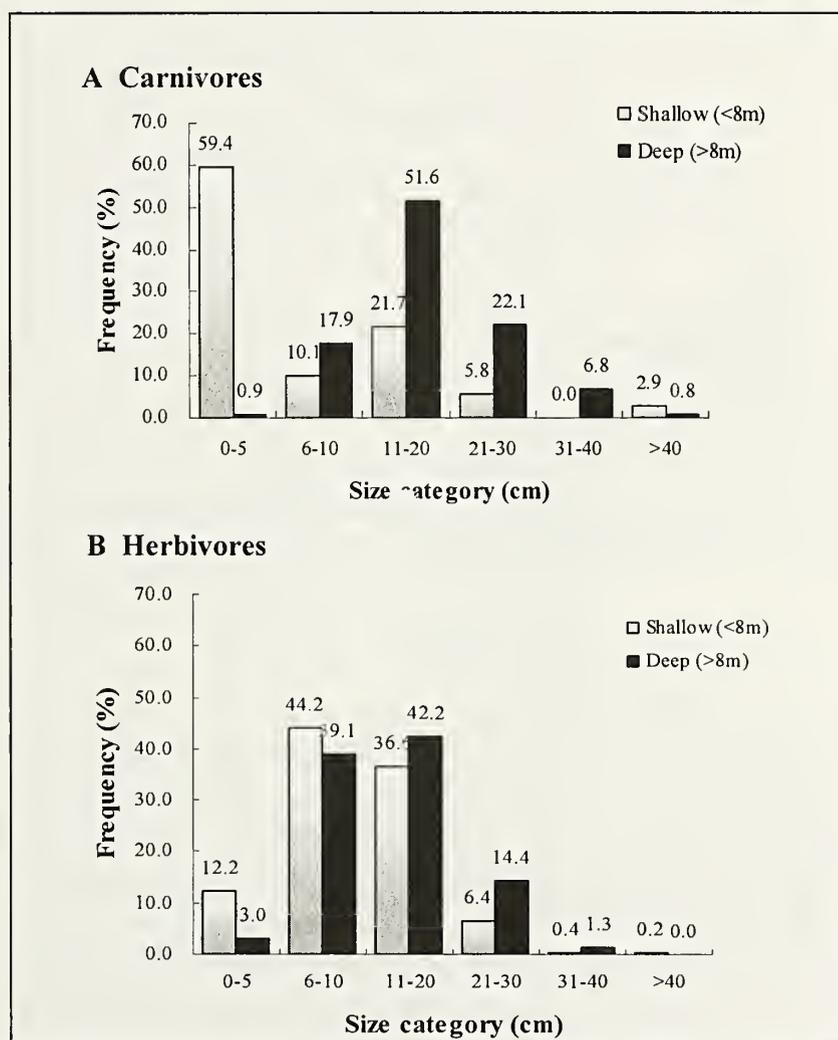


**Figure 6.** Mean (A) density (no. individuals/100 m<sup>2</sup>) and (B) length (cm) for AGRRA fishes in Mouchoir Bank (2 sites) versus the remainder of the Turks and Caicos Islands (26 sites). \* = statistically significant difference.

highest relative importance for fish density (Kruskal's index, diameter=0.0772, height=0.0502), although the overall result was not statistically significant (multiple regression,  $p>0.1$ ). Neither the average density of the key herbivores nor the average total fish density showed a significant relationship with percent live stony coral cover (linear regression,  $p>0.1$ , Fig. 9A, B).

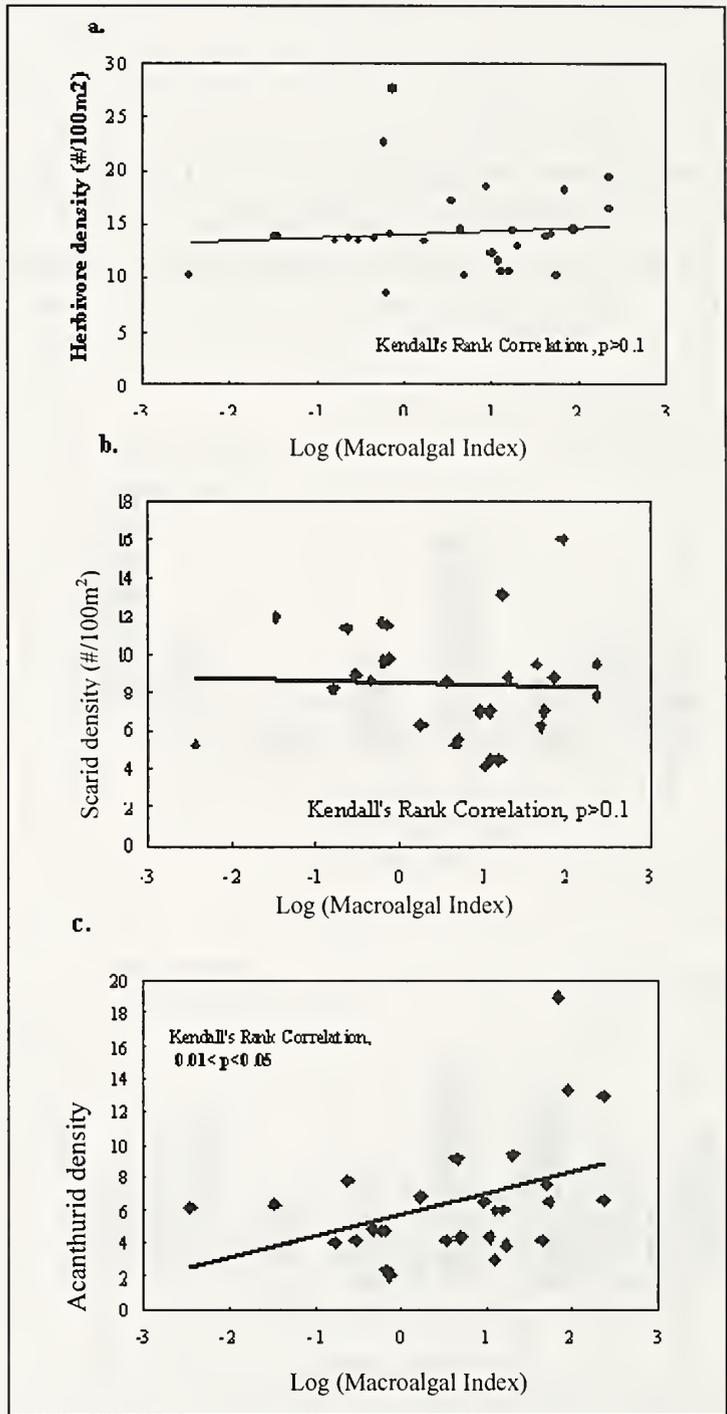
### Roving Diver Surveys

The total number of species observed during the RTD surveys was 120. The 25 most commonly seen species, along with their sighting frequency in the REEF (2000) database, are shown in Table 5. The highest total species richness occurred in West Caicos and Providenciales, while the lowest was in Mouchoir Bank. Parrotfishes were the

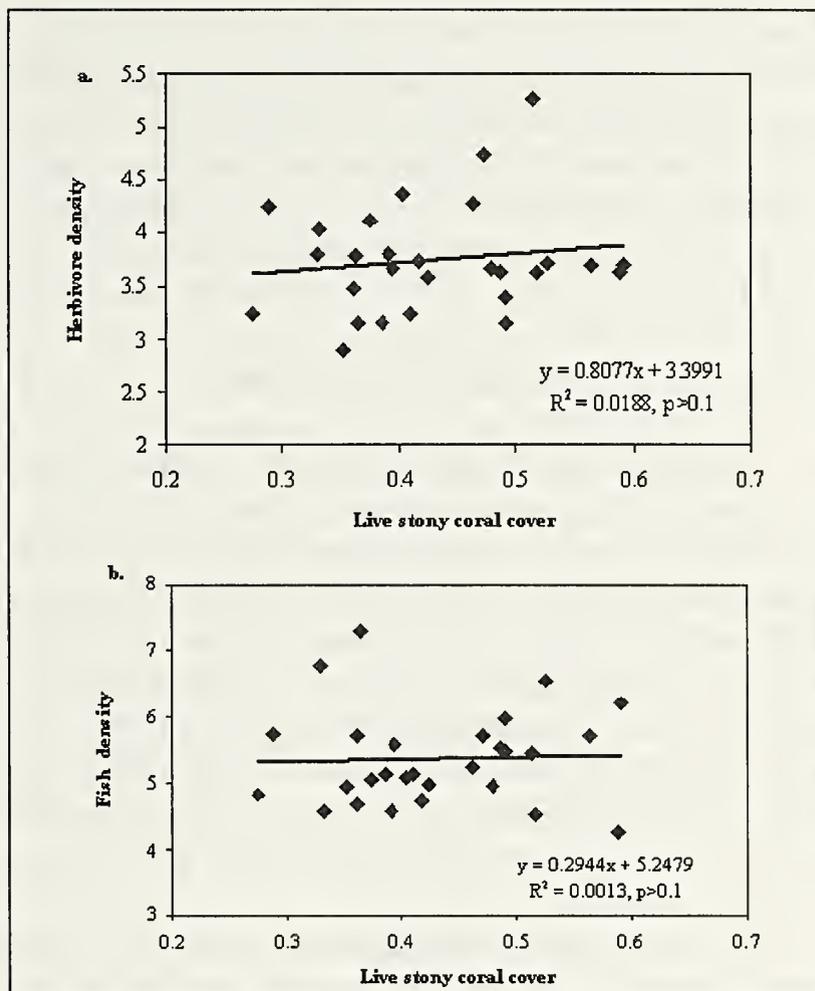


**Figure 7.** Size frequency distribution of (A) carnivores (lutjanids, select serranids) and (B) herbivores (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) at <8m (629 fishes, 3 sites) and >8 m (3,968 fishes, 25 sites) in the Turks and Caicos Islands.

most common of the AGRRA fishes seen in the Turks and Caicos. Butterflyfish and angelfish were the least commonly encountered, with the exception of Mouchoir Bank where grunts and snappers were seen least frequently.



**Figure 8.** Correlation plots between mean log macroalgal index and mean density (no. individuals/100 m<sup>2</sup>) of (A) herbivores (acanthurids, scarids  $\geq 5$  cm), (B) scarids  $\geq 5$  cm, and (C) acanthurids by site in the Turks and Caicos Islands.



**Figure 9.** Relationship between mean live stony coral cover (%; arcsin-squareroot transformed) and mean density (no. individuals/100 m<sup>2</sup>, squareroot transformed) of (A) Key AGRRA herbivores and (A) all AGRRA fishes, by site in the Turks and Caicos Islands.

## DISCUSSION

The average grouper density in the TCI (1.62/100 m<sup>2</sup>, Table 3A) was higher than Sluka et al. (1998) had found in the Florida Keys (0.01-0.13/100 m<sup>2</sup>) and the Exuma Cays, Bahamas (0.10-0.90/100 m<sup>2</sup>). Although comparisons must be made with caution due to the different methodologies used, it would appear that current fishing pressures on the Turks and Caicos Banks are relatively low. That groupers had higher densities in high-relief shelf-edge reefs may reflect some correlation between density and habitat complexity. The large numbers of grunts, particularly in Aquarium West (TC 28), were mainly small individuals and likely to have resulted from active recruitment.

The average length of groupers in the TCI was 17.5 cm. While smaller species like coney (*Epinephelus fulvus*) and graysby (*E. cruentatus*) were the most abundant, larger species such as Nassau (*Epinephelus striatus*) and black (*Mycteroperca bonaci*)

groupers were also recorded in the belt transects in eight reefs. Only two percent (6/271) of the groupers in the belt transects were larger than 40 cm in length. Equally large groupers were also sighted during several dives outside our belt transect areas. Since much of the area, whether protected or unprotected, is subject to some spearfishing, it is likely that large groupers are either so shy or so rare as to be seldom seen (Woodley et al., 2000, refer to “uncontrolled fishing in the marine parks” of the TCI.)

Illegal poaching appeared to be prevalent on Mouchoir Bank, probably due to its remote locality. We observed the use of spearguns. We deduced possible remnants of destructive fishing practices, such as chemicals and dynamite, from the peculiar way that stony corals had lost tissues and the way rubble from broken colonies was strewn on the reefs. Supporting information of such abuses was verbally provided by local fishery patrol officers. Notwithstanding our small sample size [two sites (TC11 and TC12), 20 transects], significantly lower species richness for the AGRRA fishes (Table 1), plus significantly lower densities for three of these families and smaller sizes for two families, of the AGRRA fishes (Fig. 6A,B), were recorded here. In summary, the Mouchoir Bank reefs appeared to suffer from overfishing and destructive fishing practices.

The number of species found during the RDT surveys (120) was much lower than the total number (319 as of May 2000) reported from the TCI in the Reef Environmental Education Foundation (REEF) database in 2002. No new species records resulted from our surveys; however, with a more extended sampling effort, it is likely that some new records could have been obtained.

The AGRRA belt transects are focused on ecologically and commercially important fishes to enhance the efficiency of the surveys; therefore, the data only provide species richness and diversity within a selected group of species. The RDT census provides supplementary data on the entire fish assemblage. The number of species recorded can differ substantially depending on the total number of divers, their level of experience, and the search time as seen in the REEF survey reports (at [www.reef.org](http://www.reef.org)) which clearly show different results between expert and novice divers. Since the objectives of the AGRRA project is to obtain a regional perspective on the status of Caribbean coral reefs based on a standardized methodology, the results of the RDT surveys should be treated with some caution.

The overall results from the 1999 AGRRA survey revealed that the TCI reef fish communities are relatively diverse and healthy, except on Mouchoir Bank. Although reefs accessible from Providenciales (i.e., Providenciales, West Caicos) receive the largest number of snorkelers and scuba divers, currently their fish communities appear to be in relatively good condition. This may be due to the fact that most of the area is designated as marine parks in which fishing is prohibited. Similar designations are either lacking or not well enforced in South Caicos and Turk, which historically have had higher levels of fishing activity. That fish density and diversity showed nearly significant positive relationships with the size (as diameter and height, proxies for substratum

architectural complexity) of the  $\geq 10$  cm stony corals is in general agreement with previous findings by Ault and Johnson (1998a, 1998b) and Chabanet et al. (1997). The relationships found in the TCI between the diversity of these stony corals and the diversity of ecologically and commercially important fishes are rarely addressed in the literature (but see Chabanet et al., 1997). Hence the AGRRA project will provide an excellent opportunity for continued exploration of this topic.

### ACKNOWLEDGMENTS

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Table 2. List of the AGRRA fish species observed in the belt transects in the Turks and Caicos Islands.

Scientific name	Common name	Scientific name	Common name
<b>Acanthuridae</b>	<b>Surgeonfish</b>	<b>Pomacanthidae</b>	<b>Angelfish</b>
<i>Acanthurus bahianus</i>	Ocean Surgeonfish	<i>Holocanthus ciliaris</i>	Queen
<i>Acanthurus chirurgus</i>	Doctorfish	<i>Holocanthus tricolor</i>	Rock Beauty
<i>Acanthurus coeruleus</i>	Blue Tang	<i>Pomacanthus arcuatus</i>	Gray
<b>Balistidae</b>	<b>Leatherjacket</b>	<i>Pomacanthus paru</i>	French
<i>Balistes vetula</i>	Queen Triggerfish	<b>Scaridae</b>	<b>Parrotfish</b>
<i>Cantherines macrocerus</i>	Whitespotted Filefish	<i>Scarus croicensis</i>	Striped
<i>Cantherines pullus</i>	Orangespotted Filefish	<i>Scarus taeniopterus</i>	Princess
<i>Melichthys niger</i>	Black Durgon	<i>Scarus vetula</i>	Queen
<b>Chaetodontidae</b>	<b>Butterflyfish</b>	<i>Sparisoma aurofrenatum</i>	Redband
<i>Chaetodon aculeatus</i>	LongSnout	<i>Sparisoma chrysopterus</i>	Redtail
<i>Chaetodon capistratus</i>	Four-eye	<i>Sparisoma rubripinne</i>	Redfin
<i>Chaetodon ocellatus</i>	Spotfin	<i>Sparisoma viride</i>	Stoplight
<i>Chaetodon striatus</i>	Banded	<b>Serranidae</b>	<b>Grouper</b>
<b>Haemulidae</b>	<b>Grunt</b>	<i>Epinephelus adscensionis</i>	Rock Hind
<i>Anisotremus virginicus</i>	Porkfish	<i>Epinephelus cruentatus</i>	Graysby
<i>Haemulon album</i>	Margate (White)	<i>Epinephelus fulvus</i>	Coney
<i>Haemulon aurolineatum</i>	Tomate	<i>Epinephelus guttatus</i>	Red Hind
<i>Haemulon carbonarium</i>	Caesar	<i>Epinephelus striatus</i>	Nassau
<i>Haemulon flavolineatum</i>	French	<i>Mycteroperca bonaci</i>	Black
<i>Haemulon parra</i>	Sailors Choice	<i>Mycteroperca tigris</i>	Tiger
<i>Haemulon plumieri</i>	White	<b>Other</b>	<b>Other</b>
<i>Haemulon sciurus</i>	Bluestriped	<i>Bodianthus rufus</i>	Spanish Hogfish
<b>Lutjanidae</b>	<b>Snapper</b>	<i>Caranx ruber</i>	Bar Jack
<i>Lutjanus analis</i>	Mutton	<i>Lachnolaimus maximus</i>	Hogfish
<i>Lutjanus apodus</i>	Schoolmaster	<i>Microspathodon chrysurus</i>	Yellowtail Damselfish
<i>Lutjanus mahogani</i>	Mahogany	<i>Sphyræna barracuda</i>	Great Barracuda
<i>Ocyurus chrysurus</i>	Yellowtail		

Table 3A. Mean density of AGRRA fishes, by area in the Turks and Caicos Islands.

Area	Density (#/ 100 m <sup>2</sup> )									
	Herbivores					Carnivores				
	Acanthuridae	Scaridae (≥5 cm)	Haemulidae (≥5 cm)	Lutjanidae	Serranidae <sup>1</sup>	Balistidae	Chaetodontidae	Pomacanthidae	Other <sup>2</sup>	
Providenciales	4.86	8.28	7.50	4.28	1.69	1.97	1.86	0.56	2.28	
W Caicos	5.08	6.29	1.63	6.75	2.42	1.58	2.75	0.58	1.71	
S Caicos	6.36	10.66	3.99	1.74	1.38	1.05	1.15	0.90	1.48	
Grand Turk	7.70	8.20	3.60	2.50	1.30	1.40	1.00	0.40	2.60	
Mouchoir	8.00	8.33	0.17	0.08	2.17	0.17	1.00	1.08	1.92	
<b>All TCI</b>	<b>6.45</b>	<b>8.47</b>	<b>4.00</b>	<b>3.16</b>	<b>1.62</b>	<b>1.37</b>	<b>1.47</b>	<b>0.61</b>	<b>2.11</b>	

<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.<sup>2</sup>Other = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus* and *Sphyræna barracuda*.

Table 3B. Mean size of AGRRA fishes, by area in the Turks and Caicos Islands.

Area	Length (cm)									
	Herbivores					Carnivores				
	Acanthuridae	Scaridae (≥5 cm)	Haemulidae (≥5 cm)	Lutjanidae	Serranidae <sup>1</sup>	Balistidae	Chaetodontidae	Pomacanthidae		
Providenciales	10.92	18.28	14.46	23.37	16.20	18.15	7.65	11.46		
W Caicos	10.68	17.17	19.46	22.72	16.54	18.42	8.11	18.08		
S Caicos	10.62	16.58	13.92	18.19	20.03	16.80	6.92	13.81		
Grand Turk	10.08	15.30	14.40	17.00	18.50	15.90	8.00	12.90		
Mouchoir	10.04	12.66	8.00	15.50	13.02	15.50	7.45	9.49		
<b>All TCI</b>	<b>10.46</b>	<b>16.27</b>	<b>14.83</b>	<b>19.65</b>	<b>17.65</b>	<b>17.05</b>	<b>7.66</b>	<b>13.12</b>		

<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.

Table 4. Mean density of select AGRRA fishes, by site in the Turks and Caicos Islands.

Site Code	Density (# /100 m <sup>2</sup> )				
	Herbivores		Carnivores		
	Acanthuridae	Scaridae (>5 cm)	Haemulidae (>5 cm)	Lutjanidae	Serranidae <sup>1</sup>
<i>A. palmata</i> patch reefs					
TC9	13.33	16.00	7.50	7.33	0.50
TC10	18.83	8.83	0.33	0.83	0.83
TC15	13.00	9.50	0.17	1.33	0.67
<b>Mean ±sd</b>	<b>15±3.2</b>	<b>11.4±3.9</b>	<b>2.7±4.2</b>	<b>3.2±3.6</b>	<b>0.5±0.83</b>
<i>High-relief</i> shelf-edge reefs					
TC1	6.00	4.50	4.00	0.17	1.00
TC2	7.64	6.25	1.53	0.56	0.97
TC3	2.33	11.50	13.00	8.00	0.83
TC5	6.00	4.50	4.00	0.17	1.00
TC11	6.67	7.83	0	0	1.50
TC13	4.67	11.67	5.17	2.50	1.83
TC16	3.83	13.17	6.50	2.33	1.33
TC23	6.33	12.00	3.50	7.00	2.17
TC26	4.17	9.50	4.33	3.83	0.67
<b>Mean ±sd</b>	<b>5.2±1.6</b>	<b>8.9±3.3</b>	<b>4.7±3.6</b>	<b>2.7±3.01</b>	<b>0.7±2.2</b>
<i>Low relief</i> shelf-edge reefs					
TC4	9.17	5.33	0.33	0	1.33
TC6	6.83	6.33	1.83	0	2.33
TC7	4.83	8.67	0.50	1.67	1.67
TC8	2.00	9.83	3.17	6.50	2.33
TC12	9.33	8.83	0.33	0.17	2.83
TC14	4.17	8.61	6.94	1.94	0.28
TC17	7.83	11.33	1.67	1.50	3.33
TC18	4.67	9.67	3.50	0.83	0.83
TC19	6.50	7.00	0.67	4.67	0.67
TC20	6.50	7.00	1.00	3.67	2.50
TC21	4.33	4.17	1.00	4.83	5.00
TC22	3.00	7.00	3.83	13.83	1.50
TC24	4.00	8.17	3.67	8.50	1.67
TC25	6.17	5.33	0	3.67	3.50
TC27	4.17	9.00	0.17	1.33	1.17
TC28	4.33	5.67	33.33	1.33	1.00
<b>Mean ±sd</b>	<b>5.4±2.1</b>	<b>7.62±1.9</b>	<b>3.8±8.1</b>	<b>3.4±3.7</b>	<b>0.3±5</b>

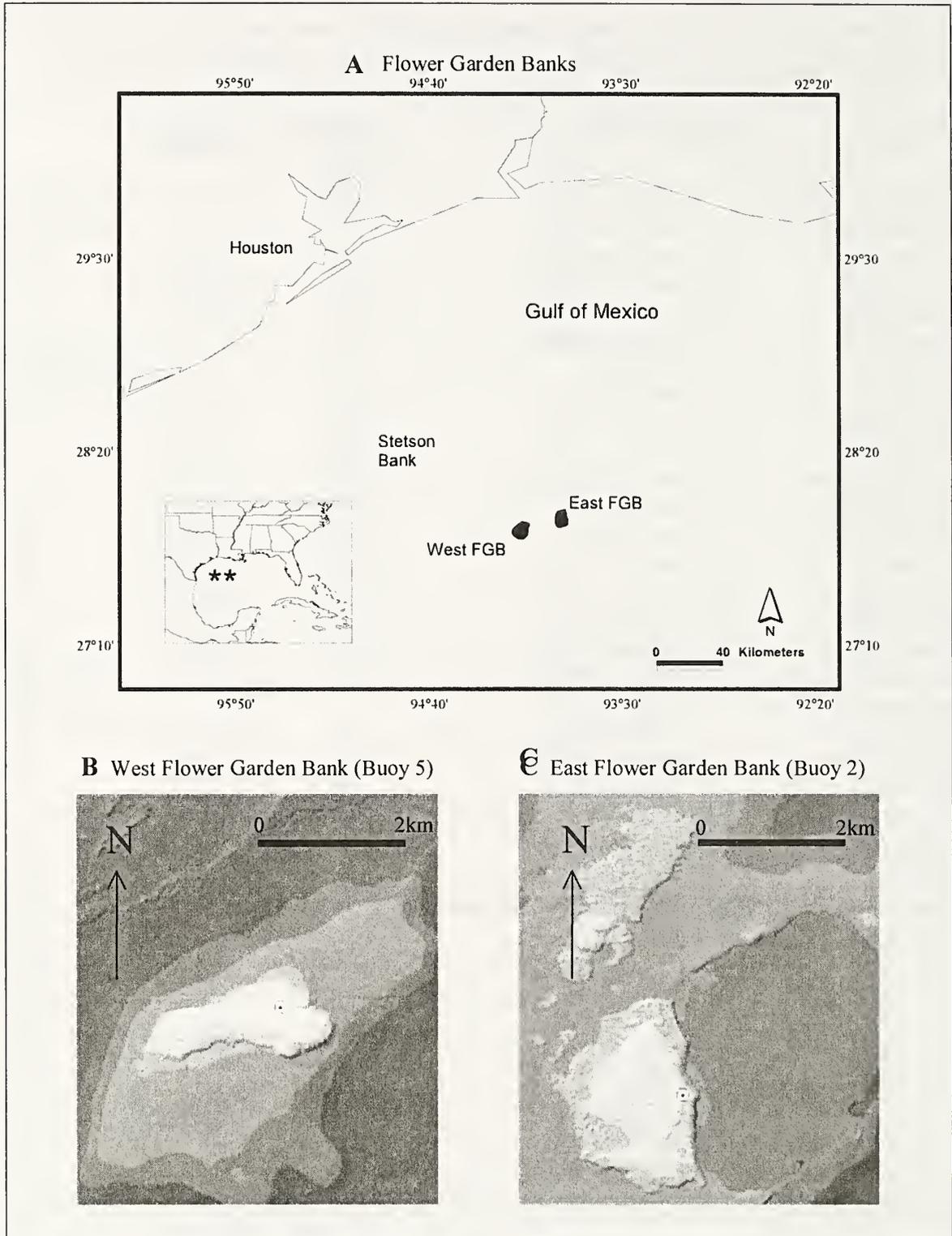
<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.

Table 5. Twenty-five most frequently sighted fish species during roving diver surveys in the Turks and Caicos Islands, with mean densities for species recorded in belt transects.

Scientific name	Common name	Sighting frequency <sup>1</sup> (%)	Density (#/100m <sup>2</sup> )
<i>Thalassoma bifasciatum</i>	Bluehead Wrasse	97	-
<i>Halichoeres garnoti</i>	Yellowhead Wrasse	97	-
<i>Scarus taeniopterus</i>	Princess Parrotfish	97	1.32
<i>Sparisoma viride</i>	Stoplight Parrotfish	94	2.08
<i>Chromis cyanea</i>	Blue Chromis	91	-
<i>Stegastes partitus</i>	Bicolor Damselfish	88	-
<i>Gramma loreto</i>	Fairy Basslet	88	-
<i>Acanthurus coeruleus</i>	Blue Tang	88	3.65
<i>Scarus croicensis</i>	Striped Parrotfish	88	1.84
<i>Epinephelus fulvus</i>	Coney	88	1.27
<i>Chaetodon capistratus</i>	Foureye Butterflyfish	88	1.38
<i>Holocanthus tricolor</i>	Rock Beauty	85	0.59
<i>Haemulon flavolineatum</i>	French Grunt	82	2.83
<i>Caranx ruber</i>	Bar Jack	82	0.80
<i>Acanthurus bahianus</i>	Ocean Surgeonfish	79	2.72
<i>Melichthys niger</i>	Black Durgon	76	1.67
<i>Ocyurus chrysurus</i>	Yellowtail Snapper	73	2.13
<i>Sparisoma aurofrenatum</i>	Redband Parrotfish	70	1.32
<i>Pseudupeneus maculatus</i>	Spotted Goatfish	67	-
<i>Bodianthus rufus</i>	Spanish Hogfish	67	0.33
<i>Chaetodon striatus</i>	Banded Butterflyfish	67	0.43
<i>Haemulon sciurus</i>	Bluestriped Grunt	64	1.53
<i>Aulostomus maculatus</i>	Trumpetfish	64	-
<i>Stegastes planifrons</i>	Threespot Damselfish	61	-
<i>Scarus vetula</i>	Queen Parrotfish	61	1.89

<sup>1</sup>Sighting frequency for the AGRRA surveys from the REEF database at

<http://www.reef.org/data/twa/surveys/index.shtml>



**Figure 1.** (A) AGRRA survey sites at the Flower Garden Banks. Location of (B) Buoy 5, West Flower Garden, (C) Buoy 2, East Flower Garden Bank. □=Buoy

# A RAPID ASSESSMENT OF THE FLOWER GARDEN BANKS NATIONAL MARINE SANCTUARY (STONY CORALS, ALGAE AND FISHES)

BY

CHRISTY V. PATTENGILL-SEMMENS<sup>1</sup> AND STEPHEN R. GITTINGS<sup>2</sup>

## ABSTRACT

Benthic and fish communities at one site on each of the East and West Flower Garden Banks were assessed using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocol in August 1999. Surveys at 20-28 m revealed high coral cover (~50%) dominated by large (mean diameter 81-93 cm) healthy corals with total (recent + old) partial-colony mortality values averaging 13%. Turf algae were the dominant algal functional group and the mean relative abundance of macroalgae was <10%. The large abundance, size and biomass of many fishes reflected the low fishing pressure on the Banks. Due to their near-pristine condition, the Flower Garden Banks data will prove to be a valuable component in the rapid assessment database and its resulting determination of regional reef condition.

## INTRODUCTION

The East and West Flower Garden Banks (EFG and WFG), located 175 km southeast of Galveston, Texas on the edge of the U.S. Gulf Coast continental shelf (Fig. 1A), were created by the uplift of Jurassic-age salt domes. Rising about 100 m above the surrounding depths to within 18 m of the surface, the Flower Garden Banks (FGB) support the northernmost coral reefs in the continental United States. The low diversity (about 21 species), high cover and large size of stony corals, and the low abundance of benthic macroalgae relative to most Caribbean reefs have been well documented (Bright and Pequegnat, 1974; Bright et al., 1974; Boland et al., 1983; Dennis, 1985; Rezak et al., 1985; Dennis and Bright, 1988; Gittings et al., 1993). The FGB are also less susceptible to bleaching than most coral reefs because they are fairly deep; the 1998 regional mass bleaching event did not occur here. The FGB are dominated by massive boulder corals (particularly *Montastraea* spp. and *Diploria* spp.) but lack acroporids and shallow-water gorgonians.

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Fish diversity is also comparatively low (approximately 260 species) but abundances are high (Pattengill, 1998). Fish families and groups that are notably absent, or only represented by one or few species, include grunts (Haemulidae), snappers (Lutjanidae), and hamlets (*Hypoplectrus* sp.). The Banks are year-round habitat for manta rays (*Manta birostris* and *Mobula hypostoma*) and whale sharks (*Rhincodon typus*) and serve as a winter habitat for several species of schooling sharks including scalloped hammerheads (*Sphyrna lewini*), silky sharks (*Carcharhinus falciformis*), and spotted eagle rays (*Aetobatus narinari*) (Childs, 2001).

A unique feature of the EFG is a brine seep at 72 m. The seep features a brine pool with a chemosynthetic bacterial assemblage that is known to be a significant exporter of carbon to the deeper parts of the Bank (Rezak et al., 1985). The seep also plays a significant role in the physiographic structure of the EFG due to the dissolution of salt which results in local faulting and subsidence.

The EFG and WFG are managed and protected by the National Oceanic and Atmospheric Administration (NOAA)'s National Marine Sanctuary System and the Department of Interior's Minerals Management Service. Together with Stetson Bank, they make up the Flower Garden Banks National Marine Sanctuary (FGBNMS). Anthropogenic impacts on the Flower Gardens are relatively low, mainly due to their distance from land. Very little fishing pressure exists on the reefs (see <http://www.sanctuaries.nos.noaa.gov/oms/pdfs/FlowerGardensRegs.pdf>; Subpart L or 15CFR922.122); spearfishing and techniques that disturb benthic habitats, including trawls, traps, and bottom long lines, are prohibited but hook-and-line fishing is permitted. The main, local source of human-induced disturbance is mechanical damage due to anchors, seismic cables, and occasionally long-line fishing tackle. Scuba diving is allowed on the Banks and moorings have been installed to reduce anchor damage. A long-term monitoring program has been in place for approximately 20 years. Historical biotic changes have been attributed primarily to regional or global events such as the die-off of *Diadema antillarum* and periodic coral bleaching.

In August 1999, an AGRRA expedition to the FGB was coordinated by NOAA and the Reef Environmental Education Foundation (REEF) in conjunction with the annual FGBNMS REEF Field Survey for volunteer fish monitoring. The Banks were chosen for the survey because "end-member" reefs, including those that are unusually luxuriant, are particularly relevant to the AGRRA program (R. Ginsburg, personal communication).

## METHODS

The survey team included seven scientists from the National Marine Sanctuary Program and three REEF experts. NOAA's two long-term monitoring sites at EFG Buoy 2 and WFG Buoy 5 (Fig. 1B, C) each of which is considered representative of the high-diversity stony coral zone on the Bank (Gittings et al., 1992), were strategically chosen for the surveys. The AGRRA protocol version 2.1 (see Appendix One, this volume) was used with the following modifications. Benthic and fish surveys were conducted simultaneously with divers dispersed over a radius of about 150 m around the moored

boat. Five divers conducted coral transects and algal quadrats during any given dive. Coral sizes were measured to the closest 5 cm, and any sediment deposited in the algal quadrats was gently removed by hand waving before estimating the relative abundance of crustose coralline algae. Cyanobacteria were counted as turf algae in 64 quadrats and as macroalgae in three quadrats at EFG. Three divers conducted the fish-belt transects and roving diver (RDT) surveys. Counts of serranids (groupers) were restricted to species of *Epinephelus* and *Mycteroperca*; scarids (parrotfishes) and haemulids (grunts) less than 5 cm in length were not tallied. All surveys were made between depths of 20 and 28m during daylight hours (7:00 a.m.-6:00 p.m.).

The benthic- and fish-transect data were entered into a custom Excel spreadsheet provided by the AGRRA organizing committee. REEF provided the RDT data in the American Standard Code for Information Interchange (ASCII) format. The percent coral cover, percent mortality, mean colony size, incidence of disease and bleaching, and relative algal abundance were calculated and compared between Banks using a t-test. A macroalgal index (calculated as macroalgal index = % relative macroalgal abundance x canopy height—a proxy for macroalgal biomass) was also used as a comparison metric. Using the fish transects as replicates, the average density (#/100 m<sup>2</sup>) and size (cm) of each fish species and family recorded were calculated for each site. The average density and size of each species and family were compared between Banks using a t-test. Transect data were also used to calculate biomass for each fish species using standardized conversion equations (Appendix Two, this volume). The RDT survey data provided a species list, frequency of occurrence, and relative abundance data for each Bank. Percent sighting frequency (%SF) for each species was the percentage of dives in which the species was recorded. An estimate of abundance (den) was calculated as: Density score =  $((n_S \times 1) + (n_F \times 2) + (n_M \times 3) + (n_A \times 4)) / (n_S + n_F + n_M + n_A)$ , where  $n_S$ ,  $n_F$ ,  $n_M$ , and  $n_A$  represented the number of times each abundance category (single, few, many, abundant) was assigned for a given species.

## RESULTS

Due to the minimum depth of the Banks, only one depth interval at each site was surveyed. At the WFG, 135 coral colonies and 55 algal quadrats were examined along 11 benthic transects and 11 RDT fish surveys and 12 fish-belt transects were performed. At the EFG, 160 coral colonies and 67 algal quadrats were surveyed on 14 transects and 15 RDT fish surveys and 12 fish-belt transects were conducted.

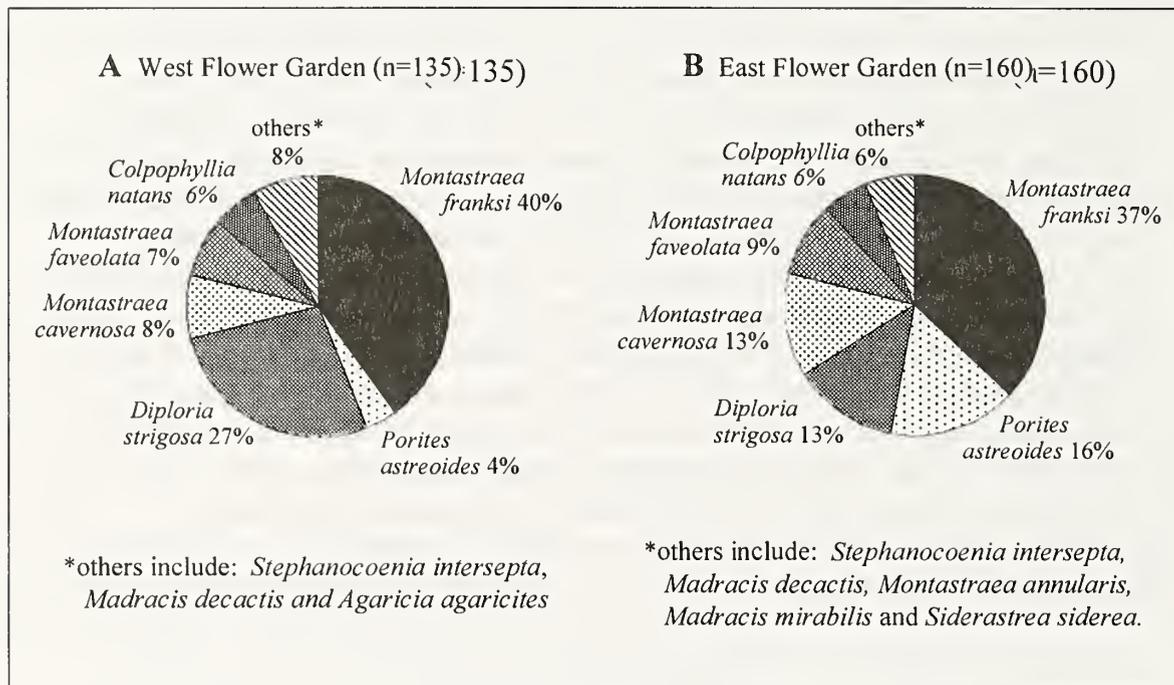
### Stony Corals

Live stony coral cover averaged 54% and 49% at the WFG and EFG, respectively (Table 1). Nine species of “large” stony corals (with diameters  $\geq 25$  cm) were recorded within the transects at the WFG and 11 species at the EFG. Dominant species at the WFG (Fig. 2A) were *Montastraea franksi* (40% of all colonies counted), *Diploria strigosa* (27%), *Montastraea cavernosa* (8%), and *Montastraea faveolata* (7%). The dominant

corals at the EFG (Fig. 2B) were *Montastraea franksi* (37%), *Porites astreoides* (16%), *Montastraea cavernosa* (13%), and *Diploria strigosa* (13%).

The average diameter of the large corals (Table 2) was significantly greater (t-test;  $P < 0.05$ ) at the WFG than at the EFG (93 versus 81 cm, respectively). None of the transected corals showed any signs of disease. Parrotfish bites were reported in about 8% of all colonies surveyed. Pale bleaching was noted in some colonies (~6-16%). Overall, for colony surfaces recent mortality averaged 2%, mean old mortality was 11.5% and total partial mortality averaged 13%.

The density of coral recruits found in the quadrats was 2.3 and 1.7 per  $m^2$  at the WFG and EFG, respectively (Table 3). The recruits were *P. astreoides* and *Agaricia agaricites*, in equal proportions.



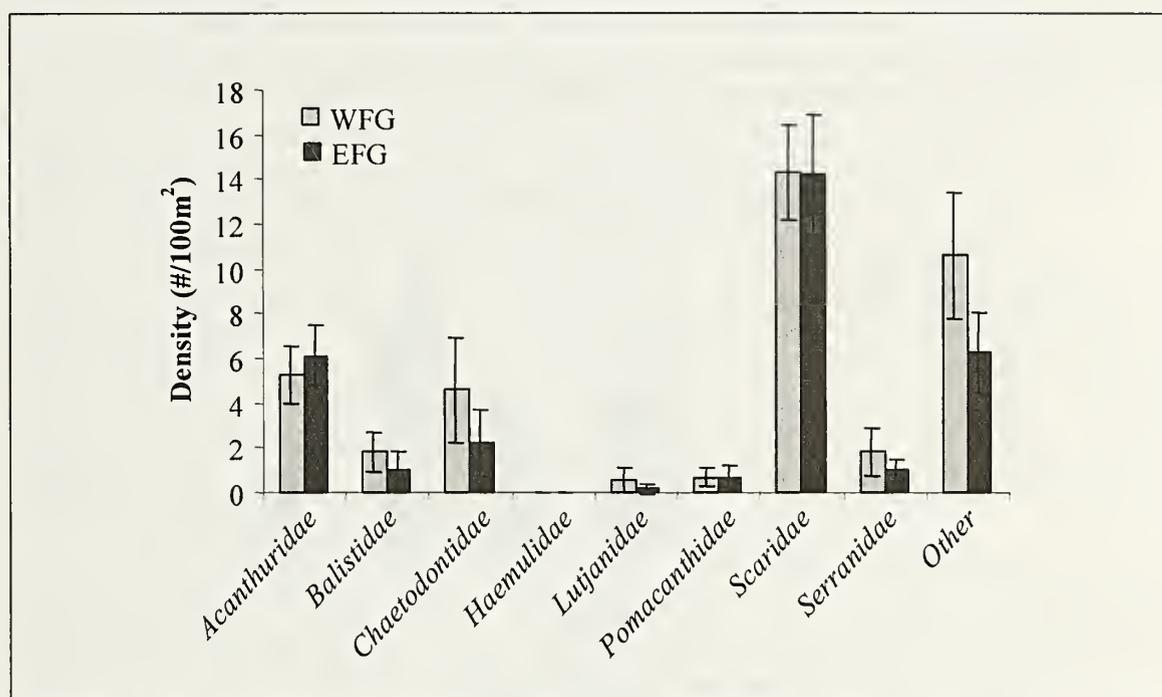
**Figure 2.** Species composition and mean relative abundance of all stony corals ( $\geq 25$  cm diameter) at (A) West Flower Garden, (B) East Flower Garden Bank reefs.

## Algae

Relative macroalgal abundance was very low ( $< 10\%$ ) on both Banks (Table 3). Average macroalgal height was 1.0 cm, yielding macroalgal indices of 8.0 and 10.0 at the WFG and EFG, respectively. A mat cyanobacterium at the EFG that was common in the algal quadrats, on the sand flats, and on several coral heads was most likely responsible for the significantly higher (80% versus 72%) relative abundance values of turf algae at the EFG (t-test;  $p < 0.05$ ). Very few individuals of *Diadema antillarum* were sighted within the transects (0.9 and 1.4/100  $m^2$  at the WFG and EFG, respectively).

## Fish

The average density of most families (Fig. 3) and species of fishes surveyed in the belt transects was similar at the EFG and WFG. Parrotfish were the most abundant fish recorded in the transects. The densities of graysby (*Epinephelus cruentatus*) and Spanish hogfish (*Bodianus rufus*) at the WFG were approximately twice those at the EFG (t-test;  $p < 0.05$ ). The density of reef butterflyfish (*Chaetodon sedentarius*) was also two and a half times greater at the WFG; however, this difference was not significant (t-test;  $p = 0.059$ ). Grunts, several species of parrotfish and snapper, hogfish (*Lachnolaimus maximus*), and gray angelfish (*Pomacanthus arcuatus*) were absent at both Banks, a distinguishing characteristic of the FGB's fish assemblage (Pattengill, 1998).

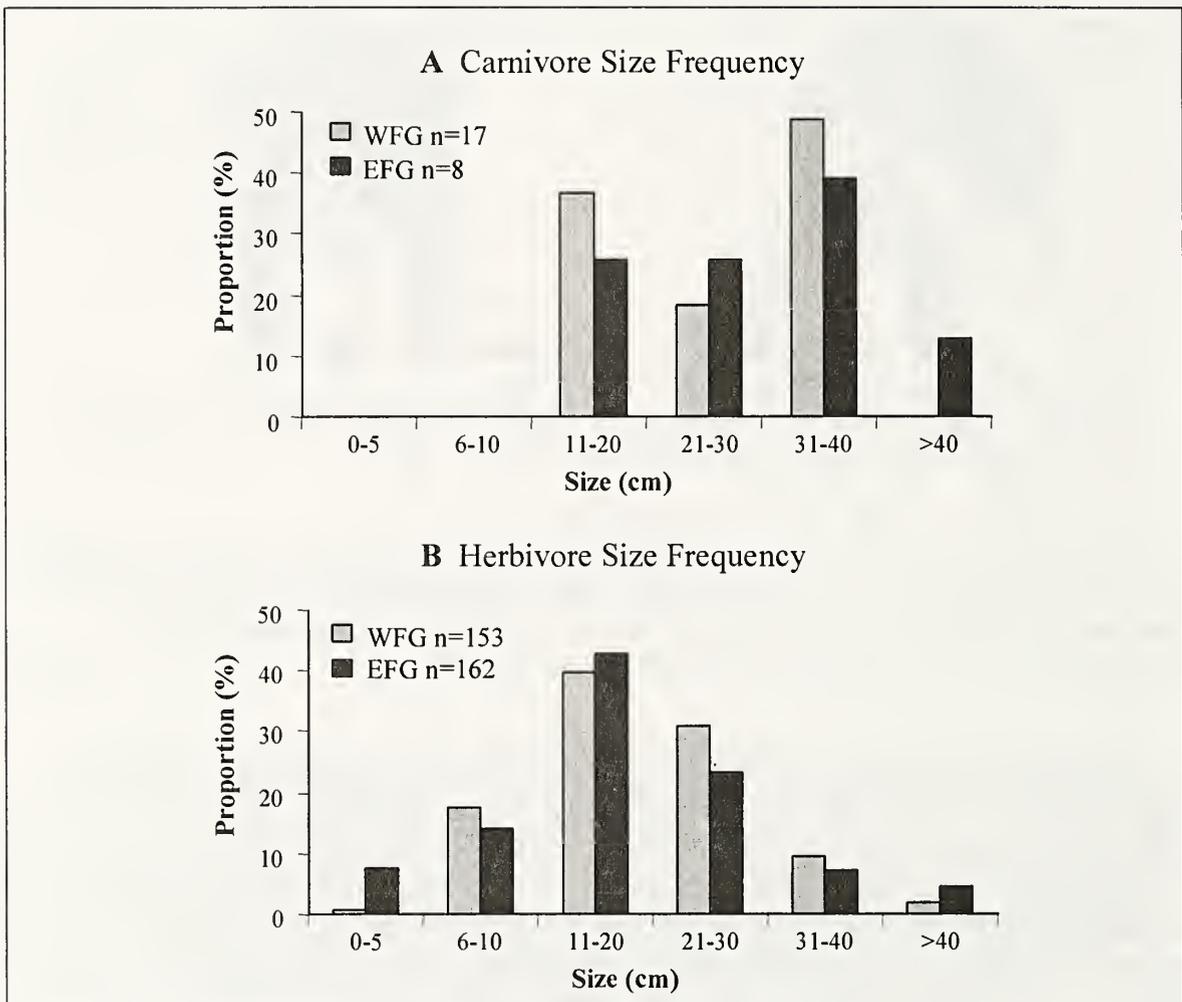


**Figure 3.** Mean fish density (no. individuals/100 m<sup>2</sup> ± sd) for eight fish families at the WFG (West Flower Garden) and EFG (East Flower Garden) Banks. Other = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Microspathodon chrysurus*, *Sphyraena barracuda*.

A total of 117 fish species were seen by the AGRRA team during RDT surveys at the EFG and WFG. Great barracuda (*Sphyraena barracuda*), sharpnose puffer (*Canthigaster rostrata*), and black durgon (*Melichthys niger*) were documented in all surveys (Table 4). Species that were relatively common at the FGB compared to most other Caribbean reefs include longsnout butterflyfish (*Chaetodon aculeatus*), blue angelfish (*Holacanthus bermudensis*), and several species of jacks (Carangidae). Individuals of the golden phase of the smooth trunkfish (*Lactophrys triqueter*), a phase that is unique to the FGBNMS (Pattengill-Semmens, 1999), were also sighted. One new record for the Banks, a sharptail eel (*Myrichthys breviceps*), was recorded at the WFG. (An individual of the same eel species had been recorded on video earlier in the summer.)

Numerically, the most abundant species were: bluehead wrasse (*Thalassoma bifasciatum*), threespot damselfish (*Stegastes planifrons*), bicolor damselfish (*S. partitus*) queen parrotfish (*Scarus vetula*), along with planktivorous creolefish (*Paranthias furcifer*) and brown chromis (*Chromis multilineata*).

Average sizes of parrotfishes and groupers were relatively high which resulted in relatively high biomass values (Table 5). The size frequency distributions of two feeding guilds, carnivores (select grouper genera and snappers) and herbivores (parrotfishes  $\geq 5$  cm, surgeonfish, and yellowtail damselfish, *Microspathodon chrysurus*), are shown in figure 4. Three-fourths of the individuals in the carnivore feeding guild were groupers (yellowmouth grouper, *Mycteroperca interstitialis*; tiger grouper, *M. tigris*; graysby, *E. cruentatus*; coney, *E. fulvus*; in descending order of density) with gray snapper (*Lutjanus griseus*) making up the remainder. Approximately 45% of these carnivores were greater than 30 cm in length (Fig. 4A) and the average size of the groupers was 25 cm. Approximately 70% of the individuals in the herbivore feeding guild were between 11 and 30 cm (Fig. 4B). The average size of the parrotfishes was 22 cm.



**Figure 4.** Size frequency distribution of (A) carnivores (lutjanids, select serranids) and (B) herbivores (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) at the WFG (West Flower Garden) and EFG (East Flower Garden) Banks.

For most species, average sizes recorded in the belt transects were similar between the EFG and WFG. Significant differences in length were detected by a t-test in the blue tang (*Acanthurus coeruleus*), which was longer at the WFG, whereas princess parrotfish (*Scarus taeniopterus*) and yellowtail damselfish were longer at the EFG.

## DISCUSSION

Results from this assessment revealed reefs with high stony coral cover that are dominated by large boulder corals. Pale bleaching was evident in some surveyed colonies but very little disease was noted and only on stony corals outside the benthic transects. Macroalgal biomass was very low. Results of the fish surveys showed an assemblage that is relatively low in diversity but high in biomass. The Banks appeared to support fewer, but larger, individual fishes in comparison to other Caribbean-area reefs. The large average size of parrotfishes and groupers likely reflected the low fishing pressure on the reefs. Recruitment success appeared to be driving the size differences for three of the surveyed fishes, giving a high abundance of juvenile blue tang on the EFG and a high number of juvenile princess parrotfish and yellowtail damselfish on the WFG.

An additional 15 REEF volunteers conducted 74 RDT surveys during the cruise. These data were not included in the AGRRA data set but were added to the REEF database, which can be accessed from the REEF Website (<http://www.reef.org>). As a result of annual field surveys at the Banks between 1993 and 1999, a total of 1,495 REEF surveys have been generated for the FGBNMS (over 1,100 survey hours). These data represent a valuable source of information for the Sanctuary management. To date, 257 fish species have been documented at the FGBNMS. A comprehensive fish species list for the FGB has been published using these data (Pattengill, 1998).

The FGB are deep reefs on offshore banks that are far removed from land and experience little anthropogenic disturbance. Described as “near pristine,” they provide an important piece of the regional picture of Mesoamerican reef condition. The reefs of the FGB have been, and continue to be, well studied. A long-term monitoring project has been in place for over 20 years. Data collected during the AGRRA assessment corroborated findings of previous studies on the condition of the FGB ecosystem. The importance of these data will be further highlighted when comparable data from dozens of sites are compiled to create a more complete picture of the current status of western Atlantic coral reefs. The FGB can then be used to help “calibrate” the AGRRA “scale” of reef condition.

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Table 1. Site information for AGRRA stony coral, algae and fish surveys on the Flower Garden Banks.

Site name/ buoy #c	Reef type	Latitude (° N)	Longitude (° W)	Survey date	Depth (m)	Benthic transects (#)	≥25 cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)	30 m fish transects (#)	RDT fish species (#) <sup>1</sup>
WFG #5	bank	27 55.30	93 48.54	August 16 99	22.5	11	12.5	54.0 ± 16.5	12	117
EFG #2	bank	27 54.32	93 35.49	August 18 99	19.5	14	11.5	49.0 ± 6.0	12	117

<sup>1</sup>Species number value at each bank is based on Roving Diver Technique (RDT) surveys.

Table 2. Size and condition (mean ± standard deviation) of all stony corals (≥25 cm diameter), by site on the Flower Garden Banks.

Site name/ buoy #	Stony corals		Partial-colony surface mortality (%)			Corals (%)				
	(#)	Diameter (cm)	Height (cm)	Recent	Old	Total	Standing dead	Bleached <sup>1</sup> Diseased	with Damselfish bites	
WFG #5	135	93.0 ± 72.0	36.0 ± 41.5	1.5 ± 4.0	12.0 ± 21.0	13.5 ± 20.5	1.5	5.5	0	8
EFG #2	160	81.0 ± 53.0	32.5 ± 26.5	2.0 ± 7.0	10.5 ± 19.0	12.5 ± 20.5	2.0	16.5	0	7.5

<sup>1</sup>Includes all colonies with any level (pale, white, etc.) and amount (partial, complete) of bleaching.

Table 3. Algal characteristics, abundance of stony coral recruits and *Diadema antillarum* (mean ± standard deviation), by site on the Flower Garden Banks.

Site Name/ buoy #s	Quadrats (#)	Relative abundance (%)		Crustose coralline alage	Macroalgal height (cm)	Macroalgal Index <sup>2</sup>	Recruits (#/0.0625 m <sup>2</sup> )	<i>Diadema</i> (#/100 m <sup>2</sup> )
		Turf algae <sup>1</sup>	Macroalgae					
WFG #5	55	9.0 ± 17.0	71.5 ± 28.5	19.5 ± 28.0	0.9 ± 0.3	8	0.15	0.9
EFG #2	67	9.0 ± 16.5	79.5 ± 22.0	11.0 ± 12.5	1.1 ± 0.5	10	0.10	1.4

<sup>1</sup>Includes cyanobacterial mats

<sup>2</sup>Macroalgal index is relative macroalgal abundance x macroalgal height

Table 4. Twenty-five most frequently sighted fish taxa on the Flower Garden Banks. Data were calculated from RDT surveys conducted during the assessment.

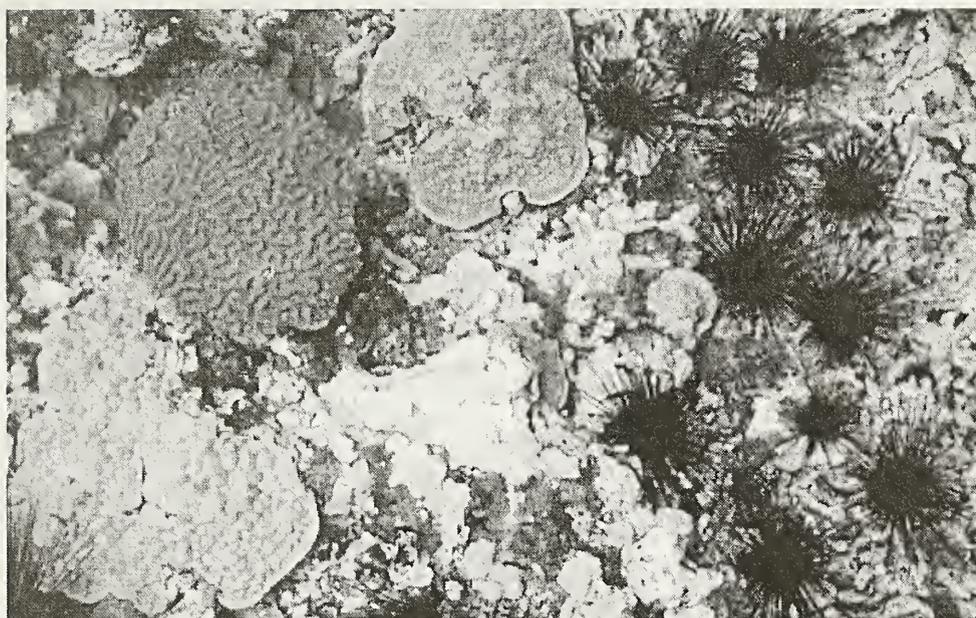
Scientific name	Common name	Sighting frequency (%)	Density score <sup>1</sup>
<i>Sphyraena barracuda</i>	Great Barracuda	100	2.8
<i>Canthigaster rostrata</i>	Sharpnose Puffer	100	2.7
<i>Melichthys niger</i>	Black Durgon	100	2.4
<i>Chaetodon sedentarius</i>	Reef Butterflyfish	96	2.2
<i>Microspathodon chrysurus</i>	Yellowtail Damselfish	96	1.9
<i>Mulloidichthys martinicus</i>	Yellow Goatfish	96	2.6
<i>Epinephelus cruentatus</i>	Graysby	96	1.8
<i>Scarus vetula</i>	Queen Parrotfish	96	3.0
<i>Sparisoma viride</i>	Stoplight Parrotfish	96	2.7
<i>Acanthurus coeruleus</i>	Blue Tang	96	2.3
<i>Lactophrys triqueter</i>	Smooth Trunkfish	92.5	1.8
<i>Stegastes planifrons</i>	Threespot Damselfish	92.5	3.2
<i>Clepticus parrae</i>	Creole Wrasse	92.5	2.8
<i>Kyphosus sectatrix/incisor</i>	Bermuda Chub/Yellow Chub	87.5	2.9
<i>Chromis multilineata</i>	Brown Chromis	84.5	3.5
<i>Stegastes partitus</i>	Bicolor Damselfish	84.5	3.1
<i>Bodianus rufus</i>	Spanish Hogfish	84.5	2.8
<i>Thalassoma bifasciatum</i>	Bluehead	84.5	3.6
<i>Scarus taeniopterus</i>	Princess Parrotfish	84	2.7
<i>Cantherhines pullus</i>	Orangespotted Filefish	80	1.6
<i>Paranthias furcifer</i>	Creole-fish	80	3.8
<i>Halichoeres garnoti</i>	Yellowhead Wrasse	77	2.3
<i>Holacanthus tricolor</i>	Rock Beauty	76	1.7
<i>Acanthurus bahianus</i>	Ocean Surgeonfish	73	2.5
<i>Chromis cyanea</i>	Blue Chromis	72.5	2.3

<sup>1</sup>See Methods for definition of density score.

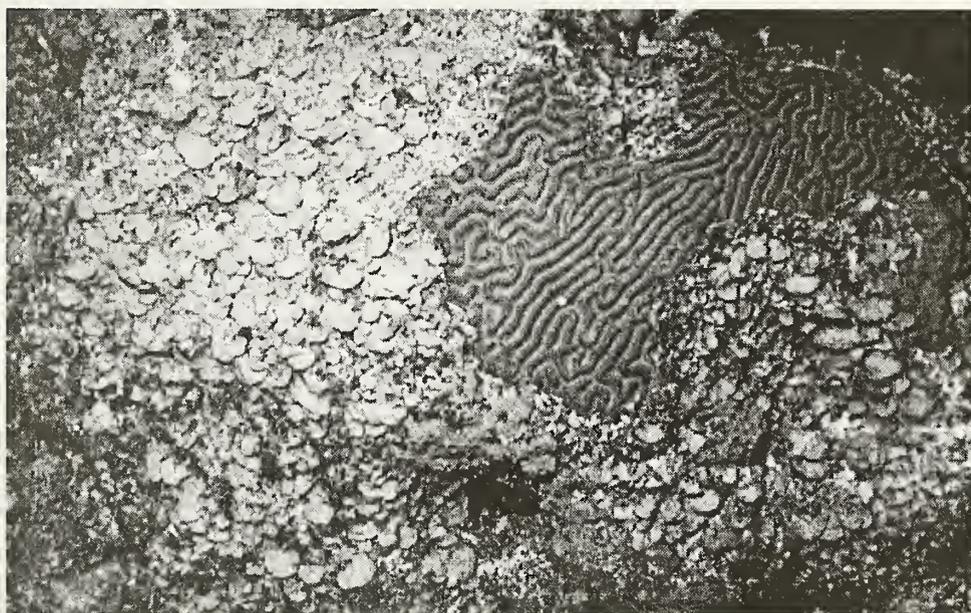
Table 5. Biomass (mean  $\pm$  standard deviation) for AGRRA fishes, by site in the Flower Garden Banks.

Site name	Biomass (g/100 m <sup>2</sup> )				
	Herbivores		Carnivores		
	Acanthuridae	Scaridae ( $\geq 5$ cm)	Haemulidae ( $\geq 5$ cm)	Lutjanidae	Serranidae <sup>1</sup>
WFG #5	960.4 $\pm$ 864.5	4263.1 $\pm$ 2984.0	0	376.9 $\pm$ 736.5	554.3 $\pm$ 1041.3
EFG #2	493.2 $\pm$ 370.3	4765.2 $\pm$ 3387.6	0	252.8 $\pm$ 875.6	343.4 $\pm$ 473.0

<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.



**Plate 12A.** The formerly ubiquitous, herbivorous sea urchin, *Diadema antillarum*, played a key role in preventing overgrowth of stony corals by macroalgae in many areas of the wider Caribbean prior to its regionwide demise in 1983-1984. Localized population increases are currently being reported, although densities everywhere are still far below pre-dieoff levels. (Photo Andrew W. Bruckner)



**Plate 12B.** On many Caribbean reefs lacking large sized herbivorous fishes, the loss of *D. antillarum* has allowed macroalgae to colonize coral skeletons and then overgrow the living coral tissues, as shown for the *Lobophora variegata* on this *Colpophyllia natans*. The result has been a shift to algal dominated reefs in some areas, as well as increased mortality and reduced recruitment of stony corals. (Photo Andrew W. Bruckner)

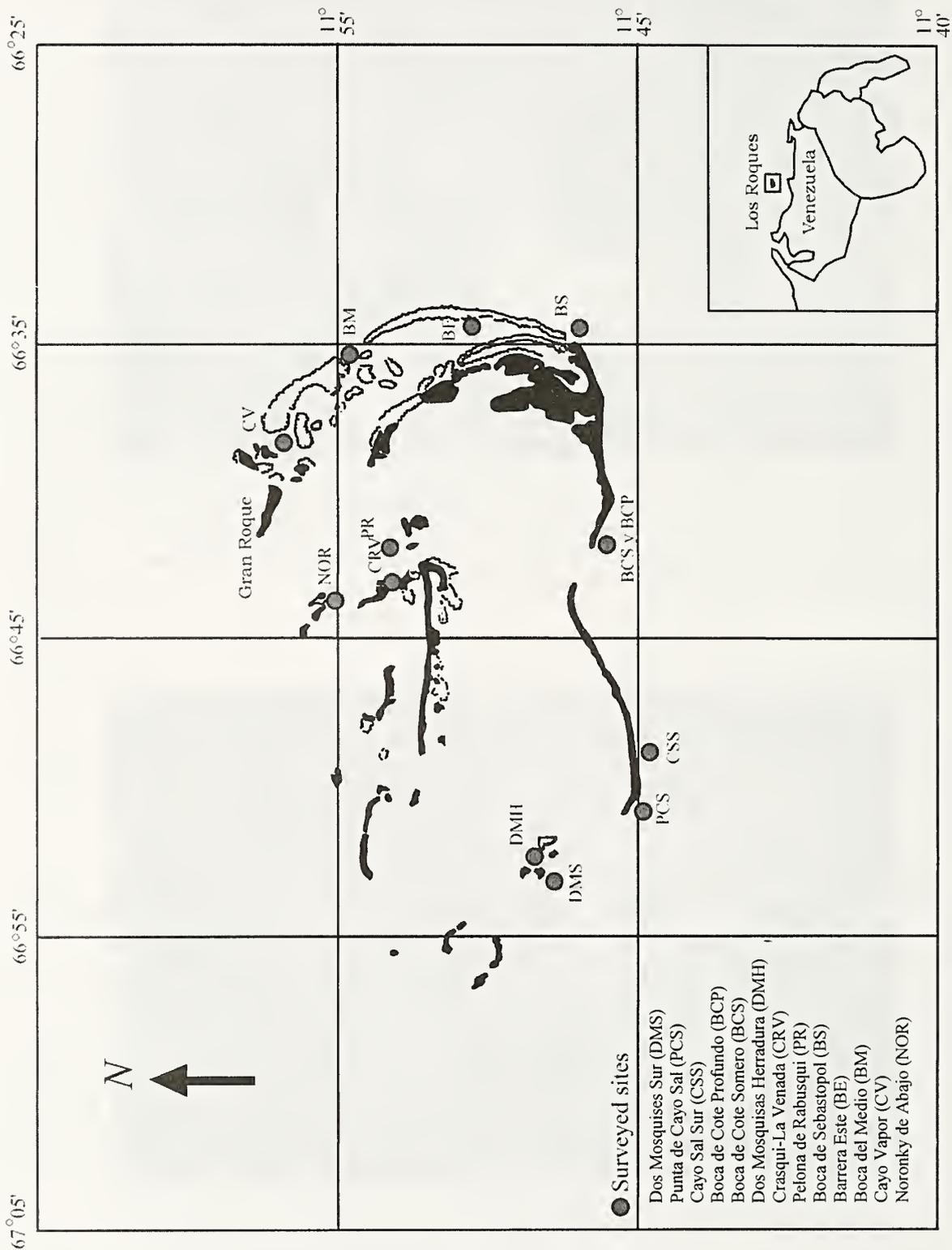


Figure 1. AGRRA survey sites in the Archipiélago de Los Roques National Park, Venezuela.

# RAPID ASSESSMENT OF CORAL REEFS IN THE ARCHIPIÉLAGO DE LOS ROQUES NATIONAL PARK, VENEZUELA (PART 1: STONY CORALS AND ALGAE)

BY

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## ABSTRACT

The status of the coral reefs in Archipiélago de Los Roques National Park, Venezuela was appraised in October, 1999 at 13 sites using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocol. The reef complex appears to be in good condition overall. Live stony coral cover was especially high (30-60%) at four sites between 8-13 m in the southern and southwestern reefs. However, “standing dead” *Acropora* were particularly conspicuous in the eastern barrier reef and yellow-band disease was present in some colonies of *Montastraea annularis* and *M. faveolata*. Average values of recent partial-colony mortality were 8-10% in stony corals of  $\geq 10$  cm diameter in three reefs. Macroalgae were scarce (<10% relative abundance) in most (12/13) sites. The Los Roques reefs are likely to serve as important sources of coral larvae traveling through the southern Caribbean. Their protected status must be maintained.

## INTRODUCTION

The Archipiélago de Los Roques National Park is Venezuela’s major coral reef complex and one of the most important in the southern Caribbean region. Located 150 km north of the central Venezuelan coast, it is outside the geographic area routinely influenced by hurricanes. Land-based anthropogenic impacts are reduced in comparison to those in Venezuela’s continental reef systems. The park, however, is subject to the impacts of commercial fishing pressures (mostly for spiny lobsters) and to increasing touristic activity (see Posada et al., this volume). Regional diseases, in particular white-

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band disease (WBD), have also impacted its stony coral populations, particularly the important framework builder *Acropora palmata* (García, 2001; León, 2001).

The archipelago, which developed during the late Pleistocene around a metamorphic-igneous platform centered in Gran Roque Island, includes two main barrier reefs. The 24 km long eastern barrier, which runs from northeast to southeast, is fully exposed to the prevailing winds and seas. Strong currents characterize its seaward side. The southern barrier extends about 30 km from southeast to southwest and is progressively more protected from waves and currents as one moves from east to west. By protecting the interior of the archipelago from the direct influence of oceanic waves and currents, the barriers have facilitated the formation of numerous patch reefs, sand cays with fringing reefs, seagrass meadows and, in the southeastern part of the archipelago, mangrove forests (Méndez-Baamonde, 1978).

Annual surface seawater temperatures at Los Roques average 26° C (sd=2° C) with a minimum of about 23° C in February and a maximum of about 30° C in May although temperatures as high as the maximum can be found during October and November (Villamizar, 1993). Vertical water transparency is relatively high (15 to 25 m).

Méndez-Baamonde (1978) provided the first qualitative description of the major reef habitats in the archipelago. However, much of the research which has been undertaken in the Los Roques coral reefs is either unpublished or in undergraduate theses. Hung (1985) found that species distribution patterns, numbers of colonies and live stony coral cover varied with habitat in the southwestern Dos Mosquises Sur (DMS) fringing reef. Coral cover was greatest (7,200 cm<sup>2</sup>/m<sup>2</sup>) at “intermediate depths” (3-18 m) in the terrace and slope zones which were dominated by the *Montastraea annularis* complex. Hung (1985) also found numerous dead colonies of *Acropora palmata* in the reef crest at DMS and at another nearby fringing reef [Dos Mosquises Herradura (DMH)]. Sandía and Medina (1987), who studied the population dynamics of *A. cervicornis* off DMS, reported a maximum dominance (~18.0% of live coral cover and 9% of the colonies) of this species on the fore-reef terrace in depths of 3-6 m where numerous dead fragments of *A. cervicornis* were also found. *A. palmata* was the dominant species in shallower (<3 m) water while massive corals (particularly *M. annularis* and *Colpophyllia natans*) and gorgonians were most common below 6 m. Subsequently, Cróquer and Villamizar (1998) studied the effect of topographic variation on reef community structure in four sites at DMS finding the largest number of stony coral species (24) in the site with the least slope. Nevertheless the between-site similarity index of species composition was higher for the reef-slope zone than for the reef flat, reef crest and deep reef flat.

García (2001) found that 6.87 % of the stony corals (n=3,344) examined during 2000 in seven Los Roques localities were affected by disease. Yellow-band (yellow-blotch) disease (YBD) and dark spots disease (DSD) were the most common (both 2.1%) with other diseases each occurring in fewer than 1% of the colonies.

León (2001) has assessed the effects of recreational diving activities by comparing one each experimental (frequently visited by divers) and control (low touristic activity) site in several of the Los Roques reefs. Live stony coral cover overall averaged 32.6 % (sd=16) and was higher in the control sites than in the experimental sites in three reefs [including Crasquí-La Venada (CRV) with 49% versus 11% and Pelona de Rabusquí (PR) with 53% versus 16%, respectively] and lower in the controls only in the

DMS reef. León (2001) also found that YBD and DSD were the most common stony coral diseases at these sites in 2000.

The purpose of the present study was to evaluate the condition in October 1999 of the Los Roques coral reefs through the application of the AGRRA protocols. Stony coral and algal components are considered in this paper; the fish assessment is given by Posada et al. (this volume). Our results provide a comprehensive overview of the archipelago's reefs, allowing comparisons with other areas in the Caribbean (Kramer, this volume), and creating a baseline against which future assessments can be compared.

## METHODS

Thirteen sites which are considered by the two senior authors to be representative of the barrier, fringing, and patch-reef habitats at Los Roques were chosen for assessment. The majority of the sites were located along the axes of the two main barrier reefs in recognition of their size and ecological importance; only the northwest sector of the archipelago was not surveyed.

Three divers employed the AGRRA Version 2.2 benthos protocol (see Appendix One, this volume) with the following modifications. The minimum diameter of individually surveyed corals was 10 cm. Coral size was recorded to the nearest 5 cm. Species of stony corals that remain small as adults (e.g., *Favia fragum*) were omitted from the counts of coral recruits. Sediments were removed fairly vigorously from the algal quadrats before estimating the abundance of crustose coralline algae. Training in stony coral identifications and in designation of the percentage of partial mortality and algal abundance was conducted the day before the surveys officially began. Field identifications of corals were corroborated by reference to Humann (1994).

Correlation coefficients were calculated to examine relationships between the percentage of standing dead corals and both the abundance and partial mortality of *A. palmata*. Distributions were normalized by using the log N+1 transformation for abundances and the arcsine root transformation for proportions.

## RESULTS

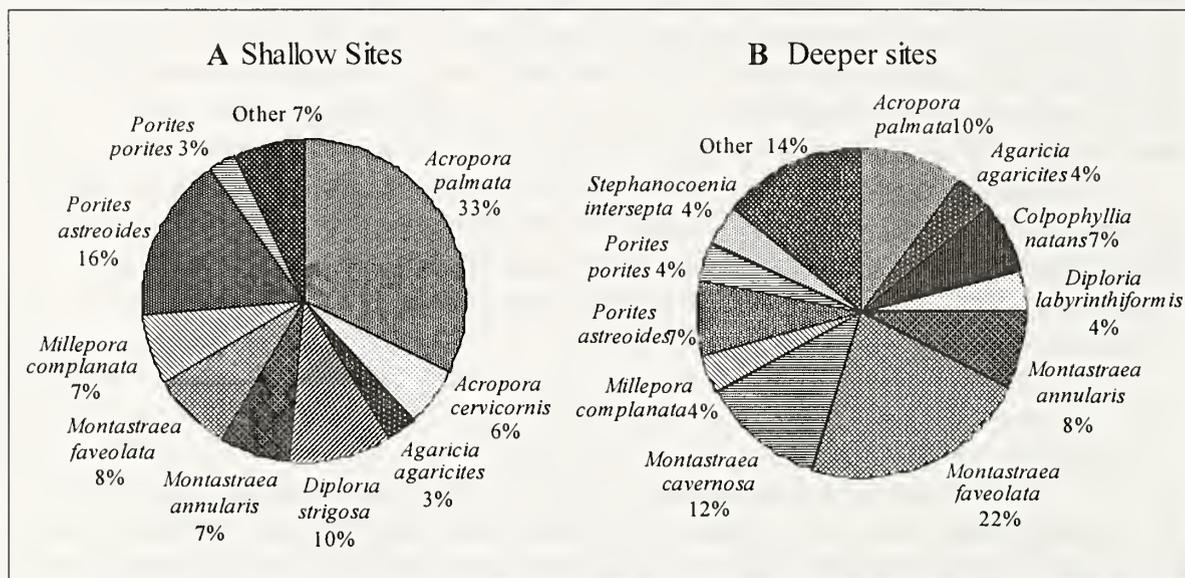
The 13 surveys were conducted at four sites each in the eastern and southern barrier reefs, at two sites each in the central-eastern and southwestern fringing reefs, and in one northeastern patch reef (Table 1, Fig. 1). Back-reef habitats were surveyed in the eastern barrier [Cayo Vapor (CV), Boca del Medio (BM), Barrera Este (BE) and Boca de Sebastopol (BS)] where our access to the fore reef was restricted by high waves. One outer reef crest [Boca Cote Somero (BCS)] and three fore reefs [Punta Cayo Sal (PCS), Cayo Sal Sur (CSS) and Boca del Cote Profundo (BCP)] were visited in the southern barrier, two of which (BCS, BCP) were located near a large mangrove forest. The reef at DMS has a maximum depth of 40 m, but the remaining fringing reefs (PR, CRV and DMH) extend no deeper than 6.5, 10, and 12 m, respectively, and the base of the Noronqui de Abajo (NOR) patch reef is at 2 m. Eight of the survey sites were considered shallow ( $\leq 6.5$  m) and five as being in deeper water (7.5-13.5 m).

## Stony Corals

The total cover of live stony corals (Table 1) varied from <10% in four shallow reefs (BS, CV, CRV, BCS) to >50% in three of the deeper southern reefs (PCS, BCP and DMS). Live coral cover in the remaining sites ranged between about 20 and 30%. The density of stony corals that were at least 10 cm in diameter was higher overall in the southern sites than along the eastern side of the archipelago (Table 1).

A total of 27 scleractinians (including *Eusmilia fastigiata*, *Favia fragum* and *Porites branneri* in addition to species on the original AGRRA coral list) and one hydrozoan (*Millepora complanata*) were large enough ( $\geq 10$  cm in diameter) for individual survey. Seven to 13 of these species occurred in each of the shallow reefs where colonies of *Acropora palmata* that were mostly dead (see below) numerically dominated most (six/eight) of the reefs, along with *Porites astreoides* and *Diploria strigosa* (Fig. 2A). Live *A. cervicornis* was found in only two of the shallow reefs (NOR, where *A. palmata* was rare, and DMH). *M. annularis* occurred in six of the shallow reefs (BCS, DMH, CV, PR, CRV and NOR). Although *M. faveolata* was present in seven shallow reefs, it was relatively abundant only in the deepest (PR). *M. complanata* was abundant in one shallow reef on the southern barrier (BCS).

Eleven to 17 species of the individually surveyed corals were recorded in each of the deeper reefs. The most common species, *M. faveolata*, *M. annularis*, *M. cavernosa*

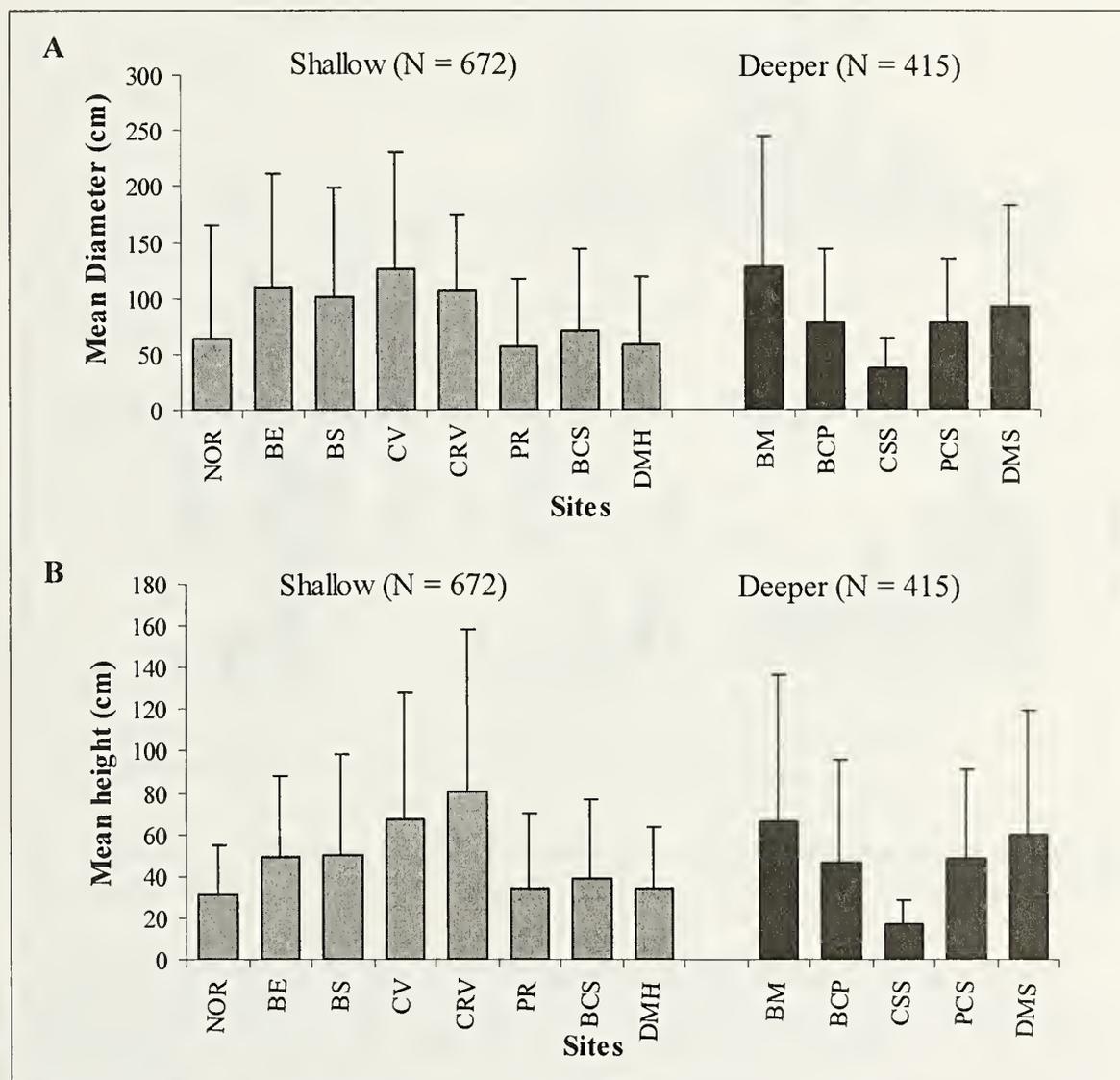


**Figure 2.** Species composition and mean relative abundance of the most abundant stony corals ( $\geq 10$  cm diameter) in (A) shallow ( $n=672$ ) and (B) deeper ( $n=415$ ) sites in Los Roques, Venezuela.

Other: (A) = *Agaricia tenuifolia*, *Colpophyllia natans*, *Diploria clivosa*, *D. labyrinthiformis*, *Favia fragum*, *Porites branneri*, *Siderastrea radians*, *S. siderea*; (B) = *A. tenuifolia*, *Dendrogyra cylindrus*, *Dichocoenia stokesi*, *D. clivosa*, *D. labyrinthiformis*, *Eusmilia fastigiata*, *F. fragum*, *Madracis decactis*, *M. mirabilis*, *Meandrina meandrites*, *Montastraea franksi*, *P. branneri*, *S. radians*, *S. siderea*.

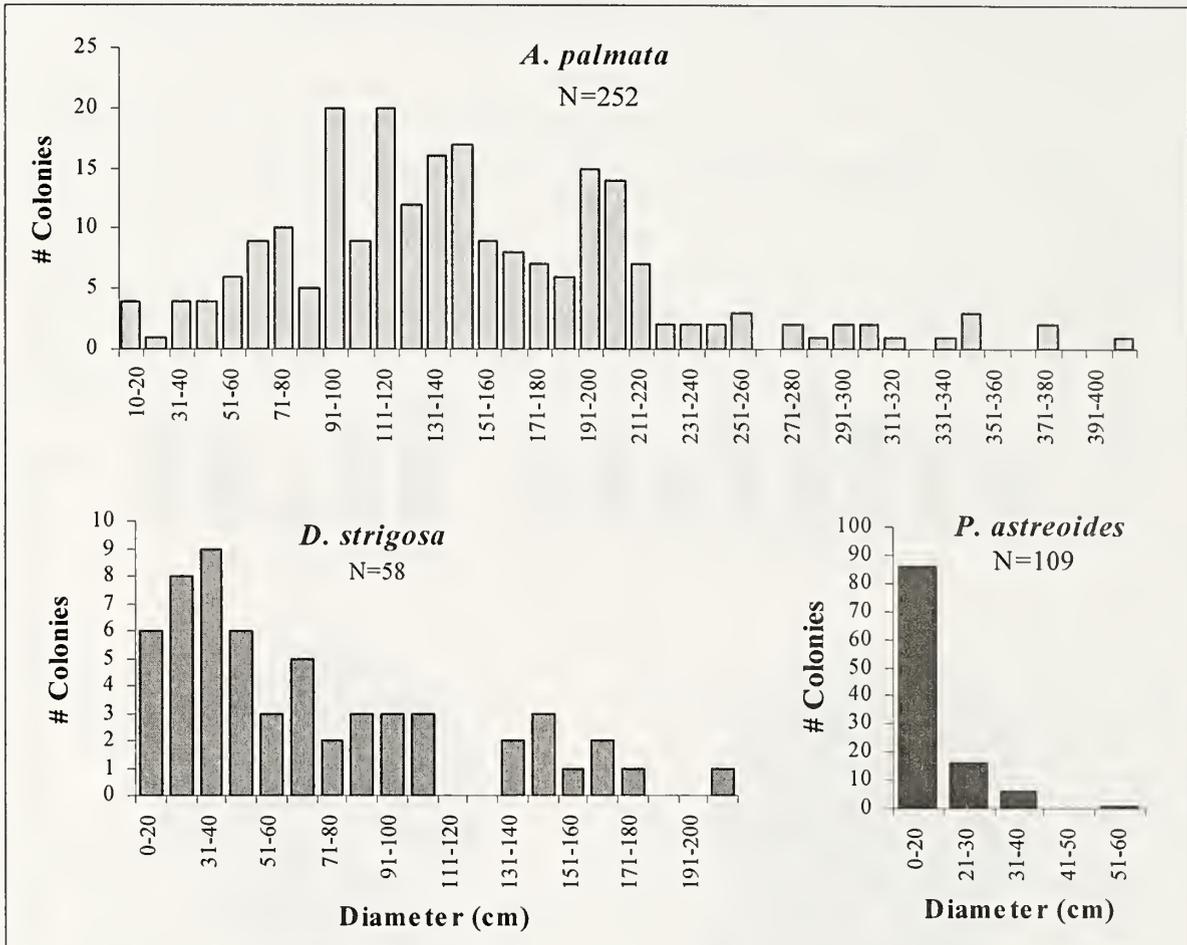
(especially at CSS) and *C. natans* (in particular at DMS), all had massive morphologies (Fig. 2B) except in the eastern barrier reef (at BM) where *A. palmata*, *P. astreoides* and *Agaricia agaricites* were predominant.

Average values in excess of 100 cm (Table 2) were found for the maximum diameters of individually surveyed stony corals in the five eastern reefs that were numerically dominated by *A. palmata* (BE, BS, CV, CRV and BM). The smallest (38 cm) was in one of the deeper, southern barrier reefs (CSS) where *M. cavernosa* was the most abundant species and the largest colonies of *M. faveolata* were less than 120 cm in maximum diameter. Average height was closely correlated with maximum diameter in most sites (Fig. 3).



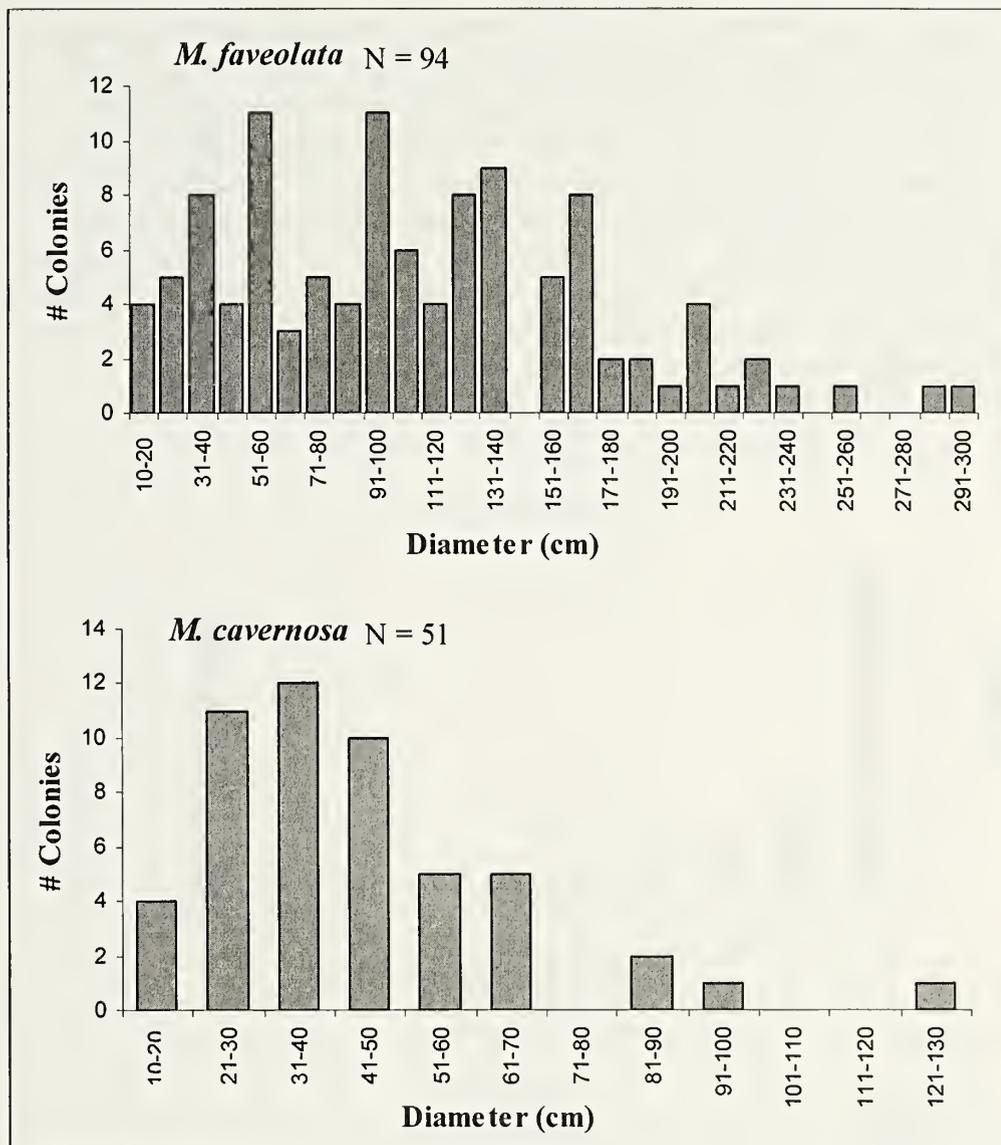
**Figure 3. (A)** Diameter and **(B)** height (mean  $\pm$  sd) of all stony corals ( $\geq 10$  cm diameter) by site in Los Roques, Venezuela. See Table 1 for site codes.

Many colonies of *A. palmata* in the shallow reefs were >100 cm in maximum diameter and the largest exceeded 400 cm (Fig. 4). A majority of the *D. strigosa* were <100 cm although some were larger (to 210 cm). *P. astreoides* was almost exclusively represented by colonies with diameters of <20 cm. *M. faveolata* in the deeper reefs resembled the shallow *A. palmata* in having a wide range of maximum diameters with the largest colonies being ~300 cm in maximum diameter (Fig. 5). Most colonies of the smaller-sized *M. cavernosa* were in the 20-70 cm size intervals.



**Figure 4.** Size-frequency distributions of ≥10 cm diameter colonies of *Acropora palmata*, *Diploria strigosa* and *Porites astreoides* at shallow sites in Los Roques, Venezuela.

No stony coral “recruits” (≤2 cm diameter) were recorded in seven (three shallow, four deeper) of the survey sites (Table 3). Their densities in the remainder were less than five corals/m<sup>2</sup> (~0.3/.0625 m<sup>2</sup>) at all but two of the shallower sites (BS, PR). *Agaricia* and *Porites* were the most common genera. *Agaricia* recruits appeared to preferentially colonize dead *A. palmata* branches (particularly in BM). Although no recruits of *Acropora*, *Montastraea* or other large massive corals were found in the quadrats, some small established *A. cervicornis* recruits were noted outside the AGRRA survey site in the crest of the patch reef (NOR).

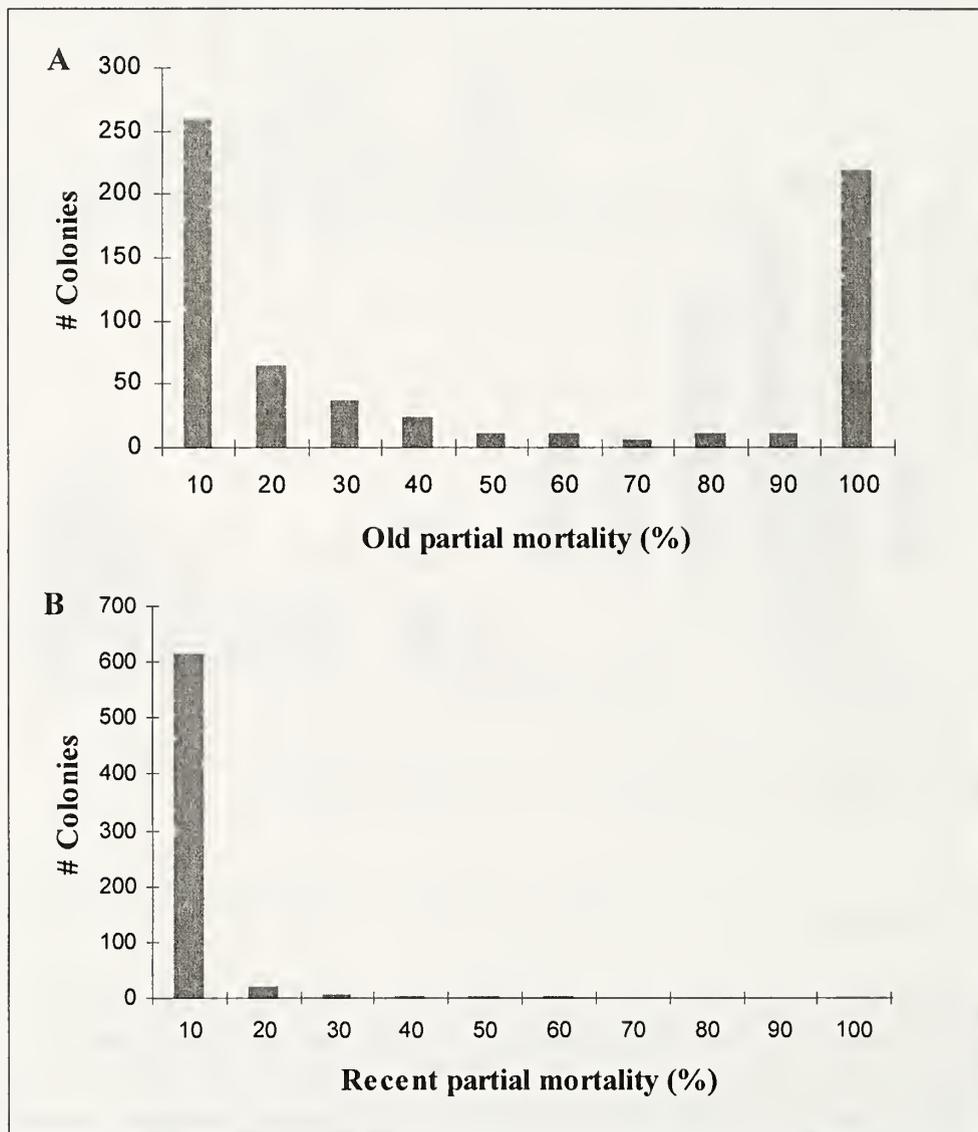


**Figure 5.** Size-frequency distributions of  $\geq 10$  cm diameter colonies of *Montastraea faveolata* and *M. cavernosa* at deeper sites in Los Roques, Venezuela.

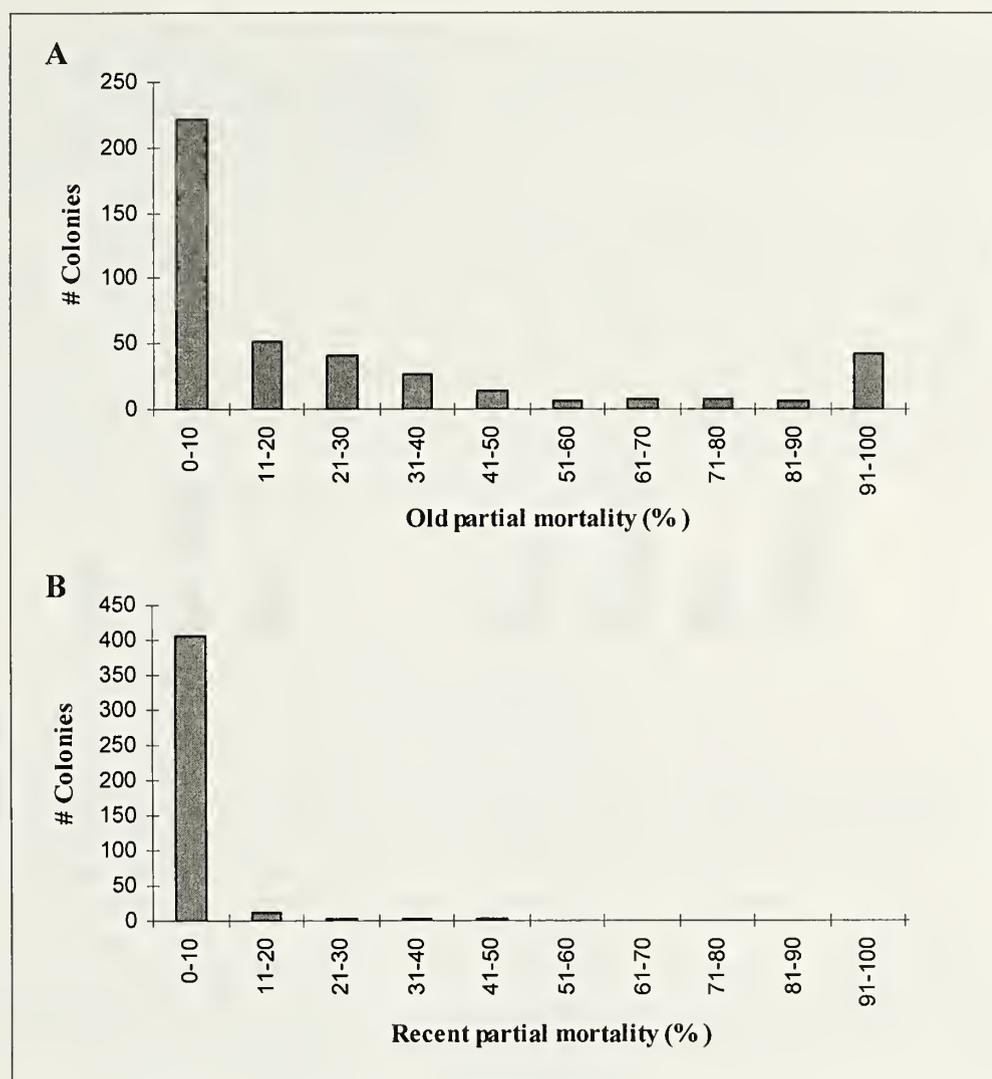
### Stony Coral Condition

No totally bleached corals were observed at the time of our surveys. At seven sites, however, fewer than 4% of the colonies were either pale or partially bleached (Table 2). There was no evidence of infection with WBD in any of the live colonies of *Acropora*. YBD was found in three sites (Table 2), occurring in *Montastraea annularis* (seven colonies), *M. faveolata* (three colonies), and *M. franksi* (one colony). A total of one colony each of *Porites porites*, *P. astreoides* and *A. palmata* also showed signs of a disease resembling YBD in overall appearance. One colony of *M. faveolata* had signs resembling those of red-band disease and a *Diploria strigosa* had white plague.

Average values of recent partial-colony mortality (hereafter recent mortality) varied from <1% in two of the deeper reefs in the southern barrier to >8% in three shallow reefs (CRV, PR and BCS) (Table 2). In addition to the above-mentioned diseases, mortality agents identified in the shallow localities included predation by scarids and the snail *Coralliophila* plus overgrowth by gorgonians and algae. However, a large number of the shallow colonies with recent mortality were associated with territories of *Stegastes planifrons* (see below). Recent mortality was somewhat lower overall in the deeper sites (Table 2, Figs. 6, 7). We observed no predation by *Diadema antillarum* on live stony corals, although such behavior had been previously observed (Villamizar, personal observations during 1995 in DMH).



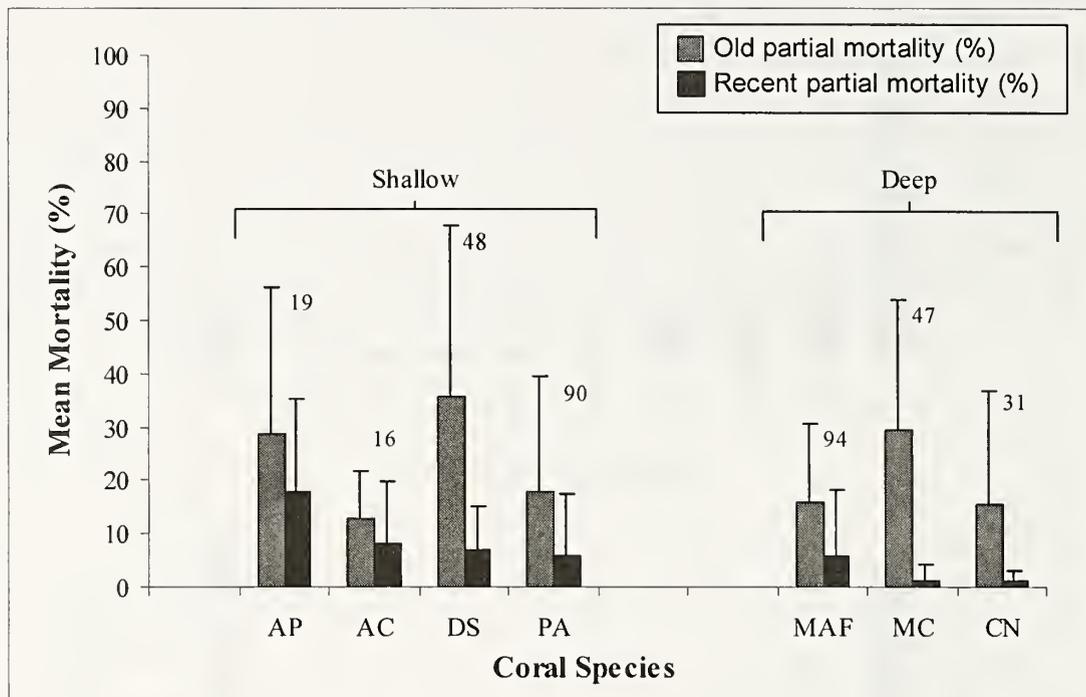
**Figure 6.** Frequency distributions of (A) old partial-colony mortality and (B) recent partial-colony mortality of all stony corals ( $\geq 10$  cm diameter) at shallow sites in Los Roques, Venezuela.



**Figure 7.** Frequency distributions of (A) old partial-colony mortality and (B) recent partial-colony mortality of all stony corals ( $\geq 10$  cm diameter) at deep sites in Los Roques, Venezuela.

Old partial-colony mortality (hereafter old mortality) ranged from  $<30\%$  in six sites (two shallow, four deeper) to  $>60\%$  in three of the shallow sites (Table 2). Total (recent + old) partial mortality was  $<25\%$  in three of the four deeper southern-to-southwestern sites,  $\sim 43-75\%$  in the four reefs along the eastern barrier, and over  $80\%$  in one of the central-eastern fringing reefs (CRV).

Of the colonies with some remaining live tissues (i.e., for which total mortality was  $<100\%$ ), recent mortality values for the dominant species varied from about  $1\%$  for *M. cavernosa* and *C. natans* in the deeper sites to approximately  $18\%$  for *A. palmata* in shallow sites (Fig. 8). The corresponding averages for old mortality were lowest in shallow *A. cervicornis* ( $\sim 13\%$ ) and highest in *D. strigosa* ( $38\%$ ). The ratio of recent mortality to old mortality was lowest in the deeper *M. cavernosa* ( $0.04$ ) and highest in the shallow *Acropora* spp. ( $0.62$  in *A. palmata*;  $0.64$  in *A. cervicornis*).



**Figure 8.** Percent old partial-colony mortality and percent recent partial colony mortality of  $\geq 10$  cm diameter stony corals with at least some live tissues (i.e., for which mortality is  $< 100\%$ ) at shallow and deep sites in Los Roques, Venezuela. AP = *Acropora palmata*, AC = *A. cervicornis*, DS = *Diploria strigosa*, PA = *Porites astreoides*, MAF = *Montastraea faveolata*, MC = *M. cavernosa*, CN = *Colpohyllia natans*.

Over 25% of the colonies were “standing dead” (completely dead but still attached to the substratum) in six survey sites. All but one of these, BM, were shallow and five sites were located in (BS, CV, BE, BM) or near (CRV) the eastern barrier. *Acropora palmata* accounted for most of these standing dead with  $< 7\%$  (17/252) of all surveyed colonies having any live tissues. Statistically significant correlations were found between the percentage of standing dead corals in each site and both the abundance of *A. palmata* ( $r=0.93$ ,  $p<0.05$ ,  $n=13$ ) and the percentage of its total mortality ( $r=0.87$ ,  $p<0.05$ ,  $n=13$ ). Only one reef (DMH) had a high proportion (11/14) of colonies of *A. palmata* still having some living tissues. Elsewhere, with the exception of one or two colonies in each site that were partially alive, the remainder were standing dead.

#### Algal groups, *Diadema* and Damselfishes

Overall, turf algae clearly dominated the algal functional groups in seven shallow and three deeper sites (Table 3). Crustose coralline algae (especially *Lithophyllum*) and *Peyssonnelia* were dominant in two deeper reefs (CSS, DMS) and one shallow (BCS) reef and both were essentially codominant with turfs in one shallow fringing reef (CRV). Macroalgae were completely absent from quadrats in three sites (one shallow, two deeper) and conspicuous only at Boca de Cote shallow (BCS) where their relative abundance was 29% at 2.5 m. Commonly observed macroalgae were *Halimeda opuntia*, *H. monile*, *Bryopsis pennata*, *Cladophoropsis* sp., *Ventricaria* sp., *Wrangelia penicillata*,

*Ceramium rubrum*, *Amphiroa fragilissima*, and *Dyctyopteris delicatula*. The cyanobacterium, *Blennothrix lyngbyacea*, was included in the macroalgal group. Macroalgal indices (macroalgal relative abundance x macroalgal height) were 0-1 at nine sites, 2-10 at three sites, but 70 in one shallow site (BCS, where macroalgae also grew somewhat taller than the average elsewhere).

The sea urchin, *Diadema antillarum*, was present in five shallow sites (Table 3). Maximum densities of about 36 individuals/100 m<sup>2</sup> were found in a shallow fringing reef (DMH). Damselfish, particularly *Stegastes planifrons*, exhibiting algal-gardening behavior in individually surveyed stony corals were recorded in all but one site (CSS), with the highest percentages found in shallow water (Table 2). Algal gardens were primarily found in live colonies of *M. faveolata* and were also present in *M. annularis*, *M. franksi* and *Millepora complanata*; in one site (CRV), the damselfish showed a high preference for standing dead colonies of *A. palmata*.

## DISCUSSION

The application of the AGRRA protocol, in concert with the recent studies of García (2001) and León (2001), has given us a general overview of the present condition of the barrier, fringing, and patch reefs in Los Roques National Park. The earlier descriptions of Méndez-Baamonde (1978) can now be updated and our results will allow comparisons of this important oceanic marine system with other reefs in the wider Caribbean (Kramer, this volume).

Hung's (1985) *area-based* estimate of percent live stony coral cover in a shallow SW fringing reef (DMH) in 1984 was highest at 2-3 m (mean=59, sd=17.5) which is about double that of our *linear* cover estimate in the same habitat in 1999 (Table 1). Numerous taxa (*Acropora prolifera*, *Agaricia* sp., *Dendrogyra cylindrus*, *Madracis* spp., *Meandrina meandrites*, *Agaricia lamarcki*) that routinely achieve diameters in excess of 10 cm were listed by Hung (1985) but absent from the AGRRA surveys. Although the *A. prolifera* may well have disappeared as a result of WBD, we are unsure if the apparent loss of the other taxa is real or an artifact of different sampling intensity. Although we found no diseased corals in this site, García (2001) reported that 5.8% of the 605 stony corals (all size classes) examined here in 2000 showed signs of disease.

In 1984, the percent of live stony coral cover in the DMS fringing reef was highest (mean=72, sd=22.5, areal estimate) at 3-18 m (Hung, 1985). Relative to that baseline, our linear-based percentage data at 12 m in 1999 (mean=60, sd=18.5) and León's (2001) areal estimate at 1-11 m in the experimental diving area (mean=52.8, sd=25.2) are somewhat reduced, but in León's (2001) control area live stony coral coverage was substantially lower (mean=33.5, sd=3.2). Of the seven sites studied in 2000 by García (2001), the frequency of diseased corals (11.6%, n=335 corals) was greatest in the DMS fringing reef and an order of magnitude larger than what we had found the previous year (Table 2).

García (2001) also noted proportionately higher incidences of disease in 2000 in two other sites that were visited by the AGRRA team (6.7% at CRV; n=492 corals, and 5.8% at Boca de Cote; n=542 corals). However, the two datasets are not strictly

comparable since García (2001) examined all size classes and surveyed in slightly different depths (1.5-5 m and 4-9 m).

Overall, during the quarter century since Méndez-Baamond's (1978) pioneering observations, the most striking change in shallow-intermediate depths at Los Roques has been the near disappearance of species of *Acropora*. Many colonies died during the early 1980's (Hung, 1985; Medina and Sandía, 1987), presumably from the effects of WBD as happened in so many areas of the wider Caribbean (Gladfelter, 1982; Aronson and Precht, 2001). Live *A. palmata* were still very rare in 1999 at the AGRRA survey sites with over half of the surviving colonies (11/17) occurring in a single shallow reef (DMH). Live colonies of *A. cervicornis* in the shallow sites were somewhat more abundant (e.g., 19 at DMH, 11 at NOR), more widespread (also occurring outside the belt transects in CRV, PR and BCS) and had a higher ratio of live/dead colonies (35/42). As we saw no recruits of *A. palmata* and those of *A. cervicornis* were only present in one site (NOR), it is very unlikely that their populations can recover any time in the near future to levels comparable to those observed in the archipelago during the 1970s.

In contrast to the reduced coverage in many shallow reef habitats, live stony coral cover values of ~30-60% at 8-13.5 m in the southern and southwestern reefs are high relative to recent reports from comparable habitats throughout much of the wider Caribbean (Kjerve, 1998; Kramer, this volume).

The high proportion of large colonies of *A. palmata* and *D. strigosa* in shallow sites (Fig. 4) and of *M. faveolata* in deeper sites (Fig.5) is evidence that, prior to the arrival of WBD, conditions for the growth and survival of stony corals at Los Roques must have been excellent for many decades or even centuries. Smaller colonies of these species may reflect past recruitment (although recruits of these species were absent in the AGRRA surveys) or subdivision of colonies from physical breakage or partial mortality of the live tissues.

The most common and widespread disease recorded in the 1999 AGRRA assessment was YBD (Table 2), which primarily occurs in *Montastraea*, and it was tied for first place with DSD in García's (2001) surveys during 2000. Given the seminal importance of *Montastraea* spp. (Figs. 2, 5) in these habitats, it would be of great concern should this trend continue.

In all but two shallow localities (BM, DMH), the key herbivore, *Diadema antillarum*, was either very rare or absent. Macroalgae were extremely sparse (<10% relative abundance) in all reefs except Boca de Cote, where the abundance of herbivorous fishes was somewhat reduced in the deeper (BCP) site (Posada et al., this volume) and where macroalgal indices (a proxy for biomass) were only elevated in the shallow (BCS) site. Hence, we suspect that nutrients are naturally exported from the large mangrove forests nearby in Sebastopol and stimulate benthic macroalgal production at Boca de Cote, particularly in shallow water where proportionately more sunlight is available for photosynthesis.

From the results obtained in this study, it is possible to conclude that the Archipiélago de Los Roques as a whole is still a "healthy" coral reef system, even allowing for the drastic declines that have occurred in its populations of *A. palmata* and *A. cervicornis*. However, the deeper sites in the southern barrier and a southwestern fringing reef were in better condition than the shallow sites. We suggest that major efforts

must be made to continue to protect this reef system, which is of great importance for Venezuela and the Caribbean region.

### ACKNOWLEDGMENTS

Special thanks to Patricia Kramer for helping us with the initial training in the application of AGRRA methods and with the surveys at Los Roques. Thanks to the Caribbean Environment Programme of the United Nations Environment Programme for its support of this study. The AGRRA Organizing Committee facilitated financial support as described in the Forward to this volume. We are grateful to INPARQUES for permission to conduct the surveys at Los Roques and to Fundación Científica Los Roques for providing access to their research facilities at Dos Mosquises. Judith Lang and Robert Ginsburg provided constructive comments and helpful advice during the preparation of the manuscript.

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Table 1. Site information for AGRRA stony coral and algal surveys in the Archipiélago de Los Roques National Park, Venezuela.

Site name	Site code	Reef type	Latitude (° N)	Longitude (° W)	Survey date	Depth (m)	Benthic transects (#)	≥10 cm stony corals		% live stony coral cover (mean ± sd)
								species (#)	colonies (#/10 m)	
<b>Shallow sites</b>										
Noronqui de Abajo	NOR	Patches(NE)	11 55.805	66 44.629	Oct 10 99	1.5	7	7	8	21.5 ± 9.5
Barrera Este	BE	Barrier (East)	11 47.540	66 53.714	Oct 8 99	1.5	7	8	12	18.5 ± 8.0
Boca de Sebastopol	BS	Barrier (East)	11 46.711	66 34.827	Oct 8 99	4	11	9	8	7.5 ± 6.5
Cayo Vapor	CV	Barrier (East)	11 57.257	66 37.360	Oct 9 99	4	6	9	9	9.0 ± 7.0
Crasqui - La Venada	CRV	Fringing (CE)	11 52.868	66 53.714	Oct 7 99	5.5	11	12	11	8.0 ± 9.0
Pelona de Rabusqui	PR	Fringing (CE)	11 52.934	66 41.342	Oct 7 99	6.5	9	13	12	25.5 ± 12.0
Boca Cote Somero	BCS	Barrier (South)	11 48.014	66 42.346	Oct 6 99	2.5	8	12	11	7.5 ± 5.0
Dos Mosquises Herradura	DMH	Fringing (SW)	11 48.014	66 53.051	Oct 6 99	2	8	10	15	27.5 ± 14.0
<b>All shallow sites (mean ± se)</b>						<b>3.5 ± 2.0</b>	<b>8.5 ± 2.0</b>	<b>10</b>	<b>11 ± 2.5</b>	<b>15.5 ± 8.5</b>
<b>Deep sites</b>										
Boca del Medio	BM	Barrier (East)	11 54.638	66 35.547	Oct 9 99	7.5	9	11	7	19.0 ± 6.5
Boca de Cote Profundo	BCP	Barrier (South)	11 45.982	66 42.346	Oct 6 99	12	6	16	18	57.0 ± 21.5
Cayo Sal Sur	CSS	Barrier (South)	11 44.134	66 50.858	Oct 5 99	9.5	8	17	17	30.5 ± 17.0
Punta Cayo Sal	PCS	Barrier (South)	11 44.529	66 51.463	Oct 5 99	13.5	5	11	13	54.0 ± 9.5
Dos Mosquises Sur	DMS	Fringing (SW)	11 47.540	66 53.714	Oct 5 99	12	7	13	15	60.0 ± 18.5
<b>All deep sites (mean ± se)</b>						<b>10 ± 3.0</b>	<b>7 ± 2.0</b>	<b>13.5</b>	<b>13 ± 3.0</b>	<b>44.0 ± 18.0</b>

Table 2. Size and condition (mean  $\pm$  standard deviation) of all stony corals ( $\geq 10$  cm diameter) by site in Los Roques National Park, Venezuela.

Site code <sup>1</sup>	Stony corals		Partial colony mortality (%)			Stony corals (%)			
	#	Diameter (cm)	Recent	Old	Total	Standing dead	Bleached <sup>2</sup>	Diseased <sup>3</sup>	with Damselfish
<b>Shallow sites</b>									
NOR	54	64.5 $\pm$ 101.5	6.0 $\pm$ 7.0	24.5 $\pm$ 33.5	30.5 $\pm$ 34.0	5.5	3.5 PB	0	26.0
BE	72	109.5 $\pm$ 102.0	2.0 $\pm$ 5.0	52.0 $\pm$ 45.5	53.5 $\pm$ 45.0	34.5	0	1.5 WP, 1.5 RB	9.0
BS	87	100.5 $\pm$ 98.0	2.0 $\pm$ 3.5	74.0 $\pm$ 43.0	74.5 $\pm$ 42.0	49.5	0	0	89.0
CV	55	127.0 $\pm$ 104.5	1.5 $\pm$ 3.0	55.5 $\pm$ 44.5	56.5 $\pm$ 44.5	43.5	0	2 YBD-like	1.5
CRV	118	106.0 $\pm$ 67.0	10.0 $\pm$ 16.0	80.0 $\pm$ 36.5	82.5 $\pm$ 33.5	64.5	1P, 1.7 PB	1.5 YBD	38.5
PR	101	57.0 $\pm$ 60.5	8.0 $\pm$ 15.0	28.5 $\pm$ 32.5	35.5 $\pm$ 34.0	11	2 PB	6 YBD	31.5
BCS	86	71.0 $\pm$ 73.0	8.0 $\pm$ 13.5	66.5 $\pm$ 40.5	70.0 $\pm$ 39.0	32.5	1 P, 3.5 PB	3.5 YBD-like	31.5
DMH	99	58.5 $\pm$ 60.5	4.5 $\pm$ 10.0	34.0 $\pm$ 37.0	37.0 $\pm$ 36.0	14.0	0	0	37.5
<b>All shallow sites (mean <math>\pm</math> se)</b>	<b>84 <math>\pm</math> 23</b>	<b>87.0 <math>\pm</math> 27.0</b>	<b>5.5 <math>\pm</math> 3.5</b>	<b>52.0 <math>\pm</math> 21.0</b>	<b>55.0 <math>\pm</math> 19.5</b>	<b>32.0 <math>\pm</math> 20.5</b>			<b>36.0 <math>\pm</math> 27.5</b>
<b>Deep sites</b>									
BM	65	128.0 $\pm$ 117.5	2.0 $\pm$ 5.0	39.5 $\pm$ 42.5	43.0 $\pm$ 42.5	16.5	3 P, 3 PB	0	29.0
BCP	101	78.0 $\pm$ 65.5	4.5 $\pm$ 11.0	17.0 $\pm$ 16.0	22.0 $\pm$ 20.0	0	1 P, 3 PB	3 YBD	20.0
CSS	101	38.0 $\pm$ 26.5	1.0 $\pm$ 2.5	28.0 $\pm$ 28.5	28.5 $\pm$ 28.5	4	0	0	0
PCS	61	78.5 $\pm$ 57.0	1.0 $\pm$ 2.0	23.0 $\pm$ 20.5	24.0 $\pm$ 20.5	0	3.5 P	3.5 UK	16.5
DMS	87	92.5 $\pm$ 89.5	4.5 $\pm$ 8.5	18.5 $\pm$ 17.5	23.5 $\pm$ 20.0	0	0	1 UK	20.5
<b>All deep sites (mean <math>\pm</math> se)</b>	<b>78.5 <math>\pm</math> 19</b>	<b>83.0 <math>\pm</math> 32.5</b>	<b>2.5 <math>\pm</math> 2.0</b>	<b>25.0 <math>\pm</math> 9.0</b>	<b>28.0 <math>\pm</math> 8.5</b>	<b>10.0 <math>\pm</math> 12.5</b>			<b>17.0 <math>\pm</math> 10.5</b>

<sup>1</sup>Site names corresponding to the site codes are given in Table 1.<sup>2</sup>P = pale; PB = partly bleached<sup>3</sup>WP = white plague; RB = red-band disease; YBP like = unknown disease resembling YBD; YBP = yellow-blotch disease; UK = cause of recent mortality unknown but disease suspected.

Table 3. Algal characteristics (mean  $\pm$  standard deviation), density of stony coral recruits and *Diadema antillarum* by site in the Archipiélago de Los Roques, National Park, Venezuela.

Site code <sup>1</sup>	Quadrats (#)	Relative abundance (%)		Crustose coralline algae		Macroalgal		Recruits (# /0.0625m <sup>2</sup> )	<i>Diadema</i> (# /100m <sup>2</sup> )
		Macroalgae	Turf algae	Turf algae	Crustose coralline algae	Height	Index <sup>3</sup>		
<b>Shallow sites</b>									
NOR	30	6.0 $\pm$ 10.5	76.0 $\pm$ 16.5	18.0 $\pm$ 13.5	1.5 $\pm$ 2.0	10	0	0	
BE	38	0	84.0 $\pm$ 15.5	16.0 $\pm$ 15.5	N <sup>2</sup>	N	0	17	
BS	45	1.0 $\pm$ 2.5	62.5 $\pm$ 24.5	36.5 $\pm$ 25.0	0.5 $\pm$ 1.0	<0.5	.33	0	
CV	20	3.5 $\pm$ 5.4	60.0 $\pm$ 25.5	36.5 $\pm$ 27.0	1.5 $\pm$ 1.5	5	0	0	
CRV	40	0.5 $\pm$ 1.0	53.5 $\pm$ 22.5	46.5 $\pm$ 22.0	<0.5 $\pm$ 0.5	<0.5	.1	2	
PR	30	0.5 $\pm$ 0.5	67.5 $\pm$ 25.0	32.5 $\pm$ 24.5	<0.5 $\pm$ 0.5	<0.5	.57	7	
BCS	15	29.0 $\pm$ 32.0	26.5 $\pm$ 25.5	44.5 $\pm$ 21.0	2.5 $\pm$ 1.0	70	.13	1	
DMH	30	1.5 $\pm$ 4.5	72.0 $\pm$ 30.0	26.5 $\pm$ 30.0	0.5 $\pm$ 1.5	1	.1	36	
<b>All shallow sites (mean <math>\pm</math> se)</b>	<b>31 <math>\pm</math> 10</b>	<b>5.5 <math>\pm</math> 9.8</b>	<b>63.0 <math>\pm</math> 17.5</b>	<b>32.0 <math>\pm</math> 11.3</b>	<b>~1.0 <math>\pm</math> 1.0</b>	<b>~12 <math>\pm</math> 26</b>	<b>.2 <math>\pm</math> .2</b>	<b>8 <math>\pm</math> 13</b>	
<b>Deep sites</b>									
BM	24	<0.05 $\pm$ 0.1	63.5 $\pm$ 26.5	36.0 $\pm$ 26.5	<0.5 $\pm$ 0.5	<0.5	0	0	
BCP	34	2.5 $\pm$ 5.5	76.0 $\pm$ 13.5	21.0 $\pm$ 13.5	1.0 $\pm$ 1.5	2.5	0	0	
CSS	19	0	37.5 $\pm$ 9.0	62.5 $\pm$ 9.0	N	N	0	0	
PCS	23	<0.05 $\pm$ <0.5	67.5 $\pm$ 8.0	32.5 $\pm$ 8.0	0.5 $\pm$ 0.5	<0.5	.22	0	
DMS	27	0	38.0 $\pm$ 6.5	62.0 $\pm$ 6.5	N	N	0	0	
<b>All deep sites (mean <math>\pm</math> se)</b>	<b>25 <math>\pm</math> 5.5</b>	<b>0.5 <math>\pm</math> 1.0</b>	<b>56.5 <math>\pm</math> 17.5</b>	<b>43.0 <math>\pm</math> 18.5</b>	<b>~0.5 <math>\pm</math> 0.5</b>	<b>~1 <math>\pm</math> 1</b>	<b>.04 <math>\pm</math> .1</b>	<b>0</b>	

<sup>1</sup>Site names corresponding to the site codes are given in Table 1.

<sup>2</sup>N = not present.

<sup>3</sup>Data from Villamizar et al. (this volume).

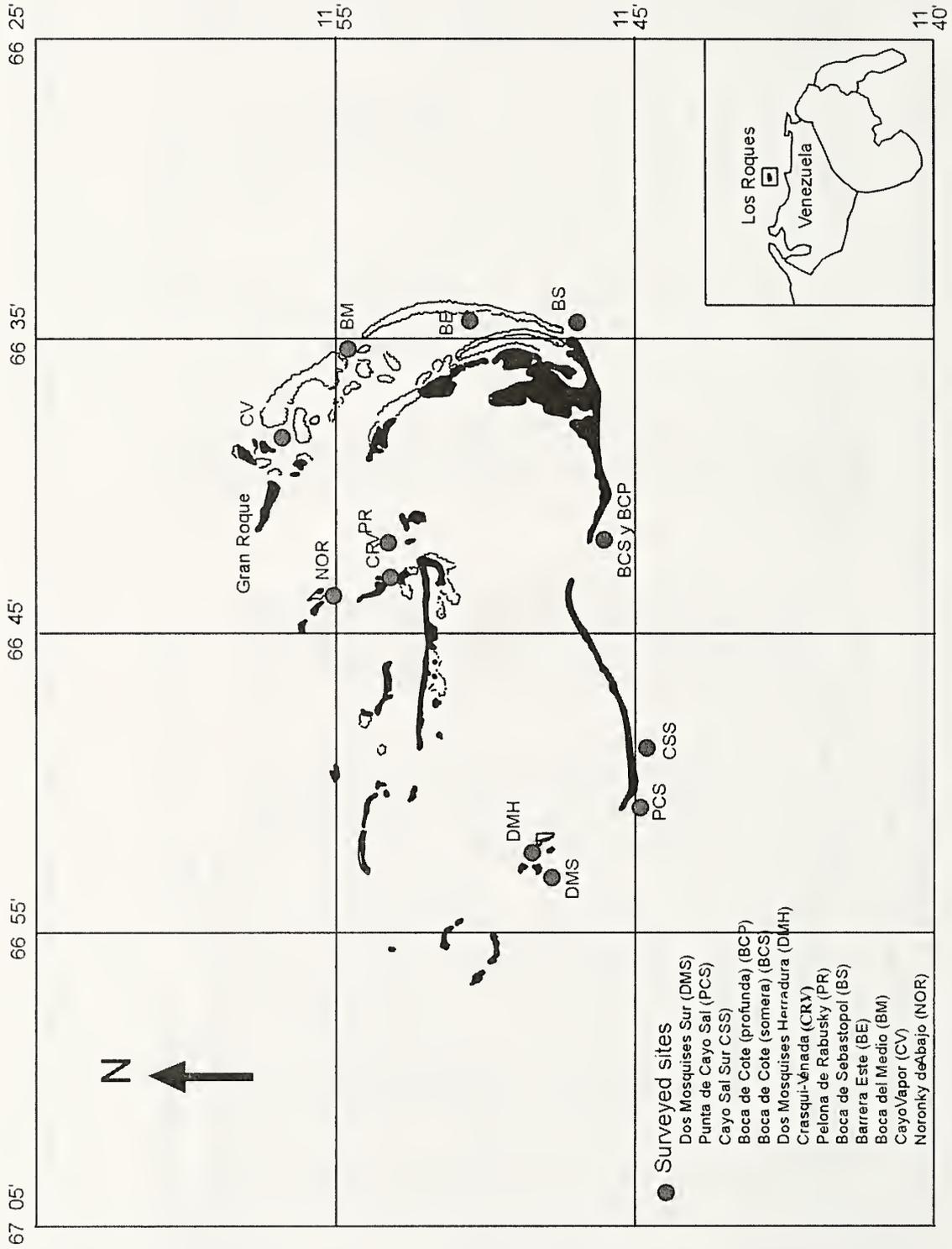


Figure 1. AGRRA survey sites in the Archipiélago de Los Roques National Park, Venezuela.

# RAPID ASSESSMENT OF CORAL REEFS IN THE ARCHIPIÉLAGO DE LOS ROQUES NATIONAL PARK, VENEZUELA (PART 2: FISHES)

BY

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## ABSTRACT

The reef fish community in Archipiélago de Los Roques National Park was evaluated by using the Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocol in 13 sites during October, 1999. *Scarus croicensis* was a dominant herbivore in eight sites. The density of key herbivores (scarids  $\geq 5$  cm and acanthurids) was higher in the shallower ( $< 8$  m) reefs than at depths of 8-12 m; commercially important carnivores (lutjanids, select serranids) were relatively scarce in the shallow eastern sites. There was a significant inverse relationship between total fish density and live stony coral cover. The 11-20 cm size class dominated the length frequency distributions of the key herbivores and carnivores. The Los Roques fish community appears to be in good condition overall and least disturbed anthropogenically in the southern barrier reef.

## INTRODUCTION

The Archipiélago de Los Roques National Park, established in 1972, is one of the oldest national marine parks in the Caribbean and is Venezuela's most important insular reef complex. Located 150 km north of the central Venezuelan coast, from  $11^{\circ} 44' - 11^{\circ} 58' N$  latitude and from  $66^{\circ} 32' - 66^{\circ} 57' W$  longitude (Fig. 1), it covers an area of approximately  $800 \text{ km}^2$ . The 42 cays and 200 sand banks collectively form an irregular oval which is delimited to the east and south by narrow barrier reefs (approximately 20 and 30 km long, respectively) that partially enclose a shallow lagoon with a mean depth of 4 m. The lagoonal floor is predominantly bare sand or mud with important expanses of seagrass and/or macroalgal beds and numerous patch reefs.

The archipelago's location makes it especially favorable for reef development for three reasons: 1) physical environmental stability is high because it is not subject to the direct impact of hurricanes; 2) anthropogenic disturbances are low because it is remote from the continental coast; and 3) water transparency is usually clear because it is remote

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from any significant runoff. As a consequence, the archipelago has as unusually diverse and luxuriant coral fauna (e.g., Méndez Baamonde, 1978; Cróquer and Villamizar, 1998; García, 2001).

The archipelago sustains an important finfish and shellfish fishery (i.e., 90% of the Venezuelan spiny lobster production) and tourist activities have increased during the last 10 years. In order to manage the rational use of its commercial and natural resources, the park area has been divided into several special-use zones, e.g., full protection ("no take"), recreation, scientific interest, while commercial and recreational fishing are allowed in the rest of the archipelago. The island of Gran Roque is the most densely populated region of the archipelago (approximately 750 inhabitants) and tourism activities are mostly concentrated there or at the nearest surrounding cays. Fishing activities are restricted to hook-and-line (hand and long lines) all year round and fish traps (spiny lobster season, Nov. 1 to Apr. 31). Skin diving is allowed only for the capture of spiny lobsters, and spearfishing is prohibited throughout the entire archipelago. Fish nets were totally banned in 1994.

Studies of the fish in the archipelago have been focused on the evaluation of the biological characteristics of several species, most of which are commercially important (e.g., Hauschild, 1984; Posada et al., 1988; Ortaz et al., 1996). A smaller but still valuable effort has been devoted to creating a complete fish inventory (e.g., Cervigón and Alcalá, 1997).

The purpose of the present study was to characterize the key elements of the fish community of the archipelago according to the protocol established by the AGRRA program (<http://coral.aoml.noaa.gov/agra>). The results will provide a baseline of the current status of the fish community which is essential for its proper management. In addition, the results are a valuable basis for regional comparisons and for evaluating subsequent changes in the park after future repeat assessments.

## METHODS

The selection of representative assessment sites in reefs where corals are abundant was based on the two senior authors' previous knowledge of the area along with advice from Patricia Kramer. Eight of the sites were in shallow water (1-7 m) and five were in depths of 8-12 meters (Table 1). Four sites were located in the fore reef of the southern barrier [Punta de Cayo Sal (PCS), Cayo Sal Sur (CSS), Boca de Cote Somero (BCS) and Boca de Cote Profundo (BCP)], while strong winds and currents forced us to restrict the four surveys of the eastern barrier [Boca de Sebastopol (BS), Barrera Este (BE), Boca del Medio (BM) and Cayo Vapor (CV)] to back-reef habitats (Fig. 1). Also surveyed were four fringing reefs surrounding small cays [Dos Mosquises Sur (DMS), Dos Mosquises Herradura (DMH), Crasquí-La Venada (CRV) and Pelona de Rabusquí (PR)] and one patch reef [Noronqui de Abajo (NOR)]. For a more detailed description of the sites and their benthos, see Villamizar et al. (this volume).

The field methodology closely followed the AGRRA Version 2.2 fish protocol (see Appendix One, this volume). All surveys were conducted by two trained observers (Posada and Alvarado). A total of 10 30 m x 2 m belt transects were swum in all reefs but one (PCS), for which there were nine transects (Table 1). Counts of serranids were

restricted to species of *Epinephelus* and *Mycteroperca*; scarids and haemulids less than 5 cm in length were not tallied. The surveys were conducted between 1000 and 1500 hours except for late afternoon assessments in the two westernmost reefs (DMS and DMH).

Consistency training was conducted for a day with natural populations prior to beginning the surveys to ensure agreement on fish sightings, identification, counts, and length estimation. Field identification of fishes was based on Humann (1994). Benthos surveys were conducted at the same time (see Villamizar et al., this volume) but were spaced sufficiently far away to avoid uncontrolled perturbations of fish behavior.

Data analysis followed the AGRRA protocol. All Spearman correlation analyses were performed at a significance level of  $p \leq 0.05$ . Statistical analyses were made using nonparametric tests due to non-normality of the data, even after transformation. Macroalgal index (% macroalgal relative abundance x macroalgal height in cm) was used as a proxy for macroalgal biomass (see Villamizar et al., this volume).

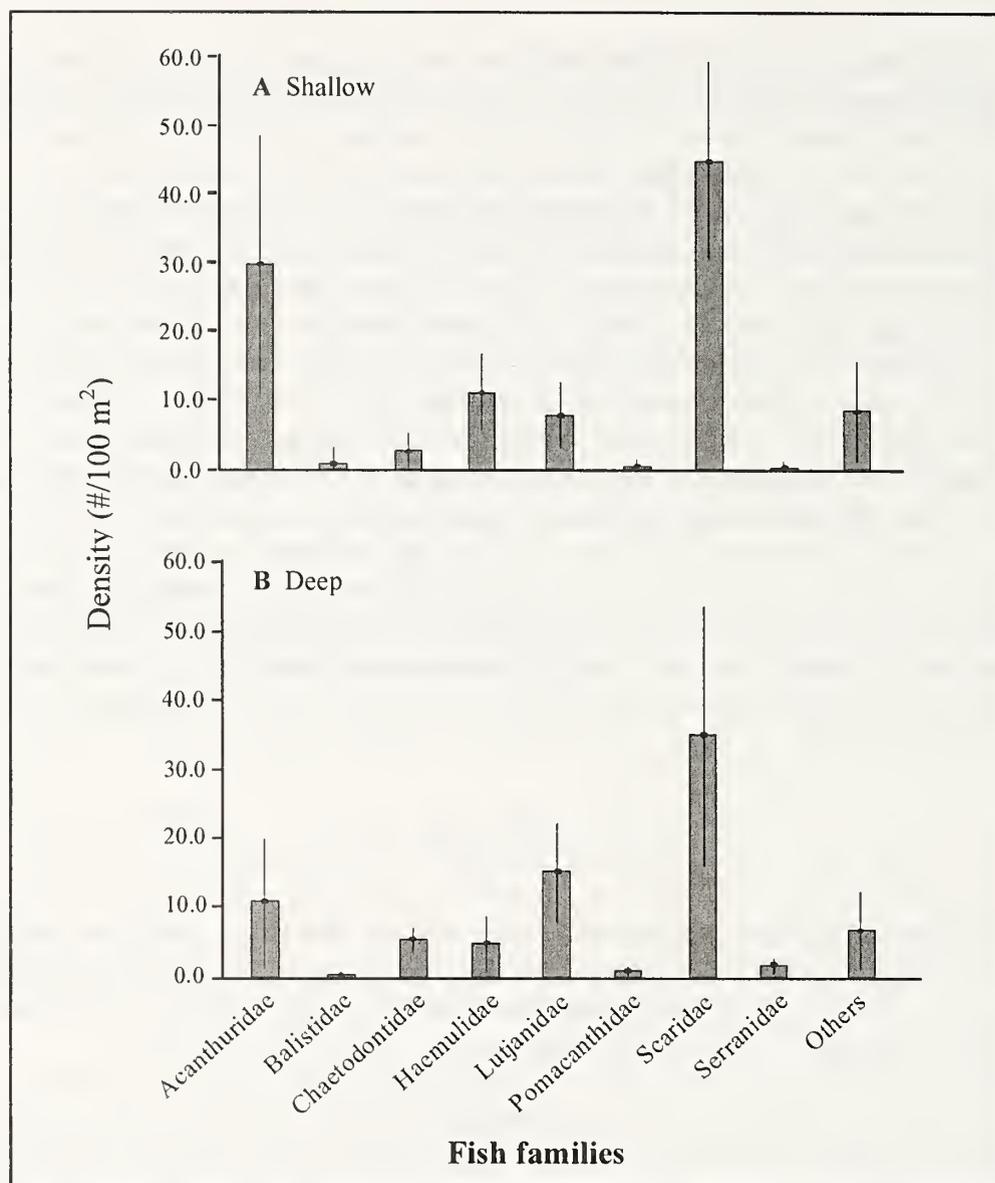
A multidimensional scaling analysis (MDS), based on the Bray-Curtis similarity index (Clarke 1993), was carried out to compare the archipelago's fish community structure on the basis of the relative abundance of the AGRRA fishes. Additional comparisons were made between our fish abundance data and that provided for Los Roques in the Reef Environmental Education Foundation (REEF)'s expert database on September 29, 1999.

## RESULTS

A total of 129 belt transects at 13 sites in the archipelago were made during October 4-10, 1999 (Table 1). Fifty-nine of the 70 species in our database of AGRRA fishes were observed during the belt-transect surveys, 11 of which were found at all sites: *Acanthurus bahianus*, *A. coeruleus*, *Haemulon flavolineatum*, *H. sciurus*, *Microspathodon chrysurus*, *Ocyurus chrysurus*, *Scarus croicensis*, *S. taeniopterus*, *S. vetula*, *Sparisoma aurofrenatum*, and *S. viride*. Two of the western reefs were most numerous in terms of AGRRA species (PCS and DMH with 36 each), while the fewest species were found in shallow reefs close to Gran Roque (NOR, BM and CV with 20, 22 and 23 species, respectively). Curiously, the reefs with the largest numbers of AGRRA species had relatively low fish densities (67.6 and 91.2 individuals/100 m<sup>2</sup>, respectively) and vice versa (Table 1).

In general, the community structure of the AGRRA fishes was dominated by herbivorous species (scarids, acanthurids and damselfishes). *Sparisoma viride* was present in 125 of the belt transects conducted during the current study (97%), followed by *Acanthurus coeruleus* (88%), *Microspathodon chrysurus* (83%), *Scarus vetula* (81%) and *S. croicensis* (81%). Carnivores were well represented by the zooplankton feeder, *Ocyurus chrysurus*, which was observed in 71% of the transects; the only large-sized serranid present in the surveys was *Mycteroperca bonaci*.

When the data for all sites were pooled, the average density by family was highest for scarids (41.0 individuals/100 m<sup>2</sup>) and acanthurids (22.5 individuals/100 m<sup>2</sup>) and lowest for balistids and pomacanthids (each 0.8 individuals/100 m<sup>2</sup>) and serranids (0.9 individuals/100 m<sup>2</sup>) (Table 2). Scarids clearly predominated at both depths (Fig. 2).

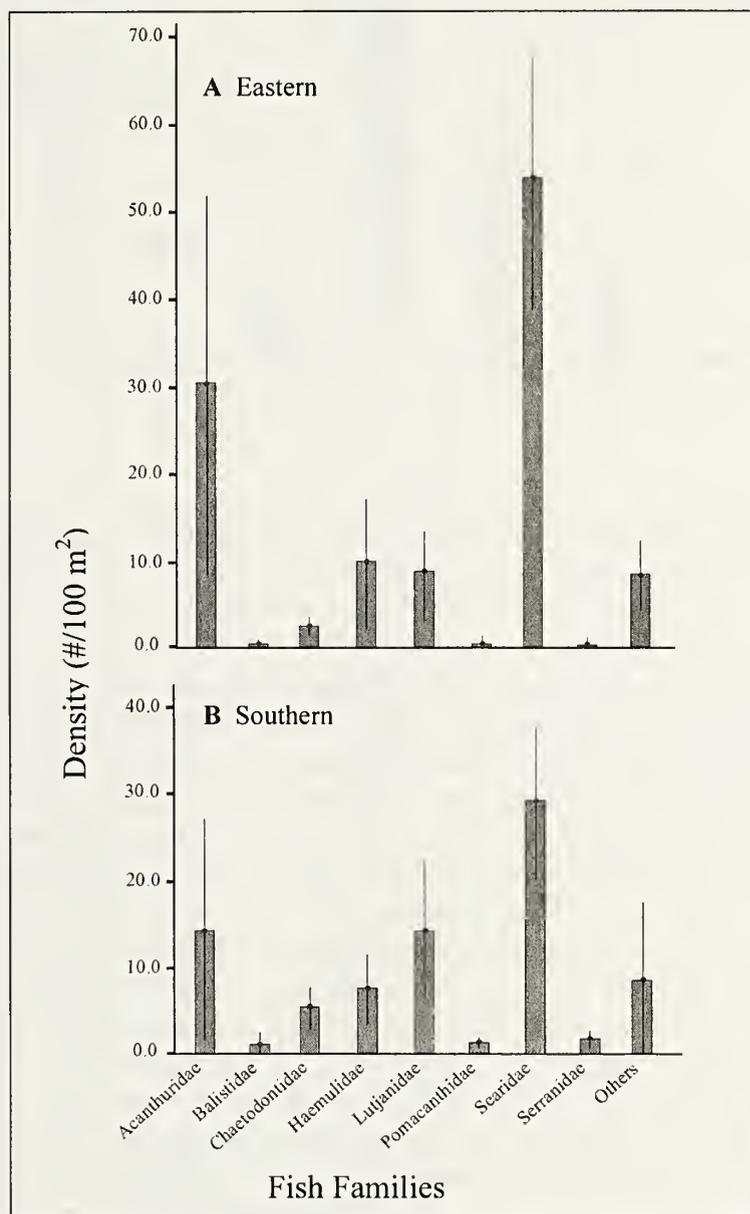


**Figure 2.** Mean fish density (# individuals/100 m<sup>2</sup> ± sd) for AGRRA fishes by depth in (A) shallow and (B) deeper sites in Los Roques, Venezuela. Others = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Microspathodon chrysurus*, and *Spyraena barracuda*.

However, both the relative abundance and the density of the scarids and acanthurids decreased with depth while the opposite trend was seen in the carnivorous serranids and lutjanids (Fig. 3). Overall, the densities of the AGRRA fishes (especially the herbivores) were highest in the reefs (many of which were shallow) located in or near the eastern barrier. Average fish densities were lower at the deeper reefs in the southern barrier where carnivorous lutjanids and serranids were more common.

*Scarus croicensis* was the numerical dominant in seven reefs, with a relative abundance that ranged between 11.0% (DMH) and 37.2% (DMS). *Acanthurus coeruleus* predominated at three shallow reefs (BCS, BE and NOR), with relative abundances of 19.1, 15.6 and 15.3%, respectively. *Microspathodon chrysurus* and *Acanthurus bahianus* showed the highest relative abundance in two sites (20.9% in BS and 22.4% in CV). In

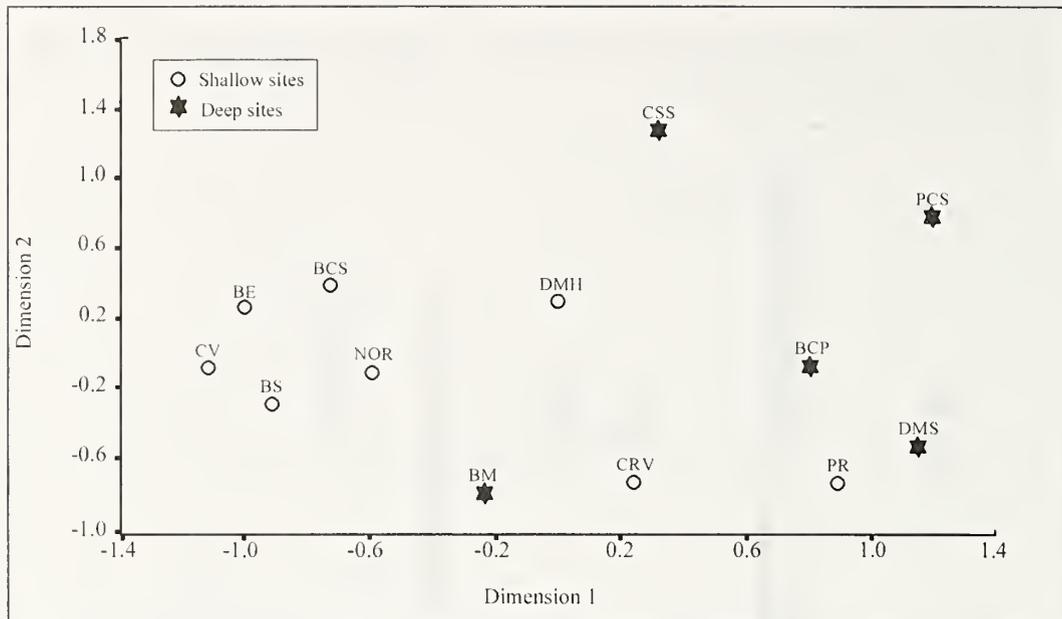
the only reef without a dominant herbivore (CSS), a lutjanid, *Ocyurus chrysurus*, had the highest relative abundance (18.4%).



**Figure 3.** Mean fish density (# individuals/100 m<sup>2</sup> ± sd) for AGRRA fishes by geographic location in (A) eastern and (B) southern sites in Los Roques, Venezuela. Others = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Microspathodon chrysurus* and *Spyraena barracuda*.

In the multidimensional ordination plot (Fig. 4), the fish communities in five (CV, BE, BCS, NOR and BS) of the shallow reefs clustered fairly well together and were separated from most (four/five) of the deeper reefs (CSS, PCS, BCP and DMS). Two of the deeper reefs (CSS and PCS), both located in the southern barrier, also appeared to form a separate subcluster.

An inverse, but significant correlation was observed between the total density of all the AGRRA fishes and coral reef complexity as represented by the percentage of live



**Figure 4.** Multidimensional ordination plot of AGRRA fish transect data in Los Roques, Venezuela.

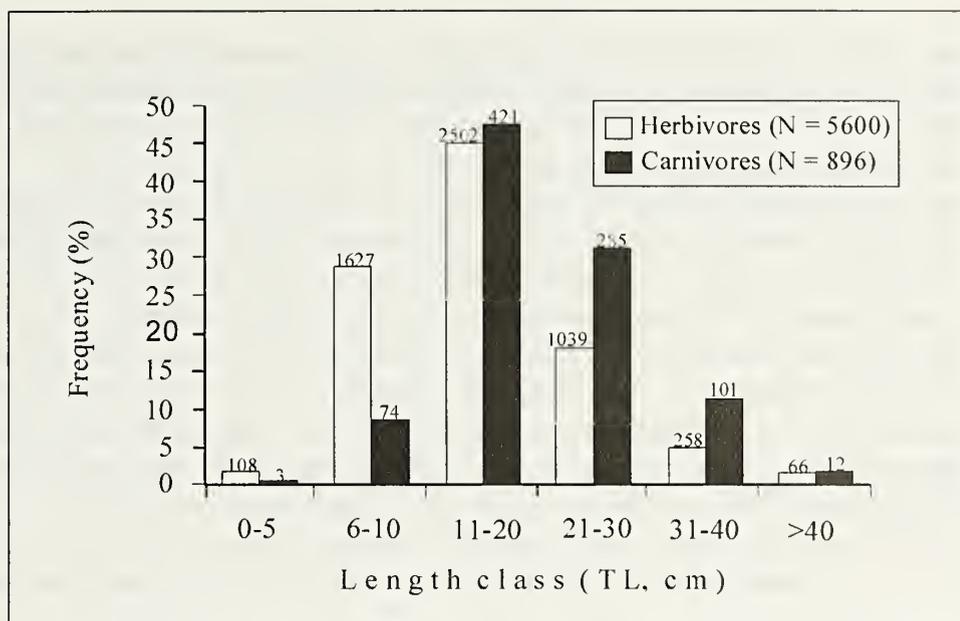
stony coral cover ( $r = -0.33$ ;  $p = 0.30$ ). However, there was no relationship between total AGRRA fish density and the diameter ( $r = 0.54$ ;  $p = 0.05$ ) or height ( $r = 0.29$ ;  $p = 0.37$ ) of the “large” ( $\geq 25$  cm diameter) stony corals nor between the ratio of these two parameters of reef rugosity ( $r = 0.36$ ;  $p = 0.24$ ). Nor was there any relationship between the total density of the key AGRRA herbivores and macroalgal index ( $r = 0.33$ ;  $p = 0.30$ ).

Significant correlations were found between the density of individual fish families and several coral reef parameters. There was a clear inverse relationship between the density of acanthurids and total live stony coral cover ( $r = -0.61$ ;  $p = 0.026$ ) as well as the number of species ( $r = -0.84$ ;  $p = 0.01$ ) and density (as number/10 m) ( $r = -0.69$ ;  $p = 0.009$ ) of the large stony corals. Lutjanid density was significantly correlated with the number of large coral species ( $r = 0.69$ ;  $p = 0.01$ ) while the density of serranids was significantly correlated with large coral density ( $r = 0.893$ ;  $p < 0.0001$ ) and total live stony coral cover ( $r = 0.80$ ;  $p = 0.0009$ ).

Size frequency distributions of the AGRRA carnivores (*Epinephelus*, *Mycteroperca* and all lutjanids) and herbivores (scarids  $\geq 5$  cm, acanthurids, *Microspathodon chrysurus*) are shown in Figure 5. The size class that was 11-20 cm in total length (TL) dominated the length frequency distribution when all species of AGRRA fishes in all sites were grouped. Similar results were observed when subsets of the data were examined by depth stratum or reef type.

## DISCUSSION

The list of species included in our AGRRA database appears to be adequate for a rapid characterization of the key elements of the Los Roques reef fish community (Table 4). For example, nearly 50% of the 59 species observed in the present belt-transect study were included in the 50 most common species listed in the REEF geographic report for Los Roques in September, 1999.



**Figure 5.** Size frequency distribution of herbivores (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) and carnivores (all lutjanids, select serranids) in Los Roques, Venezuela.

Cervigón and Alcalá (1997) listed an accumulated total of 307 fish species for the Los Roques archipelago, similar to the 306 species reported from 547 Bonaire expert REEF surveys and clearly above the 270 species found in 777 Key Largo expert REEF fish counts (<http://www.reef.org>). The high fish diversity found recently by REEF surveys at Los Roques archipelago (217 species, expert data), despite the relatively low effort invested (106 surveys), is an unequivocal sign of an intact fish community.

Herbivores that were widespread in the AGRRA surveys (*S. viride*, *A. coeruleus*, *M. chrysurus*, *S. vetula*, *S. croicensis*) have previously been reported as common and highly abundant in both coastal and insular Venezuelan reef systems (Cervigón, 1993, 1994), while *Ocyurus chrysurus* represents the major finfish target in a fishery that, at Los Roques, is oriented towards spiny lobsters (Posada and Brunetti, 1988). Scarids and acanthurids are lightly harvested since the prohibitions against trapping, spearing, and netting of reef fishes are enforced by local authorities. In consequence, the archipelago may be considered to be a nearly unfished reef fish population.

The predominance of herbivorous species in 12 of the 13 Los Roques assessment sites can be associated with good water quality (high transparency) which facilitates benthic algal photosynthetic activities. However, abundance of herbivorous fishes declines, as expected, with depth (Choat, 1991) as the abundance of carnivores increases. That one reef (CSS) was dominated by a carnivore (*Ocyurus chrysurus*) is likely due to its proximity to the reef edge and the strong currents, full of transient zooplankton, of the southern barrier.

A large number of fish species is often related to a complex coral reef topography (Roberts and Ormond 1987). However, contradictory results were observed between fish density and different measurements of reef complexity examined at Los Roques. While significant inverse relationships were observed between total AGRRA fish density and the percentage of live stony coral cover (which is not directly related to topography), there were no relationships with other indicators of reef rugosity (i.e., diameter or height

of the large corals, or the ratio of these two parameters). Probably none of these simple parameters are adequate indicators of optimum fish habitat and/or other processes (food availability, water quality, wave action, recruitment, community interactions; Williams, 1991) are simultaneously influencing fish abundance.

In our assessments, total fish density varied between 70.8 and 124 individuals/100 m<sup>2</sup>. These values are below those for quantitative fish surveys reported by Alvarado (2000) at Morrocoy National Park (101-287 individuals/100 m<sup>2</sup>), a protected coral reef located in Venezuela's central coastal zone. However, because Alvarado (2000) modified the AGRRA protocol to include all species of the Pomacentridae and Labridae, the difference between these two estimates is not considered significant. Further comparisons between these two national parks could be enlightening since the two areas differ in the degree to which each is exposed to perturbations. In contrast to Los Roques, Morrocoy National Park is located in a region highly influenced by river runoff and by a greater influence of anthropogenic activities and intense fishing pressure. In addition, its coral reefs were affected in 1996 by an as-yet-undetermined mass mortality event which, in some areas, has decreased the percentage of live stony coral cover from 40% to 1% and increased the proportion of macroalgae from 11% to 55% (Losada and Klein, unpublished manuscript).

Overall fish densities for the Scaridae, Acanthuridae, and Serranidae were very similar in Los Roques (Table 2) and Morrocoy (39.0, 22.0 and 1.0 individual/100 m<sup>2</sup>, respectively; Alvarado, 2000). However, lutjanids were significantly more abundant in Los Roques than in Morrocoy (10.7 individuals versus 3 individuals/100 m<sup>2</sup>, respectively). The low proportion of carnivorous species in the Morrocoy National Park fish community can be attributed to more intense fishing pressure and its smaller area.

Fish community structure was also dominated by herbivorous fishes in the Morrocoy National Park. However, whereas *Scarus croicensis* dominated in all eight of Alvarado's (2000) survey sites at Morrocoy, dominance in the Los Roques archipelago was distributed among several species (*Scarus croicensis*, *Acanthurus bahianus* and *A. coeruleus*) and families (Scaridae, Acanthuridae). In Kenya, the predominance of just one species of scarid has been linked by McClanahan (1994) to areas not protected from fishing activities.

While the 11-20 cm TL size class dominated the size frequency distributions in most of the fish families surveyed at both Los Roques and Morrocoy National Parks (Alvarado, 2000), there was a slight tendency for proportionately higher numbers of individuals to occur in the smaller (6-10 cm TL) size class in Morrocoy. Size classes above 20 cm TL were better represented in the Los Roques data, particularly for commercial species of carnivores. Differences in size structure between these two populations can be attributed to fishing pressure as it is well known that the larger individuals in a population tend to be targeted by most fishing techniques used to capture coral reef species (Russ, 1991).

In general terms, the reef fish communities in the Archipiélago de Los Roques National Park appear to be in a healthy state (highly diverse community, an abundance of individuals, some large-sized species, a good balance between herbivores and carnivores, etc.) in close concurrence with the good conditions of most of the coral reef sites visited during the present study (see García, 2001; Villamizar et al., this volume). Overall, the key elements of the fish community were more balanced in the deeper southern barrier and southwestern fringing reefs. Fewer commercially important carnivorous fishes were found close to Gran Roque and the eastern barrier. The cause(s) of this disparity could be

related to natural differences associated with geographic location or depth, or could possibly be the result of anthropogenic impacts from the islands. It is recommended that special attention be given to these northeastern and eastern areas in any future studies.

Thanks to its effective protection by the Instituto Nacional de Parques (INPARQUES), the Los Roques archipelago provides one of the few opportunities in the wider Caribbean to examine a minimally disturbed fish community. This feature is of crucial importance since it provides a baseline standard against which population parameters may be measured in areas that are fished. According to H. Choat and R. Robertson (personal communication), there are few reef systems in the Caribbean with comparable high abundances of large herbivorous and carnivorous fishes. Hence, the Los Roques fish populations should be periodically monitored (at least biannually) in order to become aware of any declines. Research would also be needed to determine the factor(s) that are potentially responsible for these changes and to respond with a contingency plan. Rapid assessments, as provided by the AGRRA protocols, are very useful in countries where economic support for long-term monitoring studies is limited.

### ACKNOWLEDGMENTS

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Table 1. Site information for AGRRA fish surveys in the Archipiélago de Los Roques National Park, Venezuela.

Site name	Site code	Reef type	Latitude (° ' N)	Longitude (° ' W)	Survey date	Depth (m)	≥10 cm stony corals (#/10 m) <sup>1</sup>	% live stony coral cover (mean ± sd) <sup>1</sup>	Macroalgal index <sup>1</sup>	30 m fish transects (#)	AGRRA fish species (#)	
<b>Shallow sites</b>												
Noronqui de Abajo	NOR	Patches (NE)	11 55.805	66 44.629	Oct. 10 99	1	8	21.5 ± 9.5	10	10	20	
Barrera Este	BE	Barrier (East)	11 47.540	66 53.714	Oct. 8 99	1	12	18.5 ± 8.0	N <sup>2</sup>	10	28	
Boca de Sebastopol	BS	Barrier (East)	11 46.711	66 34.827	Oct. 8 99	7	8	7.5 ± 6.5	0.5	10	32	
Cayo Vapor	CV	Barrier (East)	11 57.257	66 37.360	Oct. 9 99	1	9	9.0 ± 7.0	5	10	23	
Crasqui – La Venada	CRV	Fringing (CE)	11 52.868	66 53.714	Oct. 7 99	5	11	8.0 ± 9.0	<0.5	10	27	
Pelona de Rabusqui	PR	Fringing (CE)	11 52.934	66 41.342	Oct. 7 99	6	12	25.5 ± 12.0	<0.5	10	28	
Boca de Cote Somero	BCS	Barrier (South)	11 45.982	66 42.346	Oct. 6 99	1	11	7.5 ± 5.0	70	10	32	
Dos Mosquises Herradura	DMH	Fringing (SW)	11 48.014	66 53.051	Oct. 6 99	1	15	27.5 ± 14.0	1	10	36	
<b>Deep sites</b>												
Boca del Medio	BM	Barrier (East)	11 54.638	66 35.547	Oct. 9 99	8	7	19.0 ± 6.5	<0.5	10	22	
Boca de Cote Profundo	BCP	Barrier (South)	11 45.982	66 42.346	Oct. 6 99	9	15	36.5 ± 34.5	2.5	10	27	
Cayo Sal Sur	CSS	Barrier (South)	11 44.134	66 50.858	Oct. 5 99	8	17	30.5 ± 17.0	N	10	29	
Punta Cayo Sal	PCS	Barrier (South)	11 44.529	66 51.463	Oct. 5 99	12	13	54.0 ± 9.5	<0.5	9	36	
Dos Mosquises Sur	DMS	Fringing (East)	11 47.540	66 53.714	Oct. 5 99	8	15	60.0 ± 18.5	N	10	34	

<sup>1</sup>From Villamizar et al. (this volume).<sup>2</sup>N = no macroalgae present at this site.

Table 2. Mean density (± standard deviation) of AGRRA fishes, by depth in the Archipiélago de Los Roques National Park, Venezuela.

Sites	Density (#/100 m <sup>2</sup> )							Total AGRRA		
	Key herbivores		Key carnivores			Total AGRRA				
	Acanthuridae	Scaridae (≥5cm)	Microspathodon chrysurus	Haemulidae (≥5cm)	Lutjanidae	Serranidae <sup>1</sup>	Chaetodontidae	Pomacanthidae	Balistidae	Total AGRRA
All shallow	29.7 ± 18.65	45.0 ± 14.34	7.6 ± 7.19	11.3 ± 5.59	7.9 ± 4.61	0.4 ± 0.45	2.8 ± 2.28	0.5 ± 0.61	0.9 ± 1.59	857.2
All deep	11.0 ± 9.50	34.8 ± 19.33	5.9 ± 5.01	5.0 ± 4.10	15.3 ± 7.42	1.8 ± 1.25	5.4 ± 2.02	1.3 ± 0.66	0.5 ± 0.35	408.5
<b>All sites</b>	<b>22.5 ± 17.96</b>	<b>41.0 ± 16.47</b>	<b>6.8 ± 16.47</b>	<b>8.9 ± 5.82</b>	<b>10.7 ± 6.69</b>	<b>0.9 ± 1.05</b>	<b>3.4 ± 2.47</b>	<b>0.8 ± 0.71</b>	<b>0.8 ± 1.25</b>	<b>1265.7</b>

<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.

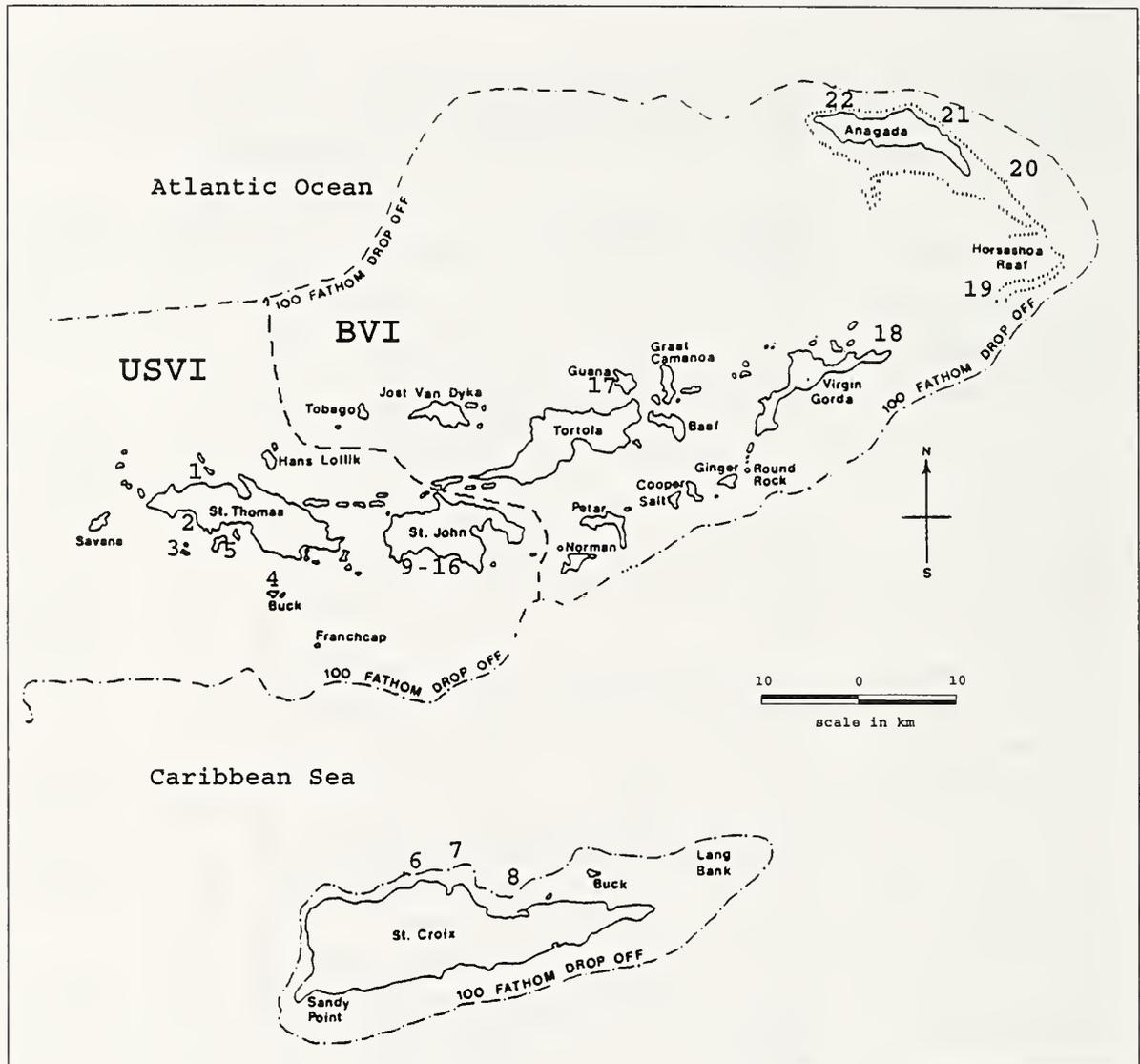


Table 4. Fifty most frequently sighted fish species in September, 1999 (REEF geographic data) for the Archipiélago de Los Roques National Park, Venezuela, with densities (mean  $\pm$  standard deviation) for species recorded in belt transects (present study).

Scientific name	Sighting frequency (%) <sup>2</sup>	Density (#/100m <sup>2</sup> )	Scientific name	Sighting frequency (%)	Density (#/100m <sup>2</sup> )
<i>Acanthurus bahianus</i> (*) <sup>1</sup>	97	7.4 $\pm$ 8.5	<i>Chaetodon striatus</i> (*)	80	0.6 $\pm$ 0.5
<i>Sparisoma viride</i> (*)	95	9.0 $\pm$ 3.6	<i>Lactophrys triqueter</i>	79	
<i>Ocyurus chrysurus</i> (*)	94	5.7 $\pm$ 4.6	<i>Coryphopterus personatus/hyalinus</i>	78	
<i>Acanthurus coeruleus</i> (*)	94	11.1 $\pm$ 8.4	<i>Epinephelus cruentatus</i> (*)	77	0.6 $\pm$ 1.0
<i>Stegastes partitus</i>	93		<i>Haemulon sciurus</i> (*)	77	1.3 $\pm$ 1.1
<i>Halichoeres garnoti</i>	93		<i>Haemulon plumieri</i> (*)	77	0.2 $\pm$ 0.5
<i>Chromis multilineata</i>	92		<i>Gramma loreto</i>	76	
<i>Haemulon flavolineatum</i> (*)	92	3.0 $\pm$ 2.2	<i>Scarus taeniopterus</i> (*)	75	2.8 $\pm$ 3.2
<i>Chromis cyanea</i>	92		<i>Caranx ruber</i> (*)	74	0.4 $\pm$ 0.6
<i>Chaetodon capistratus</i> (*)	92	2.9 $\pm$ 2.1	<i>Abudefduf saxatilis</i>	73	
<i>Aulostomus maculatus</i>	92		<i>Acanthurus chirurgus</i> (*)	72	3.9 $\pm$ 3.7
<i>Thalassoma bifasciatum</i>	91		<i>Holocentrus rufus</i>	72	
<i>Holacanthus tricolor</i> (*)	91	0.4 $\pm$ 0.5	<i>Epinephelus fulvus</i> (*)	71	0.2 $\pm$ 0.3
<i>Sparisoma aurofrenatum</i> (*)	89	2.3 $\pm$ 1.6	<i>Scarus croicensis</i> (*)	70	15.4 $\pm$ 12.9
<i>Clepticus parrae</i>	87		<i>Sphyræna barracuda</i> (*)	66	0.1 $\pm$ 0.2
<i>Mulloidichthys martinicus</i>	87		<i>Cantherhines macrocerus</i>	66	0.1 $\pm$ 0.2
<i>Serranus tigrinus</i>	86		<i>Inermia vittata</i>	64	
<i>Lutjanus apodus</i> (*)	84	1.9 $\pm$ 1.8	<i>Halichoeres maculipinna</i>	63	
<i>Lutjanus mahogoni</i> (*)	82	1.8 $\pm$ 1.8	<i>Haemulon chrysargyreum</i>	62	3.8 $\pm$ 3.1
<i>Myripristis jacobus</i>	82		<i>Stegastes planifrons</i>	62	
<i>Scarus vetula</i> (*)	82	8.1 $\pm$ 6.2	<i>Scarus coeruleus</i> (*)	59	0.5 $\pm$ 1.1
<i>Bodianus rufus</i> (*)	82	0.5 $\pm$ 0.5	<i>Coryphopterus lipernes</i>	58	
<i>Canthigaster rostrata</i>	82		<i>Caranx latus</i>	57	
<i>Coryphopterus glaucofraenum</i>	81		<i>Scarus coelestinus</i> (*)	56	0.4 $\pm$ 0.6
<i>Microspathodon chrysurus</i> (*)	80	7.0 $\pm$ 6.5	<i>Hypoplectrus puella</i>	56	

<sup>1</sup>Asterisks indicate the species is listed in the AGRRA protocol.

<sup>2</sup>The sighting frequencies for Los Roques as posted at the REEF expert database in September 1999.



**Figure 1.** AGRRA survey sites in the Virgin Islands. (1) Caret Bay, (2) Brewer's Bay, (3) Flat Cay, (4) Buck Island, (5) Sprat Bay, (6) Cane Bay, (7) Salt River, (8) Long Reef, (9) Fish Bay east outer, (10) Fish Bay west outer, (11) Fish Bay east inner, (12) Great Lameshur Donkey, (13) Great Lameshur VIERS, (14) Fish Bay west inner, (15) Great Lameshur Tektite, (16) Great Lameshur Yawzi, (17) Iguana head, (18) Eustatia Reef, (19) Herman's Reef, (20) Horseshoe Reef, (21) Jack Bay, (22) West Cow Wreck.

# A RAPID ASSESSMENT OF CORAL REEFS IN THE VIRGIN ISLANDS (PART 1: STONY CORALS AND ALGAE)

BY

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## ABSTRACT

The Atlantic and Gulf Rapid Reef Assessment (AGRRA) benthos protocol was conducted in depths of 3-14 m at 22 coral reef sites in the U.S. Virgin Islands (St. Croix, St. Thomas, St. John) and the British Virgin Islands (Anegada, Guana Island and Virgin Gorda). Live stony coral cover averaged between 10% and 35% in 85% percent of the sites. The size of colonies  $\geq 25$  cm in diameter averaged 55 cm and their composition was dominated by the genus *Montastraea*. Coral recruitment varied considerably among sites and was dominated by species that brood their larvae. Nearly all sites had stony corals that were affected by disease, bleaching or damaged by fish bites. Mean values for total (recent + old) partial mortality exceeded 40% of colony surfaces in eight sites and were between 20% and 40% for the remainder. The abundance of “standing dead” stony corals was typically less than 1.5%. The relative abundance of macroalgae exceeded 30% in 15 of 22 sites and macroalgae were dominated by *Dictyota*.

## INTRODUCTION

The Virgin Islands (18°20' N, 64°50' W) lie between two major island archipelagoes: the Greater Antilles to the west and the Lesser Antilles to the southeast. With the exception of St. Croix, the northern United States Virgin Islands (USVI), together with the British Virgin Islands (BVI) and the islands of Puerto Rico, rise from a geological shelf that is surrounded by deep water (Dammann, 1969). This shelf covers approximately 3,200 km<sup>2</sup>, contains about 500 km of shelf edge and is generally less than 100 m in depth. St. Croix sits on a similar, but smaller, shelf that is separated from the

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northerly Virgin Islands by the 4,685 m deep, 60 km wide Virgin Islands trough.

Upwelling of deep, nutrient-rich waters from the North Equatorial and Caribbean Currents, coupled with the fact that these shelves lie within the photic zone, aid in the development of deep (45 m) and shallow (2 to 20 m) fringing reefs, bank barrier reefs and patch reefs (Dammann, 1969). St. Thomas, St. John and St. Croix, the three main islands of the USVI, are surrounded by more than 90 uninhabited islands, cays, and rocks which provide hard substrata suitable for the growth of coral reefs (Towle, 1970, unpublished report; Dammann and Nellis, 1992). The three main islands of the BVI, Tortola, Virgin Gorda and Anegada, are also surrounded by nearly 50 smaller islands, cays and rocks. With the exception of Anegada and St. Croix, the Virgin Islands are of volcanic origin. Anegada is a low coral island (Dunne and Brown, 1976, unpublished report) and St. Croix has a sedimentary and carbonate origin (Hubbard, 1989; R. Watlington, personal communication). Many of the reefs in the Virgin Islands are associated with mangrove forests, seagrass beds and algal plains, which are important to reef-associated fishes.

Over the past 20 years, eight major hurricanes, numerous outbreaks of disease, and sporadic bleaching events have caused extensive coral mortality to the coral reefs surrounding the Virgin Islands (Gladfelter, 1982; Edmunds and Whitman, 1991; Rogers et al., 1991; Causey et al., 2000). Recovery from these natural disturbances is hindered by a multitude of human impacts that affect coral reefs such as overfishing, ship groundings, anchor damage and non-point source pollution (Roberts, 1993; Sebens, 1994; Rogers and Garrison, 2001). Moreover, rapid development of inland and coastal areas has dramatically increased soil erosion and sedimentation onto many of these coral reefs (Rogers, 1990; MacDonald et al., 1997; Anderson and MacDonald, 1998; Ramos, 1998, unpublished report). The cumulative effects of these human impacts reduce coral abundance, diversity, and larval recruitment and may make corals more susceptible to disease and bleaching (Nemeth and Sladek Nowlis, 2001).

In response to these threats to coral reefs, Ginsburg et al. (1996) initiated a process of rapid reef assessment. After development of the AGRRA protocols, a Caribbean-wide effort to assess the condition of coral reefs throughout the region was launched. The University of the Virgin Islands' Center for Marine and Environmental Studies joined the effort and set out to assess the reefs of the Virgin Islands using the AGRRA protocols. This paper reports on the initial findings of our benthic assessments. Results of the fish surveys are given in Nemeth et al. (this volume).

## METHODS

Site-selection criteria in the USVI and the BVI varied among the different islands but most (18/22) choices were made for strategic reasons. In St. Thomas, five sites were selected based on their inclusion in a long-term, sedimentation monitoring project, their proximity to the University of the Virgin Islands McLean Marine Science Center and the presence of historical data (i.e., Rogers, 1982, unpublished report; Nemeth and Sladek Nowlis, 2001). Three popular recreational diving sites were selected in St. Croix, one of which is within the Salt River Bay National Historic Park and Ecological Preserve. The eight sites in St. John were part of a study comparing sedimentation rates between Great

Lameshur Bay, which is within the National Park Boundary, and Fish Bay, which has been experiencing heavy development within its watershed. Four of these sites were shallow reefs (<6 m) located inside the bays and four sites were in deeper (>6 m) reefs located outside the bays. Four of the six sites surveyed in the BVI (Iguana Head in Guana Island, Eustatia Reef in Virgin Gorda, West Cow Wreck Bay and Herman's Reef in Anegada) were selected haphazardly. Eustatia Reef is a heavily visited dive site whereas Guana Island is a privately owned island with a low human population. The other two sites in Anegada (Jack Bay and Horseshoe Reef, a designated protected area) were selected because of historical surveys conducted by Dunne and Brown (1976, unpublished report) and the West Indies Laboratory (1983, unpublished report). The reefs of Anegada were included in the AGRRA survey to provide a remote reference site with low human population and little landmass. A qualitative assessment of human and natural impacts at the 22 sites surveyed in the Virgin Islands is given in Appendix A (this paper).

The AGRRA Version 2.0 benthos protocol (see Appendix One, this volume) was used. Four of us constituted the primary dive team and we were augmented by five alternates. All divers participated in at least one three-day training session of the AGRRA protocol. Training sessions were conducted in the spring of 1998, 1999, and 2000 and consistency training was used for the primary dive team at least once a year. All corals were identified to species except species of *Agaricia*. Although *A. agaricites* was a dominant species, it was not distinguished from other species of *Agaricia*. Coral and algal identification guides included Humann (1993) and Littler et al. (1989). Stony coral sizes were measured to the closest cm. We also recorded the percentage of individually surveyed ( $\geq 25$  cm diameter) colonies with parrotfish or damselfish bites. Fish-bite damage from parrotfish or damselfish was distinguished when possible. Macroalgal heights were measured to the closest 0.5 cm. When scoring the relative abundance of crustose coralline algae, divers typically removed sediment with vigorous handsweeping motions. Algal turfs were omitted from the July 2000 assessment of Anegada and Virgin Gorda sites in accordance with the May 2000 revisions of the AGRRA benthos protocol. Because of these changes, percent relative abundances of macroalgae and crustose coralline algae are not comparable to earlier surveys. Therefore, percent absolute abundance of macroalgae in the quadrats is presented to make comparisons among the sites.

Data were summarized by island groups within three geographic areas to examine general trends in coral reef condition throughout the Virgin Islands. The geographic areas were: 1) Anegada; 2) the shallow and deeper reefs of the remaining islands in the northern Virgin Islands (NVI); and 3) St. Croix. Anegada was considered a geographic unit because of its isolation from the other Virgin Islands and its unique geology (low coral island). St. Croix was considered a geographic unit also because of its isolation from the NVI, its unique geology (sedimentary/carbonate) and because it is completely within the Caribbean Sea. The NVI (USVI = St. Thomas, St. John; BVI = Guana, Virgin Gorda) were grouped because of their close proximity to one another, their similar geologic origins and topographies (high volcanic islands) and their exposure to both Atlantic waters from the north and Caribbean waters from the south. The shallow sites in St. John were analyzed separately from the deeper sites within the NVI archipelago. Comparisons of coral cover among "island/depth groups" (hereafter referred to as island groups) were made using single factor Analysis of Variances (ANOVA). Prior to statistical analysis, residuals of coral cover data were graphically analyzed for normality

using Systat statistical analysis software. Coral cover data were found suitable for statistical analysis at a significance level of  $\alpha = 0.05$ . Tukey HSD was used for post-hoc multiple comparison analyses.

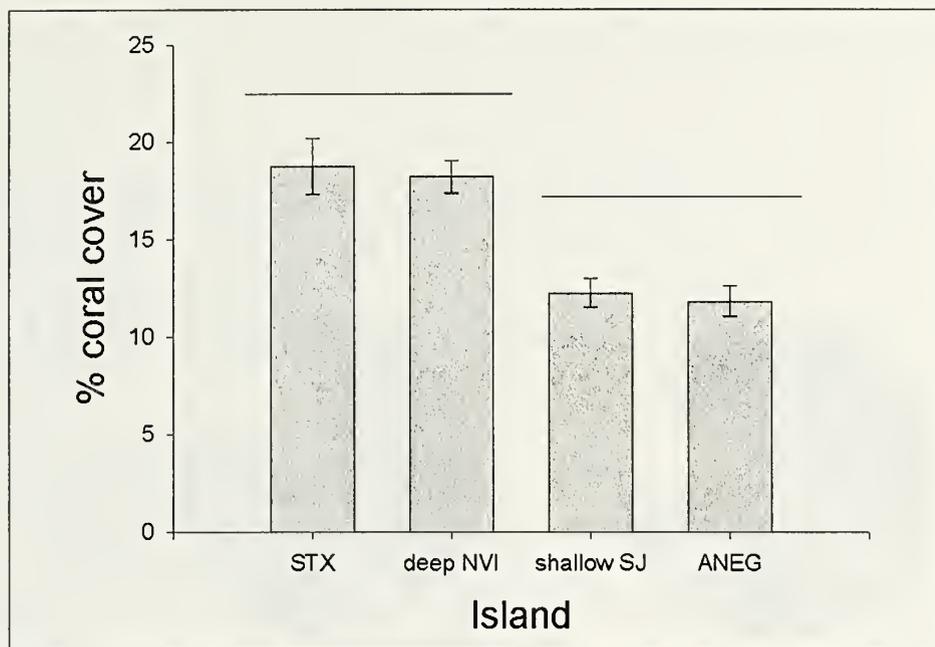
## RESULTS

### Site Characteristics

A total of 22 sites in eight islands or cays (Fig. 1) were surveyed during three time periods: May 1998; May to December 1999; and February to October 2000 (Table 1). All of the surveyed sites were in fringing reefs at depths that ranged from 3 to 14 m (Table 1). The fringing reefs in St. Thomas, St. John, Guana Island and Virgin Gorda were all located close to the shoreline and sloped relatively steeply from their crests to their bases where the reefs gave way to sandy substrata. In St. Croix, the reefs were within a few hundred meters of the Virgin Islands trough and dropped off steeply into abyssal depths. In the low island of Anegada, the fringing reefs sloped more gently than in the high volcanic NVI. All of the deeper reef surveys were carried out on their seaward slopes with the exception of Herman's Reef in Anegada which was on the leeward side of the reef crest. Most of the sites that were surveyed were relatively sheltered from the prevailing seas (i.e., from exposure to northeast swells). Sites with direct exposure to northeast swells included Jack Bay, Horseshoe Reef, and West Cow Wreck Bay in Anegada. The three reefs in St. Croix were also exposed to swells although the Virgin Islands archipelago 60 km to the north provided some protection. Reefs with moderate protection from prevailing seas included Herman's Reef in Anegada, Caret Bay in St. Thomas, and Eustatia Reef in Virgin Gorda.

Over 85% of the sites surveyed had between 10% and 35% live stony coral cover (Table 1) with means for the island groups ranging between 12% and 19% (Fig. 2). The lowest cover occurred in Fish Bay, St. John whereas the deeper Tektite site in Great Lameshur Bay, St. John had exceptionally high (nearly 50%) live coral cover (Table 1). Significant differences in percent coral cover occurred among island groups (ANOVA:  $F_{3, 341} = 12.4$ ,  $P < 0.001$ ). Anegada and the shallow St. John sites were significantly lower in coral cover than sites in the deeper NVI and St. Croix groups (Fig. 2).

The "large" stony corals that were individually surveyed (i.e., those  $\geq 25$  cm in diameter) were numerically dominated by the *Montastraea annularis* species complex (*M. annularis*, *M. faveolata*, *M. franksi*) in all island groups except St. Croix which was dominated by *M. cavernosa* (Fig. 3). The deeper NVI and shallow St. John sites were similar in overall composition and quite different from those in St. Croix and Anegada, which also differed from each other (Fig. 3). The second most common taxon was *Siderastrea siderea*, except in Anegada where *Porites astreoides* and *Diploria strigosa* were each twice as common as *S. siderea*. Other individually surveyed taxa that were each less than three percent in abundance in the four island groups are as follows: shallow St. John (*Millepora alcicornis*, *Agaricia*, *Colpophyllia natans*, *Diploria labyrinthiformis*, *D. strigosa*, *Stephanocoenia intersepta*, *Solenastrea bournoni*); deeper NVI (*M. alcicornis*, *Acropora cervicornis*, *A. palmata*, *Dichocoenia stokesi*, *Madracis mirabilis*, *M. decactis*, *S. intersepta*, *Dendrogyra cylindrus*, *Solenastrea bournoni*, *D.*



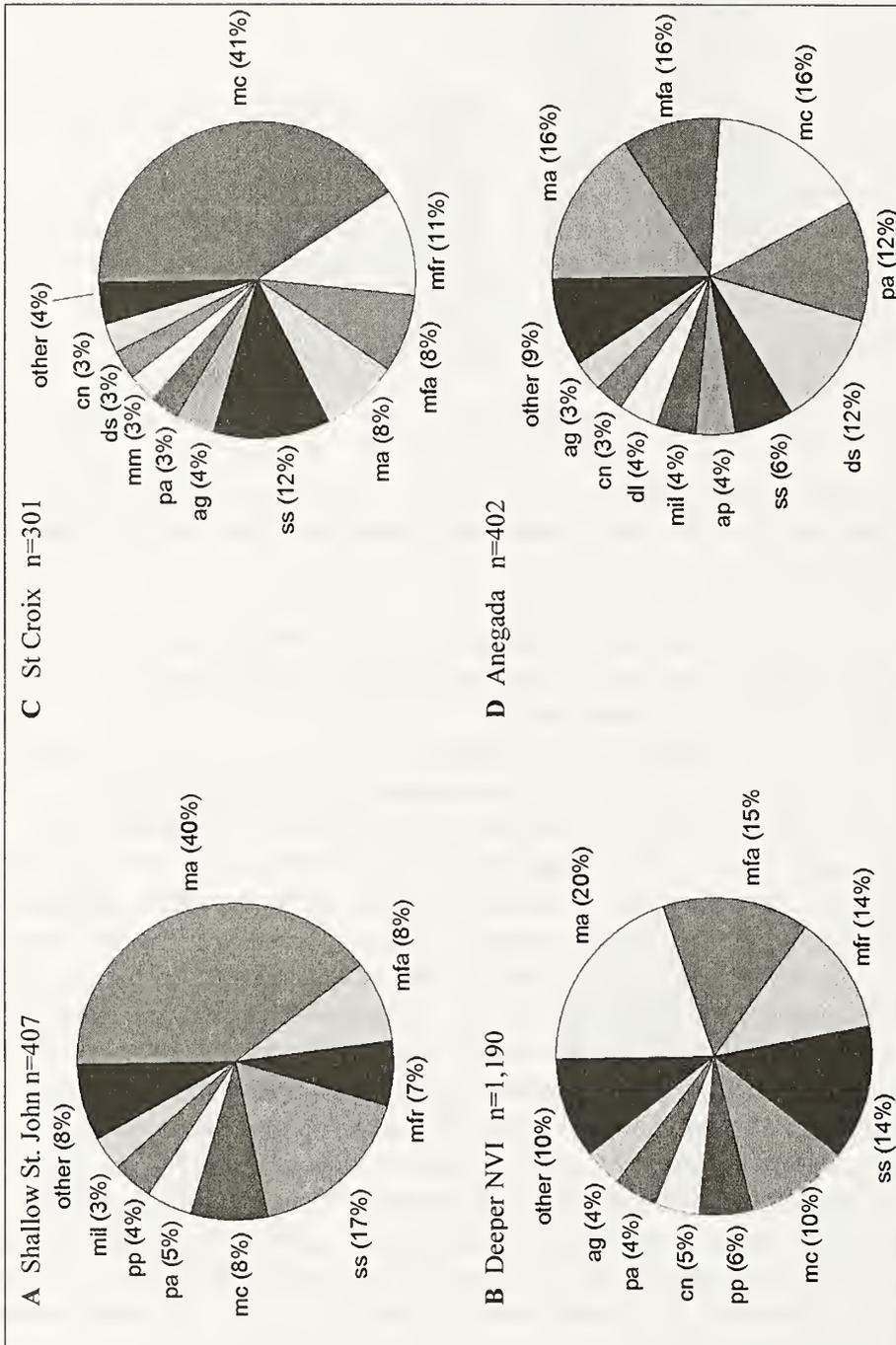
**Figure 2.** Mean live stony coral cover (percent  $\pm$  se) in St. Croix, deeper NVI (St. Thomas, St. John >6 m deep, Guana Is., Virgin Gorda), shallow St. John and Anegada. Lines connecting bars indicate no significant difference ( $\alpha = 0.05$ ).

*labyrinthiformis*, *D. strigosa*); St. Croix (*D. cylindrus*, *D. labyrinthiformis*, *M. decactis*, *Porites porites*); and Anegada (*P. porites*, *M. franksi*, *A. cervicornis*, *D. stokesi*, *S. intersepta*, *D. clivosa*, *D. cylindrus*, *M. mirabilis*).

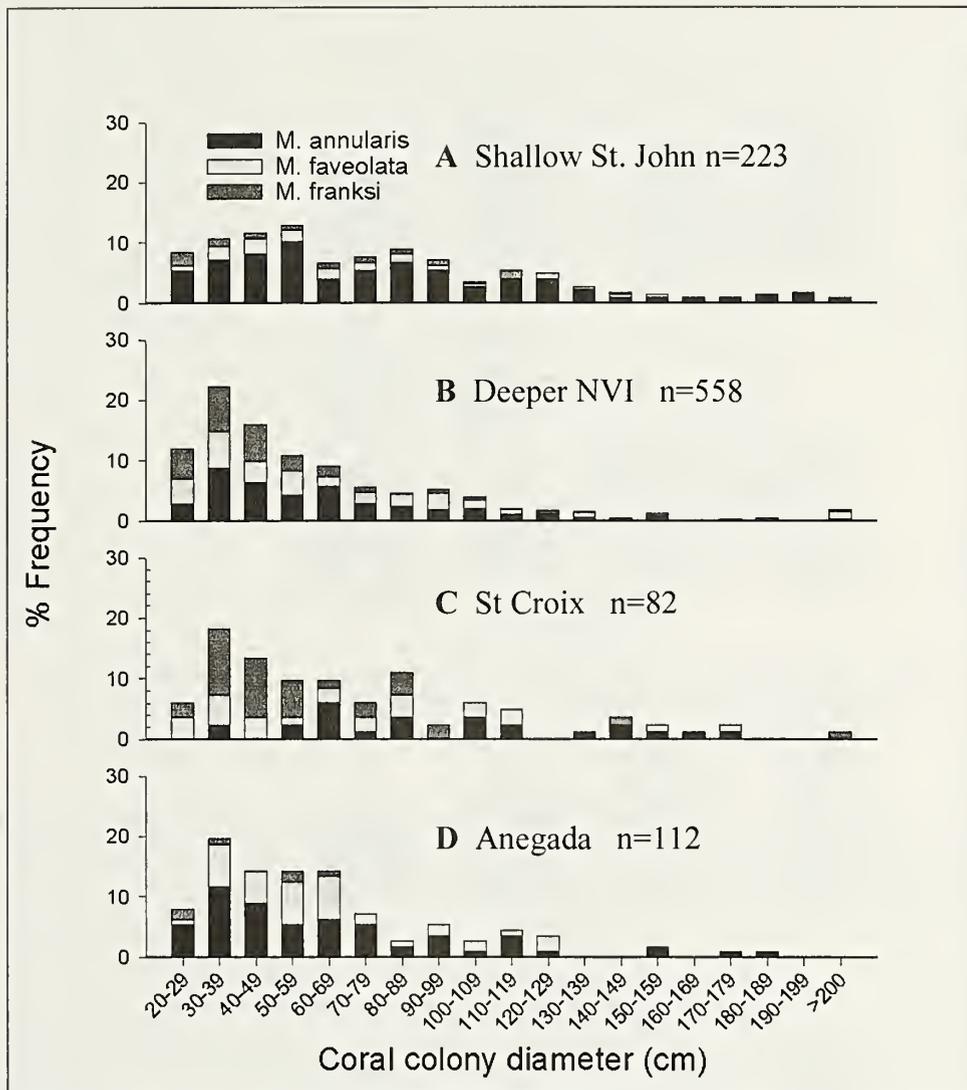
The diameter of the individually surveyed corals (Table 2) averaged 55.0 cm with exceptionally large (up to 260 cm) colonies of *M. annularis* occurring off St. John. Within the *Montastraea annularis* species complex, the size frequency distribution in most sites showed a positive skew toward smaller colonies of <50 cm diameter (Fig. 4). The shallow St. John sites were unique in that they showed a platykurtic size-frequency distribution (Fig. 4) even though the local abundance of the *M. annularis* species complex as a whole was similar to that of the deeper NVI sites (Fig. 3). Three of the four shallow sites off St. John had very large coral colonies and so exerted a proportionately greater influence in this group than did the two deeper Great Lameshur sites in the much larger set of the deeper NVI group. The mean sizes of the predominant *Montastraea* varied within and among the island groups (Fig. 5). *M. annularis* and *M. faveolata* were significantly larger than *M. franksi* and *M. cavernosa* in the deeper NVI ( $F_{3,678} = 13.72$ ,  $p < 0.001$ ) whereas, in St. Croix, *M. faveolata* and *M. franksi* were significantly smaller than *M. annularis* and larger than *M. cavernosa* ( $F_{3,201} = 22.47$ ,  $p < 0.001$ ). In the shallow reefs of St. John, *M. cavernosa* was significantly smaller than *M. annularis* ( $F_{3,251} = 3.64$ ,  $p < 0.01$ ) whereas *M. faveolata* and *M. franksi* were intermediate in size. All *Montastraea* on Anegada were similar in size (Fig. 5).

### Stony Coral Condition

Coral bleaching was recorded in all sites (Table 2) with the highest average values for large stony corals occurring in St. Croix (48% in October/December 1999,  $n=3$

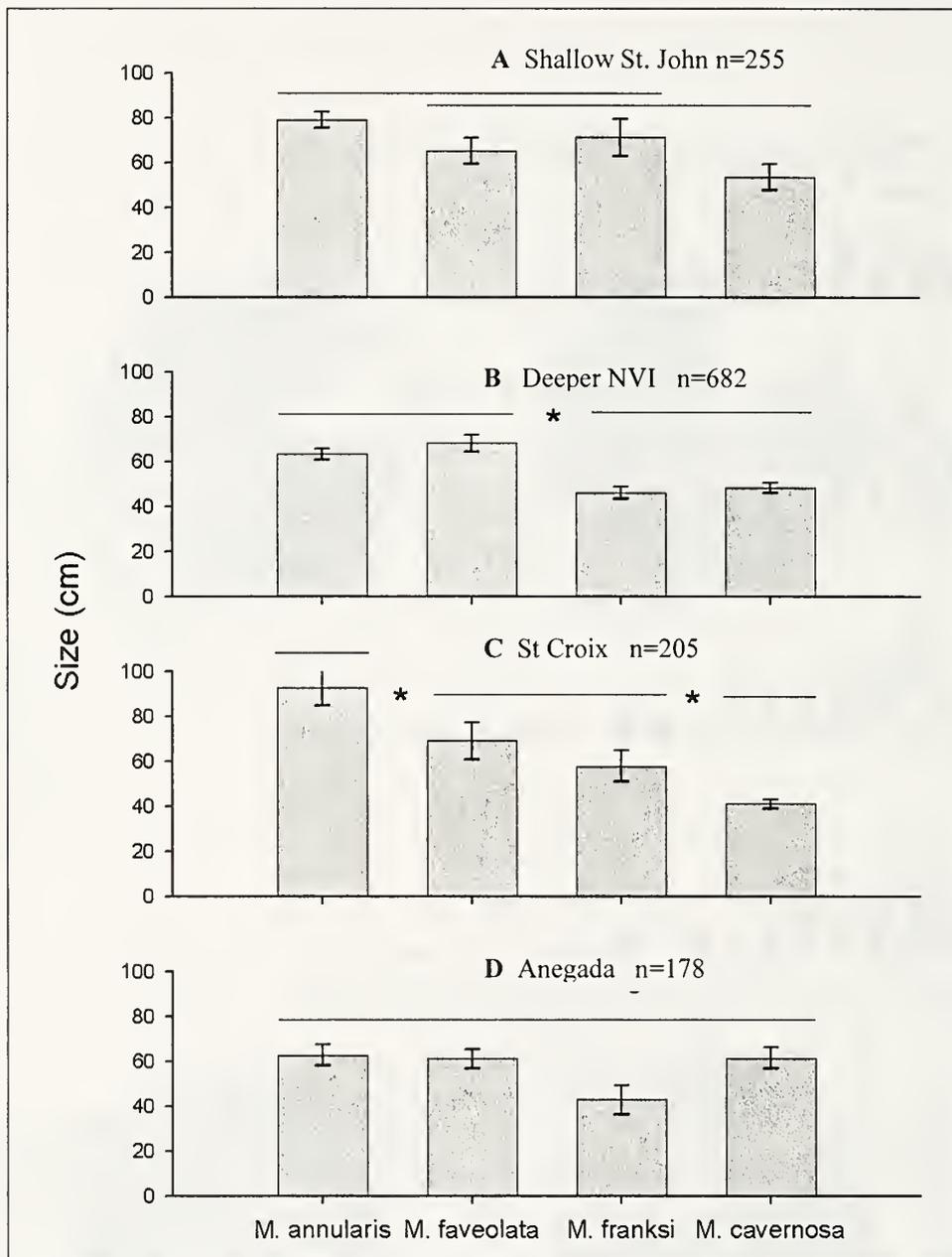


**Figure 3.** Species composition and mean relative abundance of all stony corals ( $\geq 25$  cm diameter) in (A) shallow St. John, (B) deeper NVI, (C) St. Croix, and (D) Anegada. ap = *Acropora palmata*, ag = *Agaricia*, cn = *Colpophyllia natans*, dl = *Diploria labyrinthiformis*, ds = *D. strigosa*, mm = *Madracis mirabilis*, mil = *Millepora alaicornis*, ma = *Montastraea annularis*, mfa = *M. faveolata*, mfr = *M. franksi*, mc = *M. cavernosa*, pa = *Porites astreoides*, pp = *P. porites*, ss = *Siderastrea siderea*, other = all species with  $<3\%$  abundance.



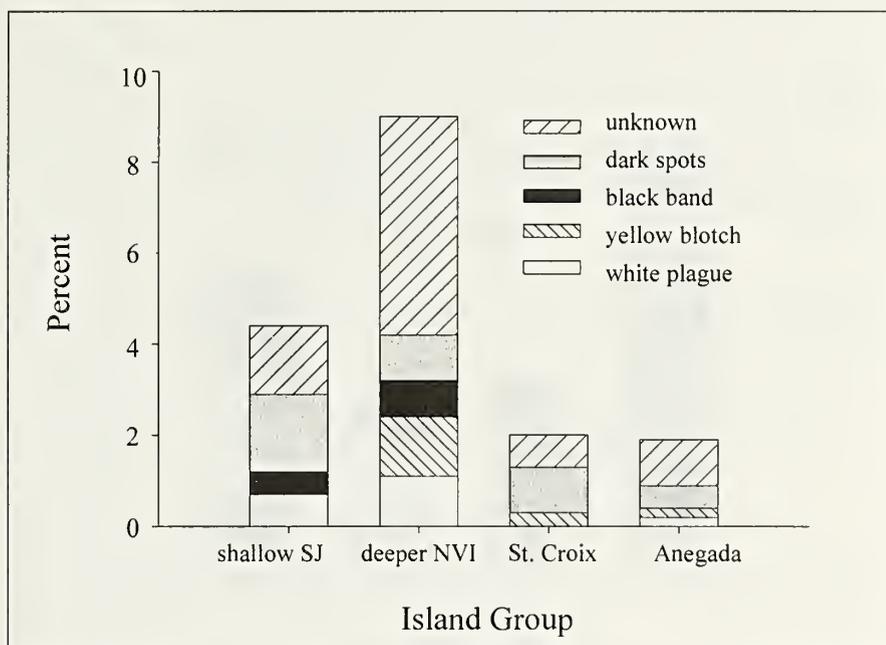
**Figure 4.** Size-frequency distribution of all colonies ( $\geq 25$  cm diameter) of *Montastraea annularis* (ma), *M. faveolata* (mfa) and *M. franksi* (mfr) in (A) shallow St. John (ma=162, mfa=34, mfr=27), (B) deeper NVI (ma=237, mfa=178, mfr=143), (C) St. Croix (ma=24, mfa=24, mfr=34), and (D) Anegada. (ma=64, mfa=42, mfr=6).

sites), in two shallow St. John sites (~38% in October/November 1999), in St. John's deeper Tektite Reef (36%, in August/November 1999) and in Anegada (28% in July 2000, n=4 sites). Occasional temperature measurements during surveys documented that normal sea-surface temperatures to 10 m depth were coolest in February (25° C), began to warm during June (26° C) and July (27° C), peaked in August (28-29° C), and began to cool during October and November (27° C). Water temperatures deeper than 12 m could be up to 1.5° C cooler than in shallower depths. The Caribbean-wide 1998 bleaching event was first detected in September 1998 in the Virgin Islands when surface sea-water temperatures exceeded 30° C. Bleaching of coral colonies peaked in October 1998 but nearly all signs of bleaching had disappeared by February 1999 (Nemeth and Sladek Nowlis, 2001). Since no surveys were conducted from June 1998 to May 1999, the 1998 bleaching event did not directly influence the results presented in this report.



**Figure 5.** Mean size of *Montastraea annularis* (ma), *M. faveolata* (mfa), *M. franksi* (mfr) and *M. cavernosa* (mc) in **(A)** shallow St. John (ma=162, mfa=34, mfr=27, mc=32), **(B)** deeper NVI (ma=237, mfa=178, mfr=143, mc=124), **(C)** St. Croix (ma=24, mfa=24, mfr=34, mc=123), and **(D)** Anegada (ma=64, mfa=42, mfr=6, mc=66). Lines connecting bars indicate no significant differences ( $\alpha=0.05$ ) in average size among species.

Signs of disease in the individually surveyed stony corals were present in most (20/22) of our assessment sites (Table 2). Divers were able to recognize four general disease types: black band, yellow blotch, white plague, and dark spots (Fig. 6). The species most susceptible to disease included *Montastrea faveolata*, *M. franksi*, *M. cavernosa*, *M. annularis*, *Colpophyllia natans* and *Siderastrea siderea*. Of particular interest, white-band disease was observed in one colony of *Acropora cervicornis* each in Caret Bay, St. Thomas and in Herman Reef, Anegada. The percent of diseased corals

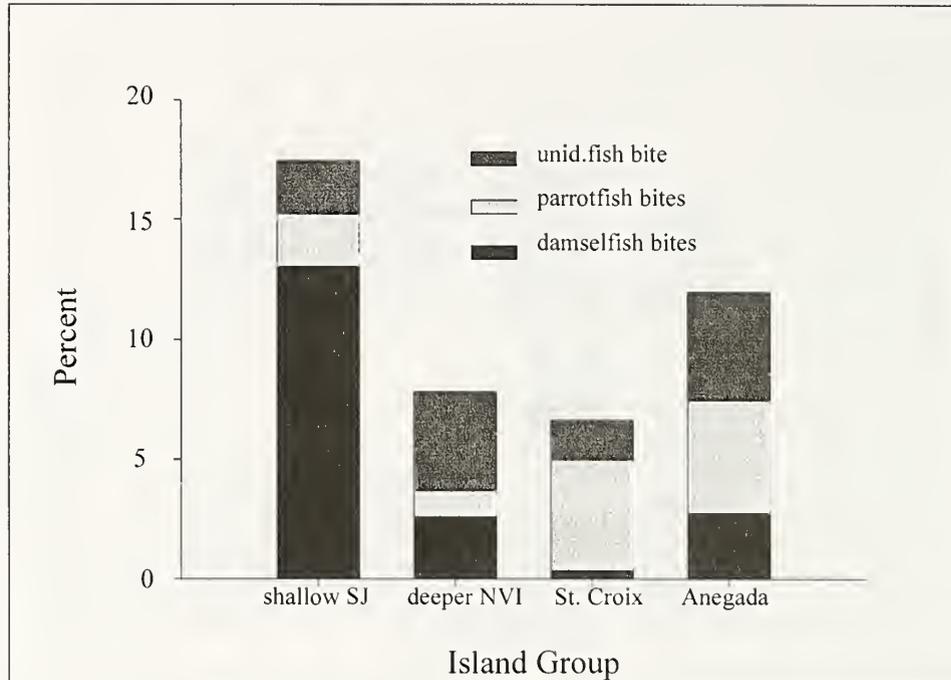


**Figure 6.** Percent damage to all stony corals ( $\geq 25$  cm diameter) from disease in shallow St. John ( $n=18$  infected corals), deeper NVI ( $n=106$  infected corals), St. Croix ( $n=6$  infected corals), and Anegada ( $n=8$  infected corals).

varied among sites within island groups with St. Croix (in October and December 1999) and Anegada (in July 2000) having lower levels of disease than the shallow St. John sites (in October/November 1999 and May/August/October 2000) and deeper NVI sites (various dates between May 1998 and July 2000). Five percent or more of the colonies were affected at 13 sites (maximum being 24% in Guana Island), 9 of which included surveys that had been made between May and November, 1999. At 4 sites in St. Thomas, sampling dates either occurred both in 1998 and 1999 (Buck Island), in 1998 and 2000 (Caret Bay), or in 1999 and 2000 (Brewers Bay, Flat Cay) (Table 1). In the first two cases no diseases were observed in 1998 but they were present in all sites on the other survey dates. The incidence of disease was considerably higher in 1999 than in 2000 for Brewer's Bay.

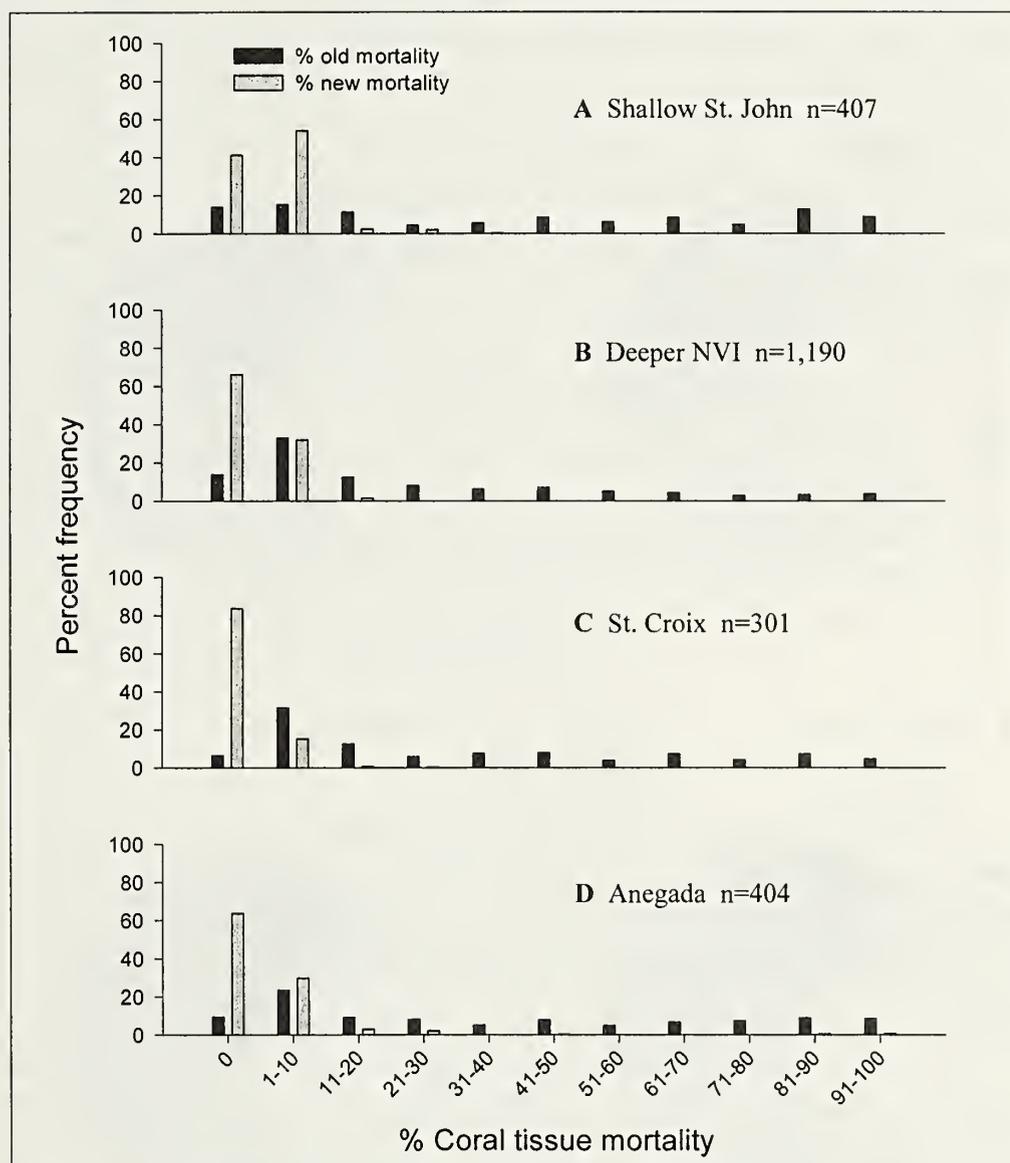
Damsel fish and parrot fish bites were responsible for causing tissue damage to 25% or more of the individually surveyed stony corals in three sites and were present in colonies at all but three sites (Table 2; Fig. 7). The most frequently attacked species included *M. annularis*, *M. faveolata*, *M. franksi*, *C. natans*, *M. cavernosa* and *P. porites*. The shallow sites in Great Lameshur Bay, St. John had the greatest percentage of tissue damage from damselfish. This was largely due to the high density of three-spot damselfish (*Stegastes planifrons*) on colonies of *M. annularis*. Damselfish overall occupied 35.4% of the stony corals in the shallow St. John sites, 7.1% in the deeper NVI, 3.2% in Anegada and 2.0% in St. Croix. Tissue damage by parrotfish was greatest in St. Croix and Anegada (Fig. 7).

Mean values for total (recent and old) partial mortality exceeded 40% of colony surfaces in eight sites and were between 20% and 40% in the remaining sites (Table 2). Recent partial mortality of colony surfaces averaged less than three percent overall in



**Figure 7.** Percent damage to all stony corals ( $\geq 25$  cm diameter) from fish bites in shallow St. John (71 corals with bites), deeper NVI (60 corals with bites), St. Croix (20 corals with bites), and Anegada (48 corals with bites).

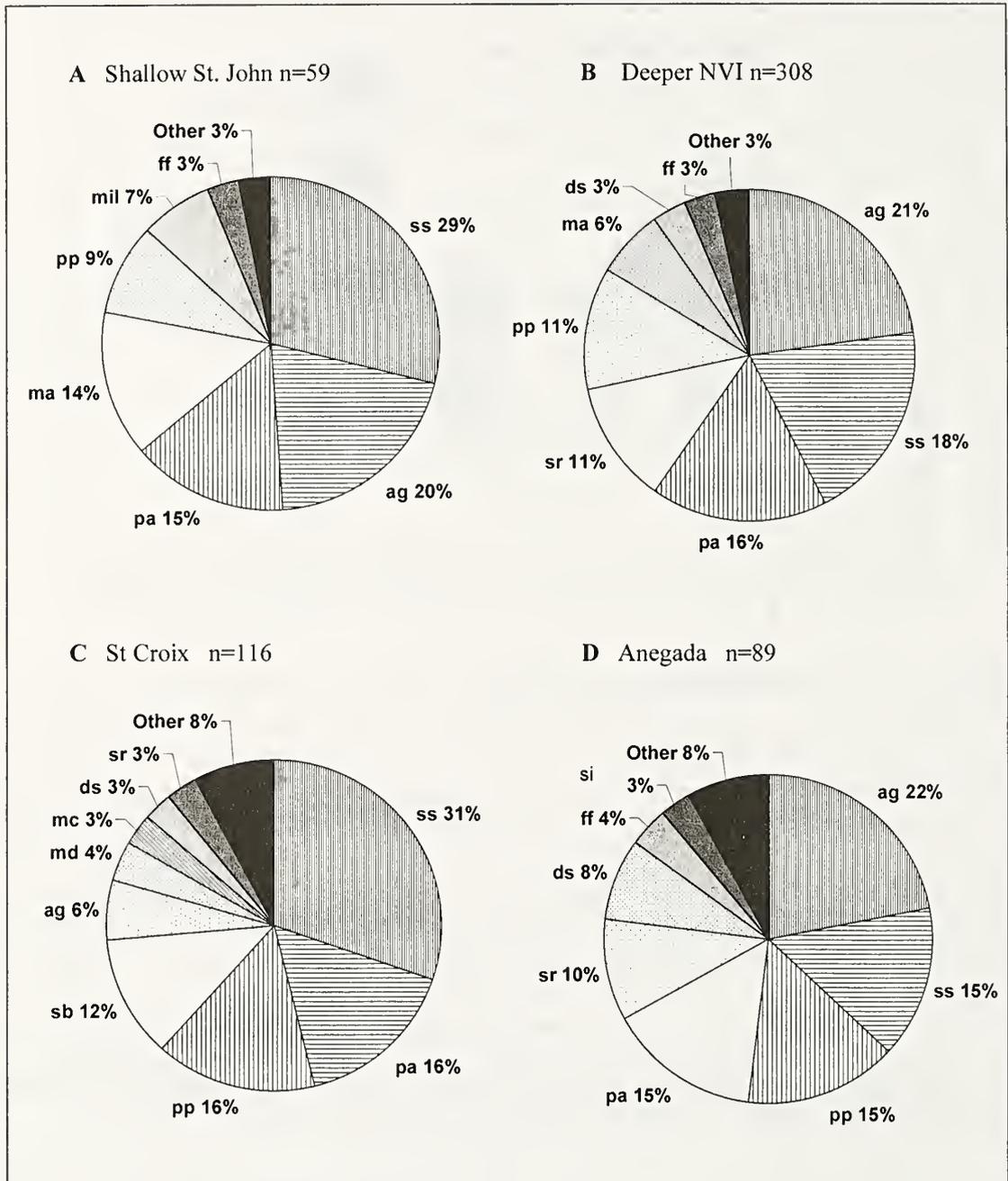
most (17/22) sites (Table 2). Although the highest rate of recent partial mortality (5.5%) for an individual reef was in Anegada (Jack Bay, Table 2), the highest rates of recent partial mortality for an island group occurred in the shallow St. John sites where over 50% of the large stony corals were affected during the surveys in 1999 and 2000 (Fig. 8). Fifty-five to 70% percent of the corals throughout the Virgin Islands had signs of old tissue mortality on more than 10% of their upper surface areas (Fig. 8). Average levels of old tissue mortality per coral colony were highest in the shallow St. John sites (43%) followed by Anegada (40%), St. Croix (34%) and the deeper NVI (27%). The percent of large colonies that were “standing dead” (no living tissues and still in growth position) was less than or equal to 1.5% at 17 sites but was relatively high (8% and 12%) in two reefs off Anegada (Table 2). The average frequency of standing dead colonies by island group was as follows: St. Croix (0.3%,  $n=1$  coral), deeper NVI (0.3%,  $n=4$  corals), shallow St. John (1.7%,  $n=7$  corals), and Anegada (9%,  $n=36$  corals). Standing dead corals in shallow St. John, deeper NVI and St. Croix consisted of a variety of species: *M. cavernosa*, *M. faveolata*, *M. annularis*, *S. siderea*, *P. astreoides*, *C. natans*, and *A. palmata*. In Anegada, however, all (12/12) of the standing dead colonies at Horseshoe Reef were *A. palmata* and most (11/14) of the standing dead colonies at Jack Bay were *M. cavernosa*. Other standing dead colonies at Anegada included *Agaricia*, *D. labyrinthiformis*, *D. strigosa*, *M. annularis*, *M. franksi*, *P. astreoides*, and *P. porites*.



**Figure 8.** Frequency distribution of recent and of old partial colony mortality of all stony corals ( $\geq 25$  cm diameter) in (A) shallow St. John, (B) deeper NVI, (C) St. Croix, and (D) Anegada.

### Stony Coral Recruitment, Macroalgae and *Diadema*

Stony coral recruitment varied considerably from site to site (Table 3). With the exception of *Siderastrea siderea*, coral recruits were dominated by species that brood their larvae. The five most abundant taxa, *Siderastrea siderea* (23%), *Agaricia* (17%), *P. astreoides* (15%), *P. porites* (13%) and *S. radians* (6%) comprised 70% to 80% of the recruits in all island groups (Fig. 9). Other recruits that were each less than three percent in abundance in the four island groups are as follows: shallow St. John (*D. strigosa*, *D. labyrinthiformis*, *M. mirabilis*); deeper NVI (*M. mirabilis*, *S. bournoni*, *M. cavernosa*, *D.*



**Figure 9.** Species composition and mean relative abundance of all stony coral recruits ( $\leq 2$  cm diameter) in (A) St. John shallow, (B) Northern Virgin Islands, (C) St. Croix, (D) Anegada. AG = *Agaricia*, ds = *Diploria strigosa*, ff = *Favia fragum*, md = *Madracis decactis*, mil = *Millepora alcornis*, ma = *Montastraea annularis* species complex, mc = *Montastraea cavernosa*, pa = *Porites astreoides*, pp = *P. porites*, sr = *Siderastrea radians*, ss = *S. siderea*, sm = *Stephanocoenia intersepta*, sb = *Solenastrea bournoni*, Other = all species with  $< 3\%$  abundance.

*labyrinthiformis*, *A. cervicornis*, *D. stokesi*, *Eusmillia fastigiata*, *S. intersepta*, *D. clivosa*, *M. alcicornis*); St. Croix (*M. mirabilis*, *S. intersepta*, *D. stokesi*, *E. fastigiata*, *M. alcicornis*, *F. fragum*, *M. annularis*); and Anegada (*D. labyrinthiformis*, *M. alcicornis*, *M. annularis*, *M. cavernosa*).

Of the 16 sites in which relative abundance estimates were made for all three algal functional groups, four were dominated by macroalgae, eight were dominated by turf algae and these two groups were essentially coequals in the remaining four. Crustose coralline algae were the least abundant at all but one site (Caret Bay, St. Thomas) where they were more abundant than turf algae. When comparing absolute abundances of macroalgae within the quadrats, over half (14/22) of the sites ranged from 25% to 45%, two were over 50% and three sites were under 10% (Table 3). *Dictyota* was the dominant macroalga in the Virgin Islands reefs and macroalgae were typically less than 3 cm in height (Table 3). Values for the macroalgal index (as relative macroalgal abundance x macroalgal height) obtained from the 1998-1999 surveys ranged from 3-35 in St. Croix, 6-247 in the deeper NVI and 46-278 in the shallow St. John sites. Corresponding values based on absolute macroalgal abundances for the entire dataset produced a similar pattern with shallow St. John>Anegada>deeper NVI>St. Croix (Table 3).

The density of *Diadema antillarum* was greatest in the shallow St. John sites (i.e., in depths of less than five meters) (Table 3).

## DISCUSSION

It was anticipated that coral reefs located near human population centers would be exposed to higher levels of sedimentation, pollution, recreational diving, anchor damage and more intense fishing and thus would be in worse condition than reefs more remote from these human impacts (Appendix A, this paper). Contrary to this assumption, our surveys found that the Anegada sites had the greatest amount of dead stony coral tissue and the lowest live coral cover among the deeper reef systems. Old mortality in Anegada was largely attributed to the fact that its reefs were historically dominated by *Acropora palmata* which was decimated during a white-band disease epizootic in the late 1970's (Dunne and Brown, 1976, unpublished report; Gladfelter, 1982). During our survey, many of these *A. palmata* were broken or severely eroded with other species of corals colonizing their surfaces; thus many did not get categorized as standing dead colonies.

The shallow St. John sites were fairly similar to the deeper NVI sites in the species composition of large corals and in coral recruitment (Figs. 3 and 9). However, the shallow St. John sites had a much higher percentage of old partial-colony mortality than the deeper NVI (43% vs. 27%) and had fewer small (<50 cm diameter) colonies of the *Montastraea annularis* species complex (29% vs. 50%) (Fig 4). The partial death or damage to these shallow-water corals was possibly due to a series of strong hurricanes impacting St. John's reefs during the past 20 years or to possible epizootic events (see below). All else being equal, large storm waves have a greater impact in shallow reefs by breaking branching corals, dislodging or toppling boulder corals, or scouring thin coral tissues (Rogers et al., 1983; Edmonds and Witman, 1991; Rogers et al., 1991; Nemeth and Sladek Nowlis, 2001). The shallow St. John sites also contained higher densities of

*Diadema antillarum*, a pattern of recolonization that is typical throughout the Caribbean (Nemeth, personal observation) following its die-off in the early 1980's (Lessios, 1988).

The shallow reefs also contained higher densities of damselfish, in particular *Stegastes planifrons*, which is known to inhabit shallow reef areas and to damage and kill living coral tissue for its algal gardens (Itzkowitz, 1977; Kaufman, 1977; Williams, 1979). This resulted in large numbers of coral colonies showing signs of tissue damage from damselfish bites. Bites by parrotfish were also assessed. Since the stoplight parrotfish, *Sparisoma viride*, is known to cause damage to coral colonies (Bruckner and Bruckner, 1998; Bruckner et al., 2000), we examined the AGRRA fish data collected during our surveys (Nemeth et al., this volume) and found that bite damage seemed to be positively associated to the average size of  $\geq 5$  cm stoplight parrotfish present in reef sites. For example, the St. Croix and Anegada reefs showed the highest incidence of parrotfish bite damage and had the largest stoplight parrotfish observed during the AGRRA fish surveys (29.2 cm and 17.7 cm, respectively). In contrast, the average size of stoplight parrotfish in the deeper NVI and in shallow St. John was 13.1 cm and 9.27 cm, respectively.

Although the 1998 bleaching event did not directly influence the results presented in this report, Nemeth and Sladek Nowlis (2001) found that sedimentation from land development made corals more susceptible to bleaching, especially during the 1998 event. Nemeth and Sladek Nowlis (2001) also documented low-to-moderate levels of seasonal bleaching during July, August and early September, when sea surface temperatures typically reach an annual high, and during the rainy season (October, November, early December) when the influx of terrigenous sediment into the marine environment is greatest. This may explain the moderate levels of bleaching observed for all sites in St. Croix (October, December) and Anegada (July), four of five sites in St. Thomas (July/August) and the deeper St. John sites (July/August), three of four shallow sites in St. John (August/ October/November) and the one site in Virgin Gorda (July). The lowest levels of bleaching typically occurred in sites surveyed from February to June.

Among the island groups, the incidence of disease was lowest in Anegada in July 2000 and in St. Croix during October and December 1999. Although little is known of the mechanisms of transmission of coral diseases, both these islands are remote relative to those in the NVI. Guana Island was unusual in that it was among the highest in live stony coral cover, among the lowest in macroalgal abundance and in the percentage of total partial-colony mortality, but it also had a considerably higher percentage of diseased corals (24% in 1999) than found at any date in any other site (Tables 1-3). Our AGRRA survey may have documented the initial stages of a disease outbreak at this site in which 65% of the affected corals were in the genus *Montastraea*. Unfortunately, most of the signs of disease in the Guana Island corals were unfamiliar to us and classified as "unknown." Alternatively, since Guana Island is privately owned, sparsely populated, and its reefs receive minimal human disturbance, its stony corals may be more capable of resisting or recovering from diseases which may have been present for some time. Outbreaks of white plague type II, which occurred repeatedly in St. John (December 1997, April, May, June; December 1998; August 2000), resulted in considerable mortality to several isolated reefs around St. John including Tektite Reef in Great Lameshur Bay (Miller et al., in press). None of our survey dates in St. John corresponded

with these peak periods in disease outbreaks. However, the percent of diseased colonies in our surveys in Tektite Reef (10% in August/November 1999) was similar to that reported by Miller et al. (in press) who found 5-12% diseased by white plague type II during the same time period. Temporal variability in outbreaks of disease may have contributed to the low percentage of disease detected at sites in Anegada and Virgin Gorda which were all sampled during a three-day period in July 2000.

Based on our comparison of populated islands with remote islands, we conclude that the condition of coral reefs around the Virgin Islands is primarily determined by large-scale natural disturbances such as hurricanes and disease epidemics. Although many reefs in the Caribbean and other parts of the world were severely damaged by the 1998 mass bleaching event, the corals around the Virgin Islands had largely recovered by 1999. The remoteness of Anegada may benefit the ecology of its reefs under certain circumstances but, following disturbances, it is possible that larval recruitment of major, reef-building corals is limited which may slow the recovery of reefs surrounding this island.

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Table 1. Site information for AGRRA stony coral and algal surveys in the Virgin Islands.

Site name	Site code	Reef Type <sup>1</sup> / Exposure	Latitude (° ' " N)	Longitude (° ' " W)	Survey Date(s)	Depth (m)	Benthic transects (#)	>25 cm stony corals (#/10m)	% live stony coral cover (mean ± sd)
<i>Northern Virgin Islands</i>									
<i>St John (shallow)</i>									
Great Lameshur, Donkey	12	F/Windward	18 18.853	64 43.312	May 26 00	3	12	8.5	18.5 ± 3.5
Great Lameshur, VIERS	13	F/Windward	18 19.094	64 43.390	Oct 23 99	5.5	25	4	13.5 ± 6.0
Fish Bay East, Inner	11	F/Windward	18 19.073	64 45.808	Nov 14 99	5	27	3.5	10.5 ± 6.0
Fish Bay West, Inner	14	F/Windward	18 19.053	64 45.878	Aug 15 00, Oct 24 00	5	12	8.5	7.0 ± 6.0
<i>St. John (deep)</i>									
Great Lameshur, Tektite	15	F/Windward	18 18.572	64 43.302	Aug 3 99, Nov 29 99	11	10	10.5	48.5 ± 13.5
Great Lameshur, Yawzi	16	F/Windward	18 18.831	64 43.596	July 23 99	13	26	4.5	16.0 ± 8.0
Fish Bay East, Outer	9	F/Windward	18 18.948	64 45.777	Feb 08 00	6.5	23	4.5	12.0 ± 5.0
Fish Bay West, Outer	10	F/Windward	18 18.850	64 45.845	Aug 02 99	7.5	29	4	13.0 ± 6.5
<i>St. Thomas</i>									
Brewer's Bay	2	F/Windward	18 20.670	64 59.157	May 26 99, June 9 00	8.5	16	6.5	19.0 ± 6.5
Buck Island	4	F/Windward	18 19.781	64 57.097	May 19 98, July 14 99	14	38	3.5	11.5 ± 6.0
Caret Bay	1	F/Windward	18 22.421	64 59.371	May 21 98, July 17 00	9.5	17	6	23.5 ± 8.5
Flat Cay	3	F/Windward	18 19.072	64 59.444	July 7 99, June 12 00	12	18	6	21.0 ± 7.5
Sprat Bay	5	F-B/Windward	18 19.718	64 55.630	Aug 31 00	10	8	13	29.5 ± 7.0
<i>Virgin Gorda</i>									
Eustatia Reef	18	F-B/Windward	18 30.50	64 20.250	July 24 00	8.5	11	9.5	17.0 ± 6.0
<i>Guana Is</i>									
Iguana Head	17	F/Windward	18 28.477	64 34.941	Aug 06 99	10	13	8.5	30.0 ± 9.5
<i>St. Croix</i>									
Cane Bay	6	F/Windward	17 46.230	64 48.530	Oct 14 99, Dec 16 99	9.5	13	7.5	30.5 ± 11.0
Long Reef	8	F-B/Windward	17 46.118	64 41.490	Oct 13 99	13.5	13	7.5	16.0 ± 3.5
Salt River East	7	F/Windward	17 47.227	64 45.330	Oct 13 99	10	17	6	14.0 ± 4.0
<i>Aneгада</i>									
Herman's Reef	19	B-B /Leeward	18 33.841	64 14.320	July 24 00	13	10	10	19.5 ± 5.5
Horseshoe Reef	20	B-B /Windward	18 39.965	64 13.890	July 22 00	10.5	17	6	14.0 ± 6.5
Jack Bay	21	F-B/Windward	18 44.961	64 19.246	July 22 00	9	23	4.5	8.0 ± 4.5
W. Cow Wreck	22	F-B/Windward	18 45.164	64 24.596	July 23 00	8.5	14	7	11.5 ± 3.5

<sup>1</sup> F = fringing; F-B = fringing-barrier; B-B = bank-barrier

Table 2. Size and condition (mean  $\pm$  standard deviation) of all stony corals (>25 cm diameter) by sites in the Virgin Islands.

Site name (Site code)	Stony corals		Partial-colony mortality (%)			Stony corals (%)			
	#	Diameter (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased	With fish bites
<i>Northern Virgin Islands</i>									
<i>St. John (shallow)</i>									
Great Lameshur, Donkey (12)	101	95.0 $\pm$ 46.0	2.5 $\pm$ 3.0	53.5 $\pm$ 31.0	56.0 $\pm$ 31.0	2.0	25.0	0	25.5
Great Lameshur, VIERS (13)	104	53.0 $\pm$ 27.5	4.5 $\pm$ 7.0	28.5 $\pm$ 30.5	33.0 $\pm$ 30.0	0	38.5	6.0	39.5
Fish Bay East, Inner (11)	100	52.0 $\pm$ 28.5	3.5 $\pm$ 6.0	39.5 $\pm$ 32.5	43.0 $\pm$ 31.5	1.0	38.0	10.0	4.0
Fish Bay West, Inner (14)	102	55.0 $\pm$ 36.0	3.0 $\pm$ 5.0	50.5 $\pm$ 37.5	53.0 $\pm$ 36.5	4.0	12.5	2.0	0.0
<i>St. John (deep)</i>									
Great Lameshur, Tektite (15)	103	75.0 $\pm$ 46.5	0.5 $\pm$ 2.5	21.0 $\pm$ 18.5	21.5 $\pm$ 19.0	0	36.0	10.0	3.0
Great Lameshur, Yawzi (16)	112	47.0 $\pm$ 25.0	1.0 $\pm$ 3.5	21.5 $\pm$ 24.0	22.5 $\pm$ 25.0	0	21.5	5.5	2.5
Fish Bay East, Outer (9)	102	58.5 $\pm$ 35.5	0.5 $\pm$ 1.5	44.5 $\pm$ 37.0	45.0 $\pm$ 37.0	1.0	10.0	6.0	4.0
Fish Bay West, Outer (10)	102	54.5 $\pm$ 38.5	1.0 $\pm$ 2.5	21.0 $\pm$ 22.5	22.0 $\pm$ 22.5	0	19.0	11.5	1.5
<i>St. Thomas</i>									
Brewer's Bay (2)	103	50.0 $\pm$ 30.0	1.5 $\pm$ 3.0	37.0 $\pm$ 33.5	38.5 $\pm$ 34.0	0	8.0	9.5	9.5
Buck Island (4)	133	41.0 $\pm$ 22.5	2.0 $\pm$ 9.0	25.5 $\pm$ 30.5	27.5 $\pm$ 30.0	1.5	23.5	4.5	13.5
Caret Bay (1)	105	53.5 $\pm$ 32.5	0.5 $\pm$ 2.0	26.0 $\pm$ 30.5	26.5 $\pm$ 30.5	0	18.0	6.5	8.5
Flat Cay (3)	107	42.0 $\pm$ 20.5	1.0 $\pm$ 2.0	22.0 $\pm$ 30.5	23.0 $\pm$ 28.0	0	18.5	10.5	3.7
Sprat Bay (5)	105	52.5 $\pm$ 30.5	3.5 $\pm$ 5.0	28.5 $\pm$ 24.0	32.5 $\pm$ 23.0	0	15.0	6.0	7.5
<i>Virgin Gorda</i>									
Eustatia Reef (18)	103	56.0 $\pm$ 42.0	2.5 $\pm$ 7.0	30.0 $\pm$ 29.0	32.5 $\pm$ 29.5	0	24.0	5.0	15.5
<i>Guana Is</i>									
Iguana Head (17)	112	55.0 $\pm$ 30.5	1.0 $\pm$ 3.0	17.5 $\pm$ 28.0	22.0 $\pm$ 23.5	0	15.0	24.0	14.5
<i>St. Croix</i>									
Cane Bay (6)	98	63.0 $\pm$ 41.0	1.5 $\pm$ 3.5	23.5 $\pm$ 25.0	24.5 $\pm$ 25.0	0	52.0	5.0	19.5
Long Reef (8)	100	48.5 $\pm$ 26.0	0.5 $\pm$ 1.5	42.0 $\pm$ 31.0	42.5 $\pm$ 31.0	0	46.0	1.0	1.0
Salt River East (7)	103	37.5 $\pm$ 14.5	<0.5 $\pm$ 1.0	35.0 $\pm$ 33.5	35.0 $\pm$ 33.5	1.0	46.5	0	0
<i>Anegada</i>									
Herman's Reef (19)	100	54.5 $\pm$ 32.0	2.0 $\pm$ 4.0	40.5 $\pm$ 30.0	42.5 $\pm$ 29.5	3.0	25.0	1.0	25.0
Horseshoe Reef (20)	102	64.0 $\pm$ 57.0	1.0 $\pm$ 2.0	38.0 $\pm$ 37.0	39.0 $\pm$ 37.0	12.0	23.5	2.0	9.0
Jack Bay (21)	100	55.5 $\pm$ 34.0	5.5 $\pm$ 20.0	38.0 $\pm$ 36.0	43.5 $\pm$ 37.0	8.0	35.0	2.0	5.0
W. Cow Wreck (22)	100	46.5 $\pm$ 25.0	1.0 $\pm$ 2.0	44.5 $\pm$ 33.5	45.5 $\pm$ 33.5	0	28.0	3.0	9.0

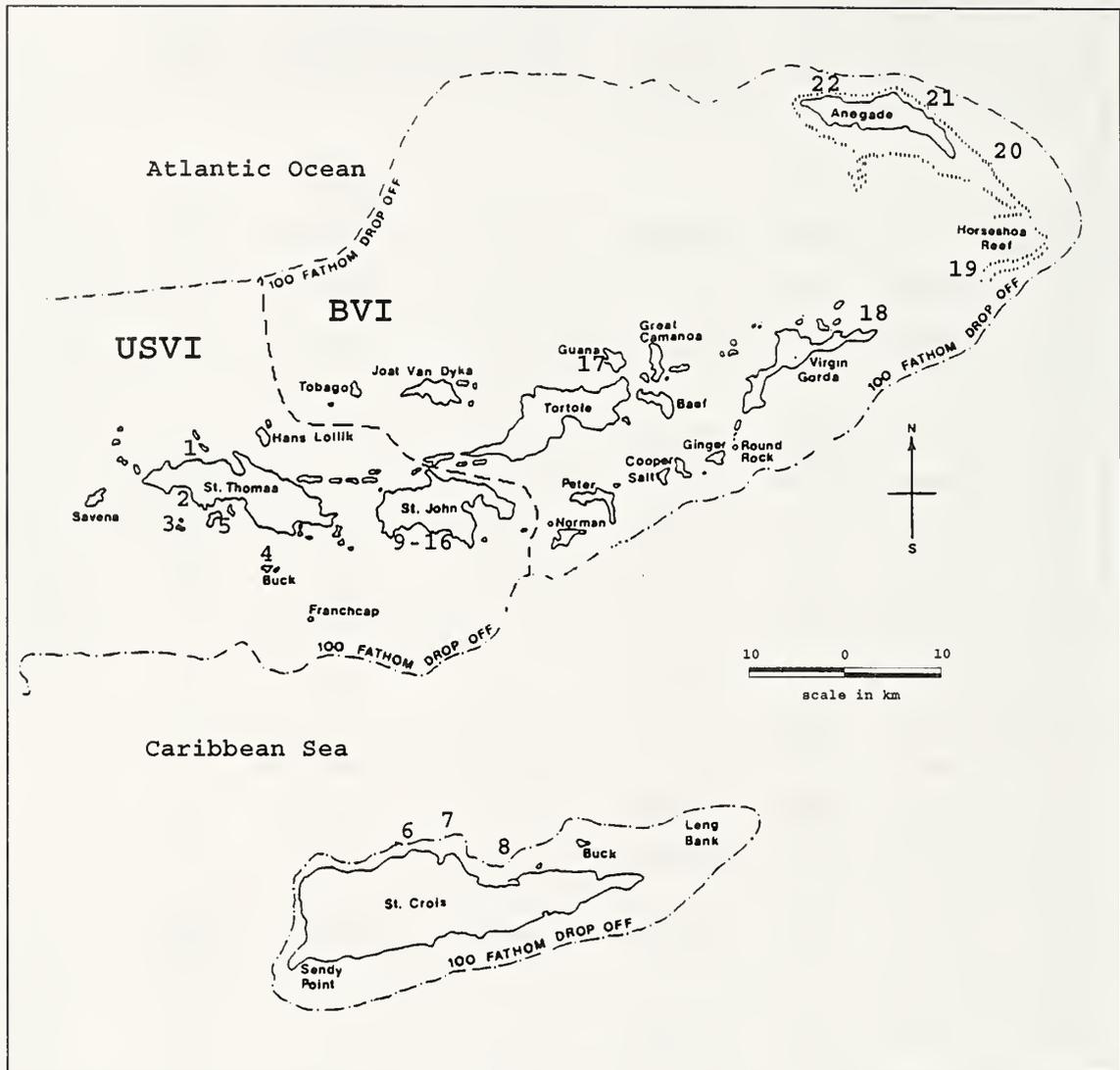
Table 3. Algal characteristics, and density of stony coral recruits and of *Diadema antillarum* (mean  $\pm$  standard deviation) by sites in the Virgin Islands.

Site name (Site code)	# quadrats	Macroalgae		% relative abundance turf algae	% relative abundance crustose coralline algae	Macroalgal		Recruits #/0.0625m <sup>2</sup>	<i>Diadema</i> #/100m <sup>2</sup>
		% relative abundance	% absolute abundance			Height (cm)	Indices Relative (Absolute) <sup>2</sup>		
<b>Northern Virgin Islands</b>									
<b>St. John (shallow)</b>									
Great Lameshur, Donkey (12)	52	28.5 $\pm$ 23.0	31.8	68.5 $\pm$ 23.0	3.0 $\pm$ 4.0	2.5 $\pm$ 1.5	71 (80)	5.0 $\pm$ 0.5	26
Great Lameshur, VIERS (13)	50	23.0 $\pm$ 18.0	36.1	59.5 $\pm$ 21.0	17.5 $\pm$ 14.0	2.0 $\pm$ 1.0	46 (72)	7.5 $\pm$ 1.0	5
Fish Bay East, Inner (11)	58	36.0 $\pm$ 26.5	38.8	49.0 $\pm$ 27.4	15.0 $\pm$ 17.0	3.0 $\pm$ 1.0	108 (116)	3.5 $\pm$ 0.5	7
Fish Bay West, Inner (14)	59	79.5 $\pm$ 28.5	51.6	11.0 $\pm$ 27.5	9.0 $\pm$ 15.5	3.5 $\pm$ 1.0	278 (181)	1.5	8
<b>St. John (deep)</b>									
Great Lameshur, Tektite (15)	54	76.0 $\pm$ 30.0	44.6	17.5 $\pm$ 23.5	6.5 $\pm$ 15.5	2.5 $\pm$ 1.5	190 (112)	8.0 $\pm$ 1.0	0
Great Lameshur, Yawzi (16)	50	37.0 $\pm$ 29.5	20.3	49.5 $\pm$ 31.5	14.0 $\pm$ 13.0	1.5 $\pm$ 1.0	55 (31)	10.0 $\pm$ 1.0	1
Fish Bay East, Outer (9)	50	6.0 $\pm$ 7.0	6.8	75.5 $\pm$ 18.0	18.5 $\pm$ 17.0	1.0 $\pm$ 1.0	6 (7)	17.5 $\pm$ 1.5	0
Fish Bay West, Outer (10)	60	51.5 $\pm$ 34.0	30.4	42.5 $\pm$ 35.0	6.0 $\pm$ 9.5	1.0 $\pm$ 0.5	52 (30)	3.5 $\pm$ 0.5	0
<b>St. Thomas</b>									
Brewer's Bay (2)	50	33.0 $\pm$ 17.0	47.1	62.5 $\pm$ 18.5	4.5 $\pm$ 7.5	2.5 $\pm$ 2.0	83 (118)	12.5 $\pm$ 1.0	4
Buck Island (4)	72	69.0 $\pm$ 25.5	38.2	19.0 $\pm$ 22.0	12.0 $\pm$ 15.0	2.5 $\pm$ 1.0	173 (96)	8.5 $\pm$ 1.0	0
Caret Bay (1)	52	55.0 $\pm$ 35.0	40.9	7.0 $\pm$ 20.0	38.0 $\pm$ 30.0	2.0 $\pm$ 1.0	110 (82)	7.5 $\pm$ 1.0	0
Flat Cay (3)	50	46.5 $\pm$ 28.0	26.3	37.0 $\pm$ 30.0	16.5 $\pm$ 19.0	2.0 $\pm$ 1.5	93 (53)	10.5 $\pm$ 1.0	1
Sprat Bay (5)	8	70.5 $\pm$ 28.5	57.6	* <sup>1</sup>	29.5 $\pm$ 28.5	3.5 $\pm$ 2.0	247 (201)	2.0	4
<b>Virgin Gorda</b>									
Eustatia Reef (18)	52	58.0 $\pm$ 31.5	43.2	*	42.0 $\pm$ 31.5	2.0 $\pm$ 1.5	116 (86)	3.0 $\pm$ 0.5	6
<b>Guana Is</b>									
Iguana Head (17)	50	8.0 $\pm$ 14.0	7.7	78.5 $\pm$ 22.5	13.5 $\pm$ 20.0	1.0 $\pm$ 1.0	8 (8)	10.0 $\pm$ 1.0	0
<b>St. Croix</b>									
Cane Bay (6)	50	17.5 $\pm$ 17.0	35.2	67.0 $\pm$ 20.0	15.0 $\pm$ 19.5	2.0 $\pm$ 1.0	35 (70)	6.0 $\pm$ 1.0	4
Long Reef (8)	70	25.0 $\pm$ 21.5	18.9	66.0 $\pm$ 24.6	8.5 $\pm$ 14.5	1.0 $\pm$ 0.5	25 (19)	16.0 $\pm$ 2.0	0
Salt River East (7)	50	5.5 $\pm$ 10.5	3.2	89.0 $\pm$ 15.5	6.0 $\pm$ 9.5	0.5 $\pm$ <0.5	3 (2)	9.0 $\pm$ 1.0	0
<b>Anegada</b>									
Herman's Reef (19)	51	52.0 $\pm$ 33.0	27.3	*	48.0 $\pm$ 33.0	2.0 $\pm$ 1.5	104 (55)	4.5 $\pm$ 0.5	0
Horseshoe Reef (20)	50	73.5 $\pm$ 27.5	35.1	*	26.5 $\pm$ 27.5	2.0 $\pm$ 1.5	147 (70)	10.0 $\pm$ 2.0	0
Jack Bay (21)	70	40.5 $\pm$ 32.8	30.8	*	59.5 $\pm$ 33.0	2.5 $\pm$ 2.0	101 (77)	4.5 $\pm$ 0.5	0
W. Cow Wreck (22)	53	65.7 $\pm$ 28.0	42.8	*	34.3 $\pm$ 28.0	3.5 $\pm$ 4.5	230 (150)	7.5 $\pm$ 1.0	0

<sup>1</sup>\* = not recorded<sup>2</sup>Macroalgal index = % relative (or absolute) macroalgal abundance x macroalgal height.

Appendix A. Qualitative assessment of human and natural impacts at 22 sites in the Virgin Islands.

Site name (Site code)	Sediment	Divers	Fishing	Exposure to prevailing seas	Historical hurricane damage	Historical disease epizootics
<i>Northern Virgin Islands</i>						
<i>St. John (shallow)</i>						
Great Lameshur, Donkey (12)	low	low	low	low	moderate	moderate
Great Lameshur, VIERS (13)	low	moderate	Moderate?	low	moderate	moderate
Fish Bay East, Inner (11)	moderate	low	low	low	low	moderate
Fish Bay West, Inner (14)	high	low	low	low	low	moderate
<i>St. John (deep)</i>						
Great Lameshur, Tektite (15)	low	moderate	moderate?	low	moderate	high (white plague)
Great Lameshur, Yawzi (16)	moderate	moderate	moderate?	low	high	moderate (white plague)
Fish Bay East, Outer (9)	moderate	low	low	low	high	moderate?
Fish Bay West, Outer (10)	high	low	low	low	high	moderate?
<i>St. Thomas</i>						
Brewer's Bay (2)	moderate	low	moderate	low	low	low
Buck Island (4)	low	high	low	low	low	low
Caret Bay (1)	low	low	low	moderate	low	low
Flat Cay (3)	low	high	moderate	low	low	low
Sprat Bay (5)	low	moderate	moderate	low	low	low
<i>Virgin Gorda</i>						
Eustatia Reef (18)	low?	high	moderate?	moderate	moderate?	moderate? (white band)
<i>Guana Is</i>						
Iguana Head (17)	low?	low?	moderate?	low	low?	low?
<i>St. Croix</i>						
Cane Bay (6)	moderate	high	low	moderate	low	low?
Long Reef (8)	moderate	high	high	moderate	moderate	high (white band)
Salt River East (7)	moderate	high	moderate	moderate	low	low?
<i>Anegada</i>						
Herman's Reef (19)	low	low	low?	moderate	high	high (white band)
Horseshoe Reef (20)	low	low	low?	high	high	high (white band)
Jack Bay (21)	low	moderate?	moderate?	high	high	high (white band)
W. Cow Wreck (22)	low	low	low?	high	high	high (white band)



**Figure 1.** AGRR survey sites in the Virgin Islands. (1) Caret Bay, (2) Brewer's Bay, (3) Flat Cay, (4) Buck Island, (5) Sprat Bay, (6) Cane Bay, (7) Salt River, (8) Long Reef, (9) Fish Bay east outer, (10) Fish Bay west outer, (11) Fish Bay east inner, (12) Great Lameshur Donkey, (13) Great Lameshur VIERS, (14) Fish Bay west inner, (15) Great Lameshur Tektite, (16) Great Lameshur Yawzi, (17) Iguana head, (18) Eustatia Reef, (19) Herman's Reef, (20) Horseshoe Reef, (21) Jack Bay, (22) West Cow Wreck.

# A RAPID ASSESSMENT OF CORAL REEFS IN THE VIRGIN ISLANDS (PART 2: FISHES)

BY

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## ABSTRACT

The Atlantic and Gulf Rapid Reef Assessment (AGRRA) fish survey was conducted in depths of 3-14 m at 22 coral reef sites in the U.S. Virgin Islands (St. Croix, St. John, St. Thomas) and the British Virgin Islands (Anegada, Guana Island, Virgin Gorda). Total fish species richness, determined by using the roving diver technique was 72-160 species. A total of 14,441 fish from 54 species of selected families were counted in belt transects. The select fish fauna was dominated by the Scaridae (35%), Acanthuridae (35%) and Haemulidae (16%). Key herbivorous fish densities among sites ranged from ~4-49 fish/100m<sup>2</sup> for scarids and from 1-60 fish/100m<sup>2</sup> for acanthurids. Predatory fish densities among sites ranged from 0.2-5.5 fish/100m<sup>2</sup> for serranids and from 0-11.5 fish/100m<sup>2</sup> for lutjanids. The density of large herbivorous fishes showed a significant negative relationship with macroalgal index and with the percent of coral colonies occupied by territorial damselfish. Also significant were the positive relationships between total fish species richness and coral cover, between chaetodontid density and total fish species richness, and between pomacanthid density and depth.

## INTRODUCTION

The Virgin Islands (18°20' N, 64°50' W) lie between two major island archipelagoes: the Greater Antilles to the west and the Lesser Antilles to the southeast. With the exception of St. Croix, the northern United States Virgin Islands (USVI), together with British Virgin Islands (BVI) and the islands of Puerto Rico, rise from a geological shelf that is surrounded by deep water (Dammann, 1969, unpublished report). This shelf covers approximately 3,200 km<sup>2</sup>, contains about 500 km of shelf edge and is generally less than 100 m in depth. St. Croix sits on a similar but smaller shelf that is separated from the northerly Virgin Islands by the 4,685 m-deep and 60 km-wide Virgin Islands trough. The three main U.S. islands of St. Thomas, St. John and St. Croix are

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surrounded by more than 90 uninhabited islands, cays, and rocks, which provide hard substrata suitable for the growth of coral reefs (Towle, 1970, unpublished report; Dammann and Nellis, 1992). Many of these reefs are adjacent to mangrove forests, seagrass beds and algal plains, which are important nursery habitats for reef-associated fishes (Austin, 1971; Parrish, 1989; Heck and Weinstein, 1989).

The USVI, like other populated regions of the world, have witnessed steady declines in catches of commercially important marine fishes (Caribbean Fisheries Management Council, 1985, unpublished report; Roberts, 1997). Millions of people in developing countries depend upon coral reef fishes for their livelihoods (Roberts, 1993). However, declining fisheries cannot only be attributed to overfishing but to a suite of other factors since evidence suggests that degradation of coral reefs can influence reef fish communities (Bell and Galzin, 1984; Bouchon-Navaro et al., 1985). Over the past 20 years, eight major hurricanes, numerous disease outbreaks, sporadic bleaching events, ship groundings, anchor damage, plus inputs of sediment and pollution have caused excessive damage to the coral reefs surrounding the Virgin Islands (Gladfelter, 1982; Rogers et al., 1991; Rogers and Garrison, 2001).

The USVI fishing community has remained relatively stable since the early 1900's. Licensed fishermen have consistently numbered approximately 400 between 1930 and today, even though the total population of the islands has increased from 22,012 to about 100,000 (Fielder and Jarvis, 1932; Department of Commerce Census, 2000). Since the early 1970's, fish stocks in the near shore USVI have been declining steadily despite government-funded research and development programs for the deep-water fisheries around the Virgin Islands plateau (Brownell and Rainey, 1971, unpublished report). The catch of fishery products fluctuated between 550,000 and 1,650,000 pounds per year between 1930 and 1997 (Fielder and Jarvis, 1932; de Graff and Moore, 1987, unpublished report; Tobias, 1997, unpublished report), but the fishing effort increased dramatically. For example, in the USVI the average number of traps fished per full-time fisherman increased from 4 in 1930, to 8 in 1967, to >100 in 1997 (Fielder and Jarvis, 1932; Dammann, 1969; unpublished report; T. Tobias, USVI Fish and Wildlife, 1998). The maximum number of traps fished by a single fisherman in 1930 was 30 traps whereas today it can be as high as 2000 traps (Fielder and Jarvis, 1932; Downs and Petterson, 1997, unpublished report). The pursuit of declining stocks not only puts fish populations at risk of commercial extinction but often increases reef damage associated with fishing (Sladek Nowlis and Roberts, 1998; Appeldoorn, 2001, unpublished manuscript; Nemeth, unpublished data).

Increased fishing pressure on nearshore and offshore fish populations has already resulted in the collapse of several species. For example, in the 1970's unregulated fishing in spawning aggregations of Nassau grouper (*Epinephelus striatus*) and yellowfin grouper (*Mycteroperca venenosa*) off St. Thomas led to the commercial extinction of these species (Olsen and LaPlace, 1978; Beets and Friedlander, 1992, 1999). In the absence of Nassau grouper, fishermen targeted red hind (*E. guttatus*) spawning aggregations. Within 10 years, the St. Thomas red hind population was on the verge of collapse when a seasonal (December through February) closed area at the red hind spawning aggregation site was implemented in 1990 (Beets and Friedlander, 1992). Within 6 years of the closure, Beets and Friedlander (1999) reported that an increase in

their average size and an improved sex ratio (red hind are protogynous hermaphrodites) were indications of red hind population recovery.

Without proper management tools, sustained fishing mortality can decrease the capacity, productivity and genetic diversity of the fishery (Bohnsack, 1990). Moreover, selective removal of ecologically important species can lead to ecosystem damage and produce major ecological shifts (Hay, 1984; Roberts, 1995; Botsford et al., 1997). In response to these threats to coral reefs, Ginsburg et al. (1996) initiated a process of rapid reef assessment. After development of the AGRRA protocols, a Caribbean-wide effort to assess the condition of coral reefs and associated fishes throughout the region was launched. The Center for Marine and Environmental Studies of the University of the Virgin Islands (UVI) joined the effort and set out to conduct a broad survey of the fish assemblages of the Virgin Islands with a focus on commercially and ecologically important species using the AGRRA protocol. This paper reports on the initial findings of our assessment. Results of the benthic surveys are given in Nemeth et al. (this volume).

## METHODS

An assessment of fish species abundance and diversity was conducted at 22 sites in the Virgin Islands (Fig. 1) between May 1998 and July 2000. Site selection criteria in the USVI and the BVI varied among the different islands, but most (18/22) choices were made for strategic reasons. In St. Thomas, five sites were selected based on their inclusion in long-term sedimentation and reef fish monitoring projects, their proximity to the UVI MacLean Marine Science Center and the presence of historical data (i.e., Rogers, 1982, unpublished report; Nemeth and Sladek Nowlis, 2001). Three popular recreational diving sites were selected in St. Croix, one of which is within the Salt River Bay National Historic Park and Ecological Preserve. The eight sites off St. John were part of a study comparing sedimentation rates between Great Lameshur Bay, which is within the National Park Boundary, and Fish Bay, which has been experiencing heavy development within its watershed. Four of these sites were shallow reefs (< 6 m) located inside the bays and four sites were deeper reefs (> 6 m) located outside the bays. Four of the six sites surveyed in the BVI (Iguana Head on Guana Island, Eustatia Reef on Virgin Gorda, West Cow Wreck Bay and Herman's Reef on Anegada) were selected haphazardly. Eustatia Reef, off Virgin Gorda, is a heavily visited dive site whereas Guana Island is a private island with few inhabitants. The other two sites on Anegada (Jack Bay and Horseshoe Reef—a designated protected area) were selected because of historical surveys (Dunne and Brown, 1976; West Indies Laboratory, 1983, unpublished reports). The reefs of Anegada were included in the AGRRA survey to provide a remote reference site with low human population and little landmass. The major difference in fishing regulations between the USVI and the BVI is that spearfishing is prohibited in the BVI. A qualitative assessment of natural and human impacts, including fishing pressure, at each site surveyed in the Virgin Islands is given in Appendix A of Nemeth et al. (this paper).

The AGRRA fish protocol Version 2.0 (see Appendix One, this volume) was used except off St. Croix where Version 2.1 was employed. Three divers (authors) constituted the primary fish survey team with R. Nemeth conducting the majority of surveys on all islands except those off St. Croix. Fish identification guides included Humann (1994),

Stokes (1984) and Robins and Ray (1986). Visual fish counts were conducted along at least ten 30 x 2 m transects, and one 30-minute roving diver survey was conducted at each site except in Brewer's Bay, St. Thomas (n = 2 surveys) and off St. Croix, where longer (60-90 minute) survey were conducted in Cane Bay (n = 3), Long Reef (n = 5) and Salt River (n = 6). During fish transects, transect width and fish lengths (cm) were estimated using a 1 m-wide T-bar constructed of polyvinylchloride pipe marked in 5 cm and 10 cm increments. In all sites except St. Croix, scarids (parrotfish) and haemulids (grunts) less than 5 cm in length were counted and identified to species whenever possible. Using transects as replicates, the average density (#/100m<sup>2</sup>) and size (cm) of each species and family were calculated for each site and island group (see below). The size distributions of herbivores [parrotfish, surgeonfish (acanthurids) and the yellowtail damselfish, *Microspathodon chrysurus*] and of the AGRRA carnivores [select groupers (serranids) and all lutjanids] were also calculated.

Data were summarized by island groups within three geographic areas to examine general trends in reef fish assemblages and to be comparable with the Virgin Islands benthic data which are presented in Nemeth et al. (this volume). The geographic areas were: (1) Anegada; (2) the shallow and deep reefs of the northern USVI and BVI (NVI); and (3) St. Croix. Anegada was considered a geographic unit because of its isolation from the other Virgin Islands and its unique geology (low coral island). St. Croix was considered a geographic unit also because of its isolation from the NVI, its unique geology (sedimentary/carbonate) and because it is completely within the Caribbean Sea. The NVI (USVI = St. John, St. Thomas; BVI = Guana Island, Virgin Gorda) were grouped because of their close proximity to one another, their similar geologic origins and topographies (high volcanic islands) and their exposure to both Atlantic waters from the north and Caribbean waters from the south. The shallow sites off St. John were analyzed separately from the deep reefs within the NVI archipelago.

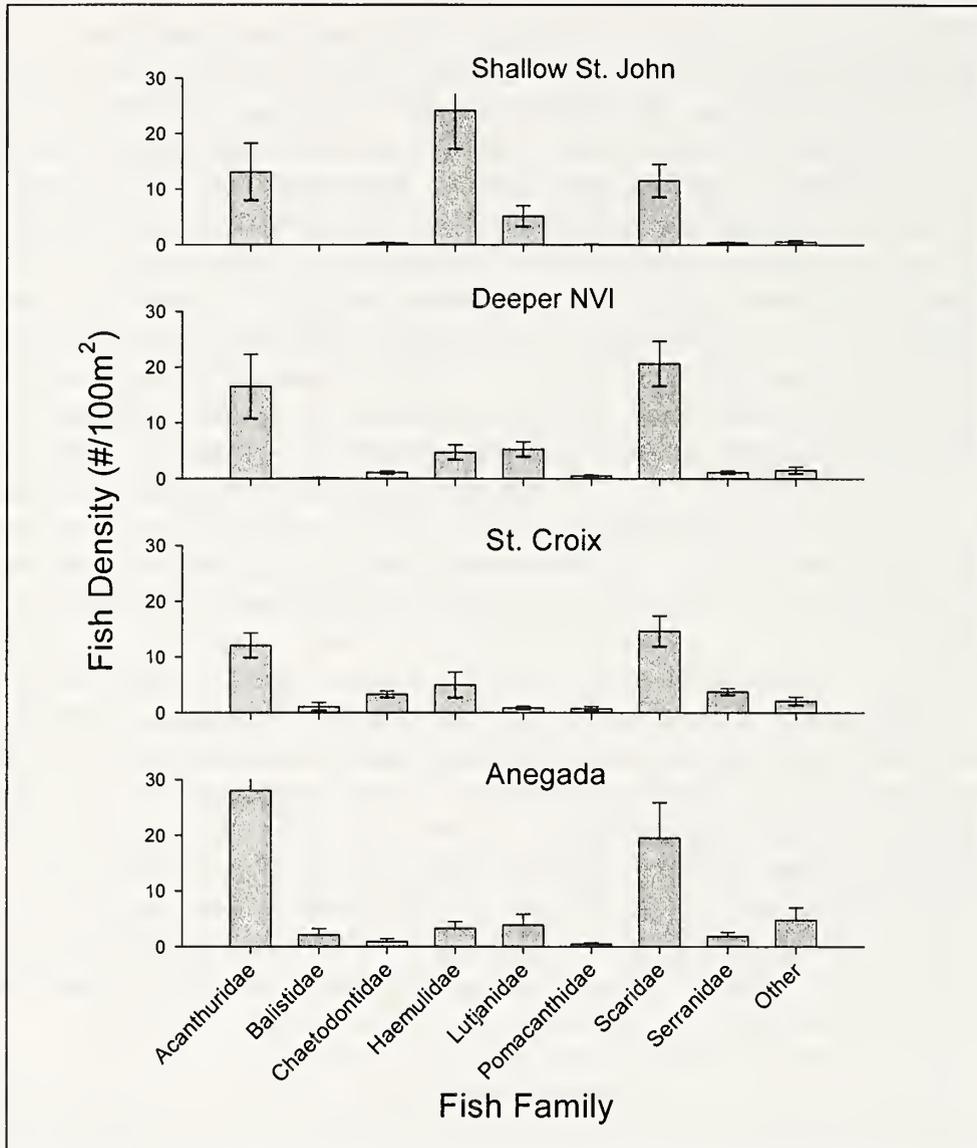
## RESULTS

A total of 256 fish transects and 34 roving diver surveys were conducted at 22 sites throughout the USVI and the BVI (Table 1). During visual belt transects of the selected AGRRA fishes (Appendix One), divers recorded 54 species and counted a total of 8,227 fish (n = 14,441 fish when grunts and parrotfishes <5 cm in Anegada and the NVI are included). Fish species richness, based upon the maximum number of species seen during any single roving diver survey, varied between a high of 74 species in Cane Bay (St. Croix) and a low of 40 species in Fish Bay (shallow St. John) with >90% (20/22) having values between 40 and 70 species (Table 1). The average number of fish species by island group was greatest in St. Croix (68 species), followed by the deep NVI reefs (59), Anegada (52) and the shallow St. John sites (47), although these differences were not significant ( $F_{3,29} = 2.75$ ,  $p = 0.061$ ). Total cumulative reef fish richness by island group was 160 species in the NVI (n = 12 surveys at 11 sites), 131 species in St. Croix (n = 14 surveys at 3 sites), 98 species in Anegada (n = 4 surveys at 4 sites) and 72 species in the shallow St. John sites (n = 4 surveys at 4 sites). The additional roving diver surveys conducted in St. Croix and the large number of deeper sites in the NVI most likely contributed to a higher number of fish species seen within these island groups. To

illustrate, divers in St. Croix counted: 58, 68 and 74 fish species during three roving surveys in Cane Bay; 45, 53, 63, 65, and 68 species during five roving surveys in Long Reef; and 37, 49, 54, 58, 60 and 63 species during six roving surveys in Salt River. The cumulative numbers of fish species seen in Cane Bay, Long Reef and Salt River reefs during these multiple surveys were 92, 88 and 117, respectively (Table 1). Based on an analysis of the six roving diver surveys at Salt River where search times averaged 73 minutes per survey, a species accumulation curve showed that at least four such surveys (i.e. 4.8 hours search time) are necessary to record at least 90% of the fish species and at least six surveys (i.e., 7.3 hours search time) to approximate total species diversity at a site. Thus, depending upon the skill level of the diver, species diversity estimates based on a single 60-minute roving diver survey may represent only 40% to 70% of the actual number of species present in a reef. The 25 most commonly seen species during roving diver surveys were represented by 11 fish families (Pomacentridae, Scaridae, Labridae, Serranidae, Acanthuridae, Holocentridae, Haemulidae, Chaetodontidae, Gobiidae, Mullidae, and Tetraodontidae) with *Thalassoma bifasciatum*, *Acanthurus coeruleus*, *Sparisoma aurofrenatum*, *S. viride* and *Haemulon flavolineatum* being ubiquitous in all sites (Table 2).

Of the 54 AGRRA species sighted in belt transects, scarids and haemulids were each represented by 9 species, lutjanids by 7 species, serranids and pomacanthids each by 6 species, balistids by 5 species, chaetodontids by 4 species, acanthurids by 3 species, labrids by 2 species, and pomacentrids, sphyraenids and carangids each by 1 species. The 25 most abundant of these species (Table 3) represented nearly 94% of all fish sighted in the belt transects. Numerically the AGRRA fish fauna in the Virgin Islands was dominated by the families Scaridae (35%, all  $\geq 5$  cm), Acanthuridae (35%) and Haemulidae (16%, all  $\geq 5$  cm) (Fig. 2, Table 3). The herbivorous scarids and acanthurids represented 7 of the 10 most abundant species, with the ocean surgeonfish (*Acanthurus bahianus*) ranked first overall. Three carnivores, the French grunt (*Haemulon flavolineatum*), tomtate (*H. aurolineatum*) and yellowtail snapper (*Ocyurus chrysurus*) were also ranked among the top 10 most abundant species (Table 3). Although newly recruited grunts were not included in this analysis, high densities of juvenile haemulids (<5 cm) were observed in the NVI off Guana Island (710.7 grunts/100 m<sup>2</sup>) and Buck Island, St. Thomas (10.7 grunts/100 m<sup>2</sup>), as well as in VIERS (8.3 grunts/100m<sup>2</sup>) and Fish Bay East Inner (3 grunts/100 m<sup>2</sup>) in shallow St. John. High densities of juvenile scarids were also recorded. Scarids <5 cm comprised 24-77% of parrotfish densities in Anegada and the NVI (Fig. 3). We found a general increase in the density of juvenile scarids off Anegada (the easternmost reefs) through the lower BVI to the shallow St. John sites where they were most abundant (Fig. 3). Juvenile densities dropped in the deep St. John sites and were slightly higher around St. Thomas. A weak negative relationship between the density of juvenile scarids <5cm and depth ( $r^2 = 0.18$ ,  $F_{20,1} = 4.48$ ,  $p < 0.05$ ) was found in a regression analysis.

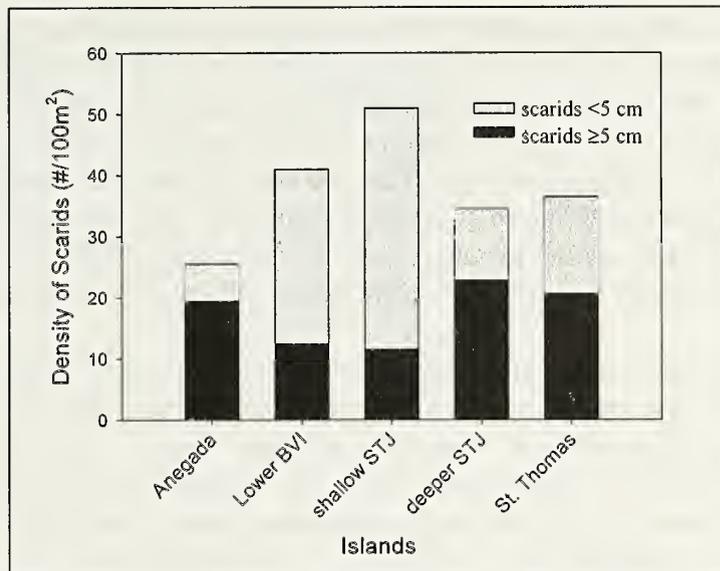
Herbivorous fish densities (# fish/100 m<sup>2</sup>) among sites ranged from 3.8 (Fish Bay West Inner, St. John) to 48.8 (Great Lameshur, Yawzi, St. John) for the Scaridae and 1.0 (Brewer's Bay, St. Thomas) to 60.5 (Fish Bay East Outer, St. John) for the Acanthuridae (Table 4). Overall, the abundance of parrotfishes (>5 cm) and surgeonfishes were similar in both shallow (11.6 scarids versus 13.1 acanthurids/100 m<sup>2</sup>) and deeper (19.4 scarids versus 18.2 acanthurids/100 m<sup>2</sup>) reefs. Predatory fish densities (# fish/100 m<sup>2</sup>) among



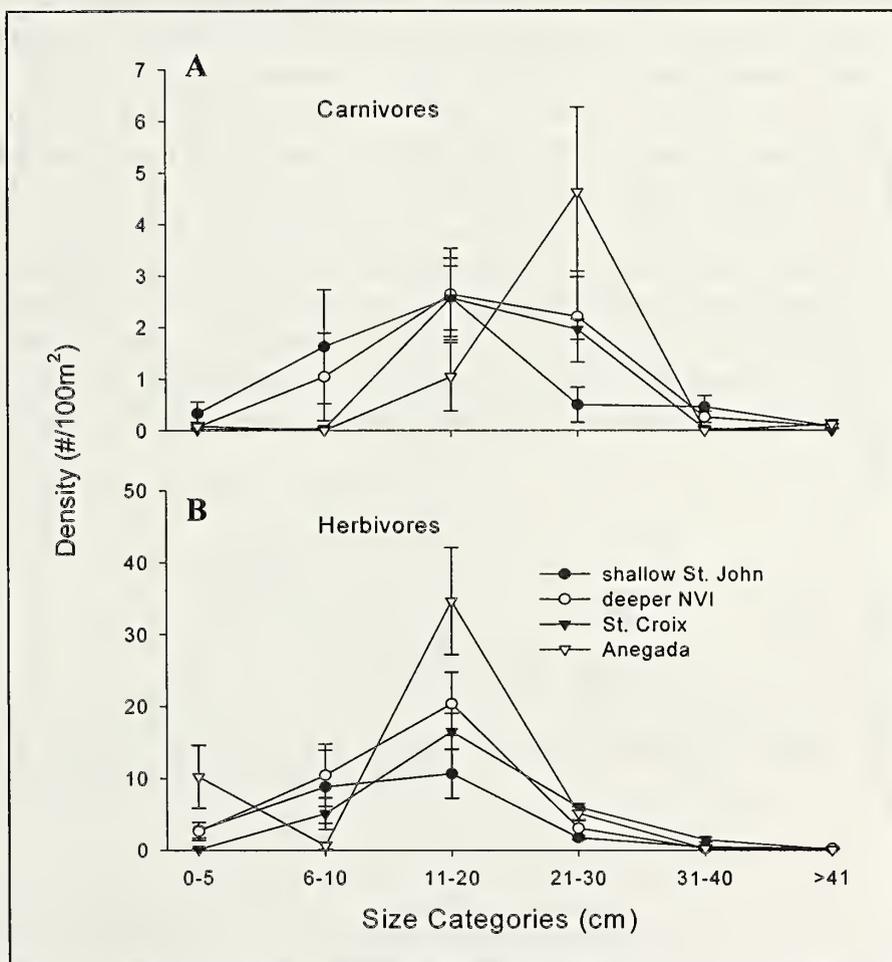
**Figure 2.** Fish density (mean no. fish/100 m<sup>2</sup> ± standard error) for AGRRA fishes in shallow St. John, deeper NVI (St. Thomas, St. John >6 m deep, Guana Is., Virgin Gorda), St. Croix and Anegada. Other = *Bodianus rufus*, *Caranx ruber*, *Lachnolaimus maximus*, *Microspathodon chrysurus*, *Sphyraena barracuda*.

sites ranged from 0.2 (Fish Bay East Inner, St. John) to 5.5 (Cane Bay, St. Croix) for the Serranidae, and from 0.0 (Herman's Reef, Anegada) to 11.5 (Sprat Bay, St. Thomas) for the Lutjanidae (Table 4). It is unusual to note that no snappers were seen at Herman's Reef during transects or roving diver surveys and no grunts were counted during transects at Horseshoe Reef (one French grunt was seen during a roving diver survey).

The size distributions of fishes comprising the two major feeding guilds (herbivores and carnivores) were found to be fairly similar among the deeper NVI reefs and those of St. Croix (Fig. 4). Anegada was unique in that it had twice the density of 21-30 cm carnivores and 11-20 cm herbivores as the other sites and an abundance of juvenile acanthurids <5 cm (Fig. 4). Jack Bay was unusual in that it had considerably smaller groupers than other sites in Anegada (Table 5). The shallow St. John sites had



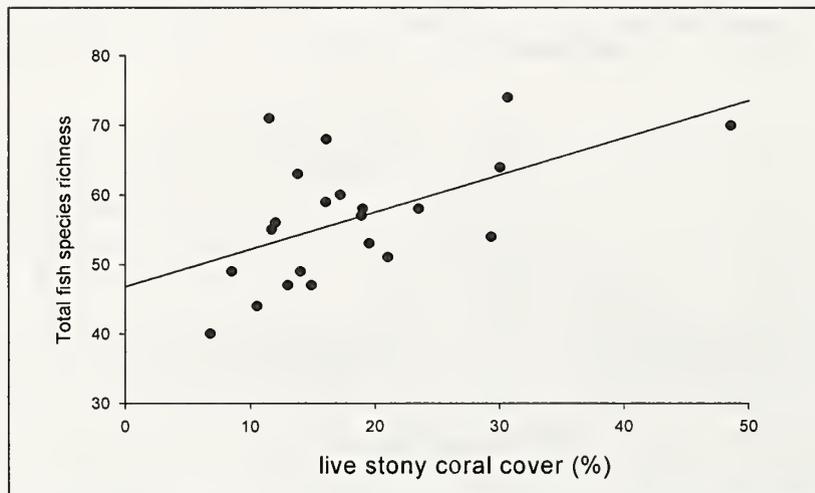
**Figure 3.** Density (mean no. fish/100 m<sup>2</sup>) of scarids  $<5$  cm and  $\geq 5$  cm in length for Aneгада (four reefs), Lower BVI (two reefs), shallow St. John (four reefs), deeper St. John (four reefs) and St. Thomas (five reefs). Scarids  $<5$  cm were not recorded in St. Croix.



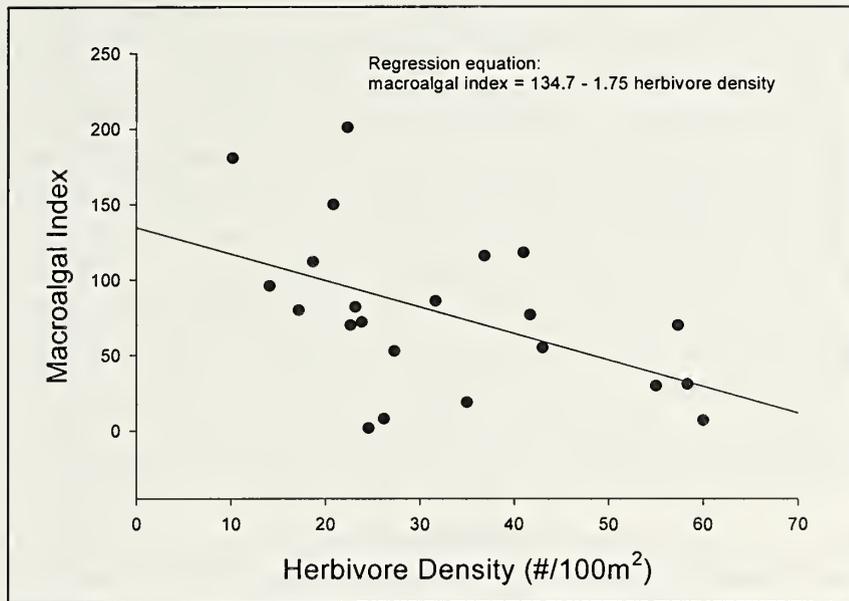
**Figure 4.** Size composition (mean density  $\pm$  standard error) of (A) carnivores (lutjanids, large serranids) and (B) herbivores (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) in shallow St. John, deeper NVI, St. Croix and Aneгада.

substantially fewer groupers and snappers in the 21-30 cm size class (Fig. 4) but over twice as many newly settled (<5 cm) scarids as the other sites (Fig. 3). Within the deeper NVI sites, two reefs in the BVI (Virgin Gorda and Guana Island) and most of the deeper sites around St. John had larger groupers than the reefs around St. Thomas (Table 5). With the exception of Buck Island and Sprat Bay, the reefs off St. Thomas also tended to have smaller snappers and parrotfish relative to the other NVI sites. One site in Great Lameshur Bay (VIERS) had smaller groupers than the other shallow sites off St. John. Similarly, Cane Bay generally had smaller snappers than the other two reefs off St. Croix (Table 5).

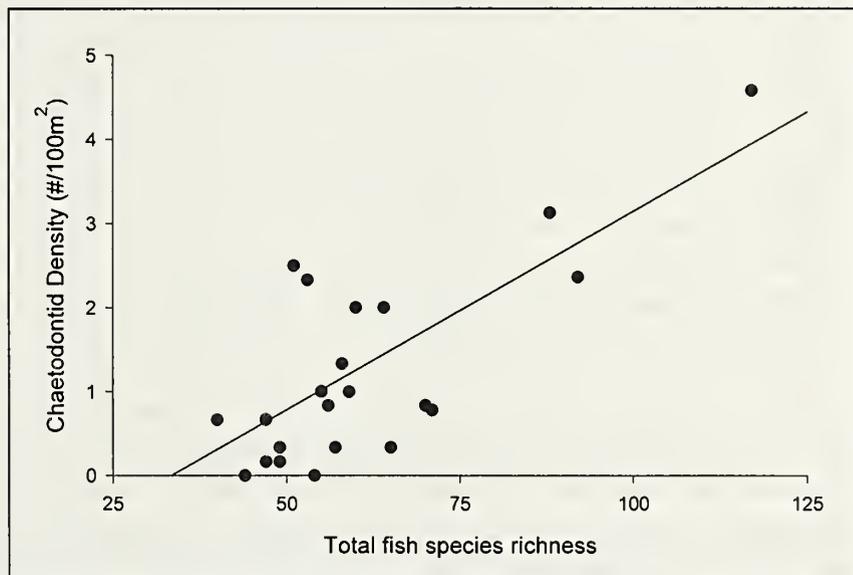
A series of multiple regression analyses were conducted between the average fish density or the total fish species richness and each of the following: depth, live stony coral cover and, for "large" ( $\geq 25$  cm diameter) stony corals, mean size and recent partial-colony mortality (Nemeth et al., this volume). Only one of these comparisons was significant. Total fish species richness (using results from a single roving diver survey per site) showed a positive relationship with percent live stony coral cover (Fig 5). Upon further analysis of fish guilds or individual species, some other interesting patterns emerged. A significant negative relationship was found between the density of herbivorous fishes greater than 5 cm in length and macroalgal index calculated from percent absolute abundance of macroalgae (Fig 6). This relationship remained significant even when herbivore density was compared to macroalgal index calculated with percent relative abundance of macroalgae ( $r^2 = 0.28$ ,  $p < 0.04$ ) or when compared directly to absolute macroalgal abundance ( $r^2 = 0.64$ ,  $p < 0.001$ ). Significant positive relationships were also found between the density of butterflyfish (Chaetodontidae) and total fish species richness from either a single roving diver survey per site ( $r^2 = 0.19$ ,  $p < 0.04$ ) or from results of multiple roving diver surveys (Fig. 7). However, no significant relationship was found between the density of chaetodontids and percent live stony coral cover ( $p < 0.62$ ). A significant positive relationship was found between the density of angelfish (Pomacanthidae) and depth (Fig. 8). Finally an interesting negative relationship was found between the density of large herbivores ( $\geq 10$  cm) and the percent of large stony corals occupied by territorial damselfishes (*Stegastes planifrons*, *S. diencaeus* and *S. fuscus*; Fig. 9).



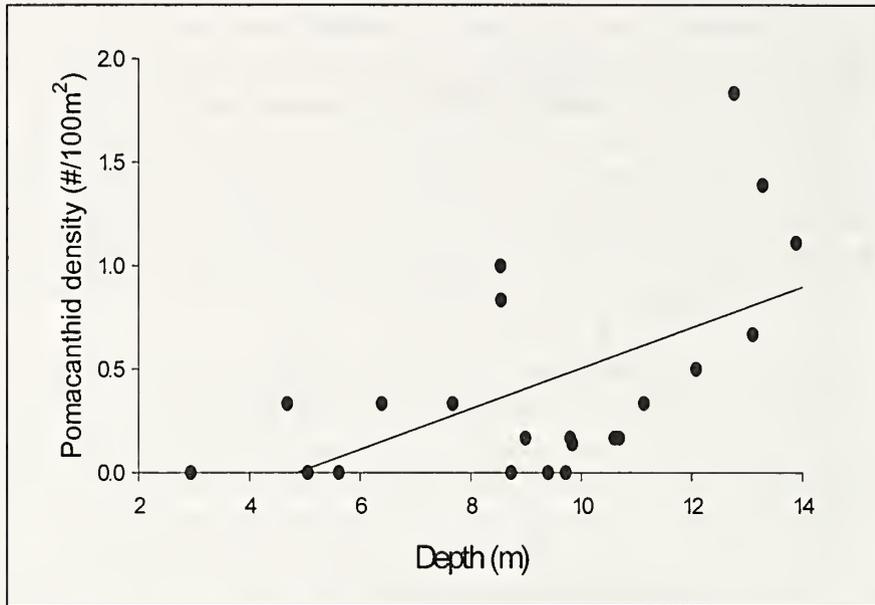
**Figure 5.** Regression between total fish species richness and percent live stony coral cover ( $r^2 = 0.30$ ,  $p < 0.01$ ), by site in the Virgin Islands.



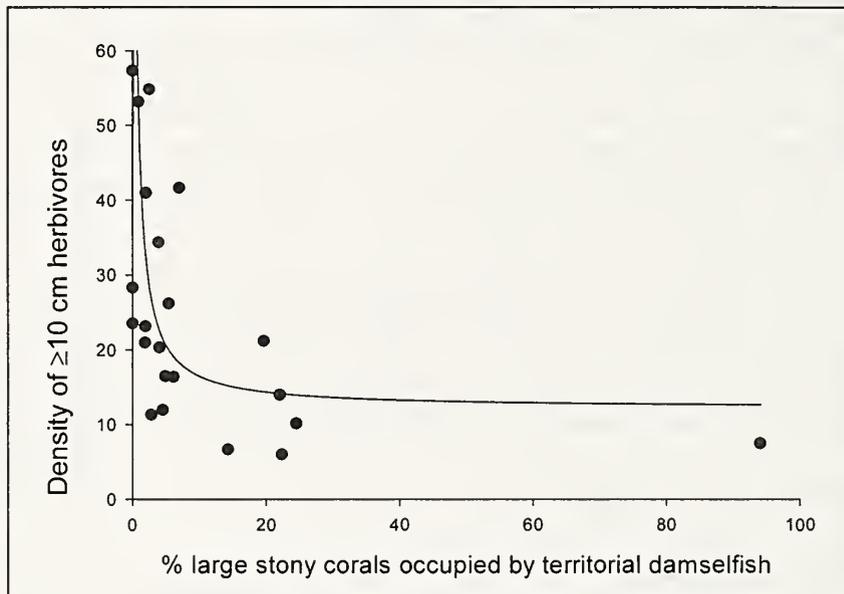
**Figure 6.** Regression of herbivore (acanthurids, scarids  $\geq 5$  cm, *Microspathodon chrysurus*) density versus the macroalgal index (absolute abundance of macroalgae x macroalgal height) by site in the Virgin Islands.  $r^2 = 0.24$ ,  $F_{20,1} = 6.33$ ,  $p < 0.02$ .



**Figure 7.** Regressions between chaetodontid density and total fish-species richness ( $r^2 = 0.564$ ,  $< 0.0001$ ) by site in the Virgin Islands.



**Figure 8.** Regression of pomacanthid density and depth ( $r^2 = 0.32$ ,  $p < 0.005$ ) by site in the Virgin Islands.



**Figure 9.** Relationship between the density of adult ( $\geq 10$  cm length) herbivores (acanthurids, scarids, *Microspathodon chrysurus*) and the percent of large ( $\geq 25$  cm) stony corals occupied by territorial damselfish by site in the Virgin Islands. Nonlinear stepwise regression analysis produced a significant fit ( $r^2 = 0.97$ ,  $p < 0.0001$ ) with an inverse second order quadratic equation ( $F = y_0 + a/x + b/x^2$ ).

## DISCUSSION

Fish assemblage structure was relatively similar among island groups within the USVI and BVI. Herbivorous fishes, especially the parrotfishes and surgeonfishes, dominated the reefs at all sites and, overall ( $\geq 5$  cm scarids only), comprised 72% of the

fish fauna. The shallow St. John sites deviated from this pattern with the  $\geq 5$  cm Haemulidae showing a five-fold increase in abundance relative to the deeper reef areas throughout the region. The abundance of haemulids in shallow reefs can be attributed most likely to the proximity of seagrass beds (Parrish, 1989; Muehlstein and Beets, 1992; Nagelkerken et al., 2000a,b,c, 2001), which are prevalent along the south coast of St. John. Juvenile scarids were a major component (24% to 77%) of parrotfish densities in Anegada and the NVI. There was a general increase in density of juvenile parrotfish from the easternmost reefs of Anegada to the shallow reefs of St. John, which may indicate an upstream source of larval fish recruitment (Roberts, 1997). However, a weak negative relationship with depth or the lower number of large carnivores on the shallow reefs of St. John may also explain some of the spatial variation in juvenile scarid densities. Although temporal variation in recruitment patterns may have influenced juvenile scarid densities on the shallow St. John reefs, many of the fish surveys were conducted during similar time periods.

The remaining selected fish families [i.e., snappers, groupers, angelfish, butterflyfish, and triggerfish (balistids)] were similar in abundance among the deep reef areas but much less common in the shallow St. John sites. Total fish species richness was also similar among the island groups, although estimates were greatly influenced by the number of sites visited or the number of roving diver surveys performed at a site. Based on species accumulation curves, a minimum of six roving diving surveys or more than seven hours search time would be needed to approximate the actual species diversity at a site. Although this level of effort would not be practical considering that AGRRA is supposed to be a rapid assessment technique, it is recommended that future AGRRA surveys conduct at least one, but preferably two, 60-minute roving diver surveys.

Multiple regression analyses detected some interesting regional patterns of abundance associated with benthic parameters which accounted for a significant portion of the variability in total fish species richness and in the density of butterflyfish and angelfish. The significant relationship between total fish species richness and percent cover of live stony corals (Fig. 5) suggests that species packing occurs where well-developed reefs provide greater structural complexity (Risk, 1972; Luckhurst and Luckhurst, 1978; Bell and Galzin, 1984; Roberts and Ormond, 1987). In earlier studies of chaetodontid fish assemblages around the world, significant relationships were found between their densities and the percent living coral cover and coral diversity (Bell et al., 1985; Bouchon-Navaro et al., 1985; Bouchon-Navaro and Bouchon, 1989; Roberts et al., 1992). Our analysis differed from these results in that the density of butterflyfishes was not associated with any benthic factor. A positive relationship between butterflyfish density and total fish species richness in the Virgin Islands (Fig. 7) complemented the results of Roberts et al. (1992), who found a positive relationship between the density and total abundance of butterflyfishes and species richness in the Red Sea.

Important differences in fish size distribution patterns were apparent among the island groups. Within the Virgin Islands archipelago, Anegada is the most remote and least populated island, and its reefs probably experience the lowest level of fishing pressure relative to the other islands. Moreover, large portions of the extensive reef system surrounding Anegada, along with 12 other sites in the BVI, are marine conservation parks protected by the BVI National Parks Trust. All commercial and recreational fishing is prohibited within marine park boundaries (BVI National Parks

Trust, unpublished). These factors probably contributed to a greater abundance of the large-size classes of predators and herbivores in Anegada as well as in the two other BVI reefs (Iguana Head, Guana Island and Eustatia Reef, Virgin Gorda).

Patterns also emerged when the average sizes of several commercially important fish families were compared among islands. In general, most sites off St. Thomas had smaller groupers, snappers, grunts and parrotfishes than deeper reefs off other islands (i.e., St. John, St. Croix, Virgin Gorda and Guana Island). These differences were probably due to a suite of factors specific to each island. For example, St. Thomas has the greatest human population density of all the islands, the largest number of seasonal workers from other Caribbean islands (who often supplement their diet by fishing in near shore waters), and the highest number of tourists who participate in recreational or sport fishing activities (US Department of Commerce, 2000; Virgin Islands Department of Fish and Wildlife, 2001, unpublished report). Fishing regulations, primary target species, and gear types may also influence harvest rates. In a survey of recreational fishers in the Virgin Islands, Jennings (1992) reported that nearly 65% of the fishers targeted groupers and snappers. He also found that up to 8% speared fish in the St. Thomas/St. John area, whereas none spear fished in St. Croix. In contrast, spear fishing is prohibited throughout the BVI and within the Virgin Islands National Park waters surrounding approximately 50% of St. John. In the Florida Keys, sites protected from spear fishing had higher densities and larger sizes of several species especially snappers and grunts (Roberts and Polunin, 1993). Even low levels of protection within an area have resulted in increased biomass of commercially important fishes (Bohnsack, 1996; Roberts and Hawkins, 1997). In a recreational fisher survey conducted in the USVI in 2001 by the Virgin Islands Division of Fish and Wildlife, 67% of the people interviewed indicated that they spear fished or bottom fished less than 3 miles from shore (unpublished report). These and other factors would greatly intensify the fishing pressure on near-shore stocks and result in over-harvest around the St. Thomas area (Jennings, 1992).

By focusing on the variation in fish size within each island or island group, we also observed that some sites had consistently smaller fish across most commercially important families (Table 5). These sites included the VIERS shallow reef and Tektite deep reef (St. John), Brewer's Bay (St. Thomas), Cane Bay (St. Croix) and Jack Bay (Anegada). Although many natural factors may contribute to the impacts on reef fisheries, one of the most important similarities among all of these sites seems to be accessibility to skin or scuba divers. Easy access from shore and popularity of a dive site (i.e., presence of a public mooring) will concentrate fishing activity in these locations. This is especially true when commercial dive charters allow patrons to spear fish. VIERS reef, Brewer's Bay, Cane Bay and Jack Bay are unique among all the sites surveyed since they are easily accessible from shore by skin and scuba divers who may participate in spear fishing (R. Nemeth, personal observation). Brewer's Bay, in particular, is a beach frequently (e.g. weekly) used by local spear fishermen (R. Nemeth, personal observation). Tektite reef is also a popular dive destination, and may experience increased fishing pressure even though it is within the Virgin Islands National Park. The fact that Rogers and Beets (2001) found no apparent difference in fish assemblage structure inside or outside the national park boundaries indicates that fishing pressure is still substantial at some sites within the park.

Further support of the hypothesis that size structure is largely determined by access to divers can be gained by looking at reefs off islands that consistently had larger fish in most commercially important families. Sites such as Caret Bay (St. Thomas) and Fish Bay East Outer (St. John) are rarely, if ever, visited by skin or scuba divers, and Salt River (St. Croix), although a popular dive area, is within the Salt River Bay National Historic Park and Ecological Preserve. Thus fishing pressure at these sites is very low relative to more popular or unprotected sites.

Overall densities of parrotfishes and surgeonfishes in the shallow and deeper reefs of the Virgin Islands were comparable to those reported by Lewis and Wainwright (1985) with the exception that parrotfishes of all sizes in the shallow St. John sites were 7.5 times more abundant than in Belize (50.9 versus 6.7 scarids/100m<sup>2</sup>). The significant negative relationship between herbivore density and macroalgal index (Fig. 6) suggests that grazing pressure of herbivorous fishes is an important factor in determining the structure of tropical benthic communities (Lewis and Wainwright, 1985; Lewis, 1985, 1986). Grazing pressure was strongly correlated to herbivore density in Belize (Lewis, 1985, 1986). Lewis (1986) showed a nearly 30% increase in the cover of macroalgae (primarily *Padina* and *Dictyota*) relative to controls when herbivorous fishes were excluded from experimental plots in back-reef habitats in Belize. Total macroalgal index calculated from Lewis (1986) was 150 within shallow back-reef exclusion cages compared to an average macroalgal index of 126 on shallow reefs and 99 on deeper reefs of the Virgin Islands archipelago (Nemeth et al., this volume).

In the Virgin Islands, the density of large ( $\geq 10$  cm) herbivores (parrotfishes and surgeonfishes) was negatively related to the percent of large stony corals occupied by territorial damselfishes (Fig. 9). The presence of territorial damselfishes, which aggressively defend their algal turfs, has been shown to have a dramatic effect on the social behavior, abundance, and feeding strategies of other herbivorous competitors (Williams, 1979; Hourigan, 1986). In the Caribbean and Pacific, several studies have shown that parrotfishes and surgeonfishes form large schools to feed in the territories of other species, especially territorial damselfishes (Vine, 1974; Robertson et al., 1976; Hourigan, 1986). Robertson et al. (1976) found that schooling parrotfishes and surgeonfishes, which settled and fed en masse within damselfish territories, had higher feeding rates and fewer aggressive episodes than non-schooling individuals. The experimental removal of territorial damselfish within a defined area typically results in a rapid increase in the abundance of other herbivorous fishes (Robertson et al., 1976; Hourigan, 1986; Nemeth, unpublished data). The negative relationship between the density of large herbivores and the density of territorial damselfish could be due to an actual competitive exclusion of herbivores in the area and/or an artifact of the belt-transect method, which may underestimate the densities of schooling herbivores. By comparing the relative densities of herbivores from fish transects and roving diver surveys, we found that density estimates were comparable in magnitude at 85% (11/13) of the sites where damselfish occupied less than 10% of the coral heads but in only one of the six (17%) sites where damselfish occupied more than 10% of the coral heads. This suggests that, in areas with high densities of territorial damselfishes, the abundance of large herbivores were underestimated using fish transects since roving schools of parrotfishes and surgeonfishes can move unpredictably over large areas of coral reef (Robertson et al. 1976).

Considering the wide range of benthic conditions, human factors, and management regulations that could potentially influence fish assemblage structures, we were surprised to find that the abundance and size structure of commercially and ecologically important species were relatively similar among island groups of the Virgin Islands. Within the island groups, the primary notable differences were the larger size structure of herbivores and carnivores in Anegada and the greater abundance of haemulids in the shallow reefs of St. John. The significant relationships between selected fish families and various benthic parameters generally supported the observations of previous studies within the Caribbean and other tropical regions. Within each island, site-specific differences in fish-size structure may be explained by variations in fishing pressure from recreational fishermen, including spear fishers.

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Table 1. Site information for AGRRA fish surveys in the Virgin Islands.

Site name	Site code	Reef Type <sup>1</sup> / Exposure	Latitude (° ' " N)	Longitude (° ' " W)	Survey Date(s)	Depth (m)	>25 cm stony corals (#/10m) <sup>2</sup>	% live stony coral cover (mean ± sd) <sup>2</sup>	# 30 m fish transects (# RDT) <sup>3</sup>	Total fish species (#) <sup>4</sup>
<b>Northern Virgin Islands</b>										
<i>St John (shallow)</i>										
Great Lameshur, Donkey	12	F/W	18 18.853	64 43.312	May 26 00	3	8.5	18.5 ± 3.5	10 (1)	57
Great Lameshur, VIERS	13	F/W	18 19.094	64 43.390	Oct 23 99	5.5	4	13.5 ± 6.0	10 (1)	47
Fish Bay East, Inner	11	F/W	18 19.073	64 45.808	Nov 14 99	5	3.5	10.5 ± 6.0	10 (1)	44
Fish Bay West, Inner	14	F/W	18 19.053	64 45.878	Aug 15 00, Oct 24 00	5	8.5	7.0 ± 6.0	10 (1)	40
<i>St. John (deep)</i>										
Great Lameshur, Tektite	15	F/W	18 18.572	64 43.302	Aug 3 99, Nov 29 99	11	10.5	48.5 ± 13.5	10 (1)	70
Great Lameshur, Yawzi	16	F/W	18 18.831	64 43.596	July 23 99	12.5	4.5	16.0 ± 8.0	10 (1)	59
Fish Bay East, Outer	9	F/W	18 18.948	64 45.777	Feb 08 00	6.5	4.5	12.0 ± 5.0	10 (1)	56
Fish Bay West, Outer	10	F/W	18 18.850	64 45.845	Aug 02 99	7.5	4	13.0 ± 6.5	10 (1)	47
<i>St. Thomas</i>										
Brewer's Bay	2	F/W	18 20.670	64 59.157	May 26 99, June 9 00	8.5	6.5	19.0 ± 6.5	10 (2)	58 (65) <sup>5</sup>
Buck Island	4	F/W	18 19.781	64 57.097	May 19 98, July 14 99	14	3.5	11.5 ± 6.0	16 (1)	71
Caret Bay	1	F/W	18 22.421	64 59.371	May 21 98, July 17 00	9.5	6	23.5 ± 8.5	10 (1)	58
Flat Cay	3	F/W	18 19.072	64 59.444	July 7 99, June 12 00	12	6	21.0 ± 7.5	10 (1)	51
Sprat Bay	5	F B/W	18 19.718	64 55.630	Aug 31 00	10.5	13	29.5 ± 7.0	10 (1)	54
<i>Virgin Gorda</i>										
Eustatia Reef	18	F B/W	18 30.50	64 20.250	July 24 00	9	9.5	17.0 ± 6.0	10 (1)	60
<i>Guana Island</i>										
Iguana Head	17	F/W	18 28.477	64 34.941	Aug 06 99	10	8.5	30.0 ± 9.5	10 (1)	64
<i>St. Croix</i>										
Cane Bay	6	F/W	17 46.230	64 48.530	Oct 14 99, Dec 16 99	10	7.5	30.5 ± 11.0	12 (3)	74 (92) <sup>5</sup>
Long Reef	8	F B/W	17 46.118	64 41.490	Oct 13 99	13.5	7.5	16.0 ± 3.5	24 (5)	68 (88) <sup>5</sup>
Salt River East	7	F/W	17 47.227	64 45.330	Oct 13 99	10	6	14.0 ± 4.0	24 (6)	63 (117) <sup>5</sup>
<i>Anegada</i>										
Herman's Reef	19	B B /L	18 33.841	64 14.320	July 24 00	13	10	19.5 ± 5.5	10 (1)	53
Horseshoe Reef	20	B B /W	18 39.965	64 13.890	July 22 00	10.5	6	14.0 ± 6.5	10 (1)	49
Jack Bay	21	F B/W	18 44.961	64 19.246	July 22 00	9	4.5	8.0 ± 4.5	10 (1)	49
W. Cow Wreck	22	F B/W	18 45.164	64 24.596	July 23 00	8.5	7	11.5 ± 3.5	10 (1)	55

<sup>1</sup>F = fringing; F B = fringing barrier; B B = bank barrier; W = windward, L = leeward

<sup>2</sup>From Nemeth et al. (this volume)

<sup>3</sup>RDT = roving diver technique surveys

<sup>4</sup>Number of fish species based on the maximum number seen on any one RDT survey.

<sup>5</sup>Number in parenthesis = cumulative number of species for multiple RDT surveys.

Table 2. Twenty-five most frequently sighted fish species during roving diver surveys for all sites combined in the Virgin Islands, with density (mean  $\pm$  standard deviation) for species in belt transects.

Fish Species	Rank <sup>1</sup>	Sighting Frequency <sup>2</sup>	Density (#/100m <sup>2</sup> )
<i>Thalassoma bifasciatum</i>	1.	100	-
<i>Acanthurus coeruleus</i>	2.	100	7.0 $\pm$ 6.58
<i>Sparisoma aurofrenatum</i>	3.	100	4.4 $\pm$ 3.03
<i>Sparisoma viride</i>	4.	100	3.6 $\pm$ 2.42
<i>Haemulon flavolineatum</i>	5.	100	3.3 $\pm$ 3.90
<i>Scarus taeniopterus</i>	6.	94.7	4.8 $\pm$ 8.82
<i>Stegastes partitus</i>	7.	89.5	-
<i>Mulloidichthys martinicus</i>	8.	86.8	-
<i>Epinephelus cruentatus</i>	9.	81.6	0.7 $\pm$ 1.04
<i>Serranus tigrinus</i>	10.	81.6	0
<i>Acanthurus bahianus</i>	11.	80.7	7.7 $\pm$ 7.94
<i>Halichoeres garnoti</i>	12.	80.7	-
<i>Chaetodon capistratus</i>	13.	80.7	1.0 $\pm$ 0.95
<i>Scarus croicensis</i>	14.	78.1	3.6 $\pm$ 3.12
<i>Chromis multilineata</i>	15.	75.4	-
<i>Chromis cyanea</i>	16.	75.4	-
<i>Holocentrus rufus</i>	17.	70.2	-
<i>Epinephelus fulvus</i>	18.	68.4	0.7 $\pm$ 1.03
<i>Abudefduf saxatilis</i>	19.	62.3	-
<i>Stegastes diencaeus</i>	20.	59.6	-
<i>Canthigaster rostrata</i>	21.	58.8	-
<i>Coryphopterus glaucofraenum</i>	22.	57.9	-
<i>Myripristis jacobis</i>	23.	57.0	-
<i>Stegastes planifrons</i>	24.	56.1	-
<i>Halichoeres maculipinna</i>	25.	56.1	-

<sup>1</sup>Rank of species with the same sighting frequency was determined by calculating the REEF density index for each species based upon the weighted abundance categories: S (single=1), F (few = 2), M (many = 3) and A (abundant = 4). Equation to calculate weighted density average was:

$$\text{Density} = [(S*1) + (F*2) + (M*3) + (A*4)] / (\text{number of surveys in which species was observed})$$

<sup>2</sup>Sighting frequency (%SF) for each species was calculated using the equation:

$$\%SF = [S + F + M + A] / (\text{number of surveys})$$

Table 3. The total number and density (mean  $\pm$  standard deviation) of the twenty-five most common species counted in belt transects for all sites combined in the Virgin Islands.

Fish species	Rank	AGRRA fishes without <5 cm scarids + haemulids		AGRRA fishes with <5 cm scarids + haemulids <sup>1</sup>	
		Sum (#)	Density (#/100 m <sup>2</sup> )	Sum (#)	Density (#/100 m <sup>2</sup> )
<i>Acanthurus bahianus</i>	1	1176	7.7 $\pm$ 7.94		
<i>Acanthurus coeruleus</i>	2	1070	7.0 $\pm$ 6.58		
<i>Scarus taeniopterus</i>	3	743	4.8 $\pm$ 8.82	1640	10.5
<i>Sparisoma aurofrenatum</i>	4	677	4.4 $\pm$ 3.03	911	6
<i>Scarus croicensis</i>	5	556	3.6 $\pm$ 3.12	869	5.5
<i>Sparisoma viride</i>	6	549	3.6 $\pm$ 2.42	920	6
<i>Haemulon flavolineatum</i>	7	510	3.3 $\pm$ 3.90	4926	32
<i>Haemulon aurolineatum</i>	8	309	2.0 $\pm$ 5.35	311	2
<i>Acanthurus chirurgus</i>	9	294	1.9 $\pm$ 5.05		
<i>Ocyurus chrysurus</i>	10	204	1.3 $\pm$ 2.17		
<i>Microspathodon chrysurus</i>	11	191	1.2 $\pm$ 2.11		
<i>Sparisoma rubripinne</i>	12	153	1.0 $\pm$ 1.82	166	1
<i>Chaetodon capistratus</i>	13	147	1.0 $\pm$ 0.95		
<i>Epinephelus cruentatus</i>	14	113	0.7 $\pm$ 1.04		
<i>Epinephelus fulvus</i>	15	113	0.7 $\pm$ 1.03		
<i>Haemulon plumieri</i>	16	104	0.7 $\pm$ 1.75	108	0.7
<i>Haemulon sciurus</i>	17	100	0.6 $\pm$ 1.94	100	0.6
<i>Lutjanus mahogani</i>	18	93	0.6 $\pm$ 1.56		
<i>Scarus vetula</i>	19	79	0.5 $\pm$ 0.76	285	2.05
<i>Caranx ruber</i>	20	77	0.5 $\pm$ 1.06		
<i>Holocanthus ciliaris</i>	21	74	0.5 $\pm$ 0.95		
<i>Lutjanus apodus</i>	22	74	0.5 $\pm$ 1.03		
<i>Lutjanus griseus</i>	23	59	0.4 $\pm$ 1.40		
<i>Lutjanus jocu</i>	24	57	0.4 $\pm$ 1.91		
<i>Melichthys niger</i>	25	57	0.4 $\pm$ 1.12		

Except St. Croix, where  $\leq$ 5 cm scarids and haemulids were not counted.

Table 4. Density (mean  $\pm$  standard deviation) of AGRRA fishes by site in the Virgin Islands.

Site name	Site code	Density (#/100 m <sup>2</sup> )						Macroalgal index Rel. (Abs.) <sup>3</sup>
		Herbivores			Carnivores			
		Acanthuridae	Scaridae ( $\geq 5$ cm)	<i>M. chrysurus</i> <sup>1</sup>	Haemulidae ( $\geq 5$ cm)	Lutjanidae	Serranidae <sup>2</sup>	
<b>Northern Virgin Islands</b>								
<i>St. John (shallow)</i>								
Great Lameshu Donkey	12	7.5 $\pm$ 7.0	10.8 $\pm$ 7.3	0	28.5 $\pm$ 25.8	2.5 $\pm$ 5.5	0.6 $\pm$ 1.0	71 (80)
Great Lameshu VIERS	13	8.0 $\pm$ 5.0	17.7 $\pm$ 12.7	0	32.5 $\pm$ 29.4	3.0 $\pm$ 2.5	0.5 $\pm$ 2.0	46 (72)
Fish Bay East Inner	11	28.5 $\pm$ 50.5	14.0 $\pm$ 16.3	0.3 $\pm$ 1.0	32.3 $\pm$ 23.1	10.5 $\pm$ 15.0	0.2 $\pm$ 0.5	108 (116)
Fish Bay West Inner	14	8.5 $\pm$ 13.0	3.8 $\pm$ 4.0	0	3.5 $\pm$ 5.5	4.5 $\pm$ 7.5	0.4 $\pm$ 1.0	278 (181)
<i>St. John (deep)</i>								
Great Lameshu Tektite	15	5.5 $\pm$ 3.0	13.7 $\pm$ 10.8	0.2 $\pm$ 0.5	5.0 $\pm$ 5.5	10.0 $\pm$ 29.5	2.5 $\pm$ 2.0	190 (112)
Great Lameshu Yawzi	16	10.0 $\pm$ 4.5	48.8 $\pm$ 30.1	0.2 $\pm$ 0.5	15 $\pm$ 17.2	2.0 $\pm$ 2.0	1.0 $\pm$ 1.5	55 (31)
Fish Bay East Outer	9	60.5 $\pm$ 78.0	6.8 $\pm$ 5.0	0	8.5 $\pm$ 17.3	3.0 $\pm$ 3.5	0.4 $\pm$ 1.0	6 (7)
Fish Bay West Outer	10	45.0 $\pm$ 57.0	22.2 $\pm$ 17.3	2.3 $\pm$ 1.4	7.0 $\pm$ 18.8	3.6 $\pm$ 3.0	0.7 $\pm$ 1.0	52 (30)
<i>St. Thomas</i>								
Brewer's Bay	2	1.0 $\pm$ 1.5	40.7 $\pm$ 58.2	0	2.0 $\pm$ 4.3	1.0 $\pm$ 2.0	0.5 $\pm$ 0.5	83 (118)
Buck Island	4	3.5 $\pm$ 3.0	9.7 $\pm$ 8.9	0.1 $\pm$ 0.4	7.0 $\pm$ 17.4	3.0 $\pm$ 5.5	2.0 $\pm$ 2.5	173 (96)
Caret Bay	1	9.5 $\pm$ 5.0	14.5 $\pm$ 10.7	1.7 $\pm$ 1.4	0.5 $\pm$ 0.7	0.5 $\pm$ 1.0	1.0 $\pm$ 1.0	110 (82)
Flat Cay	3	8.0 $\pm$ 4.5	28.0 $\pm$ 13.9	0	2.7 $\pm$ 3.5	2.0 $\pm$ 3.0	1.5 $\pm$ 1.0	93 (53)
Sprat Bay	5	4.0 $\pm$ 3.0	17.8 $\pm$ 15.4	0.5 $\pm$ 0.8	1.0 $\pm$ 1.4	11.5 $\pm$ 31.0	1.0 $\pm$ 1.0	247 (201)
<i>Virgin Gorda</i>								
Eustacia Reef	18	21.0 $\pm$ 44.5	12.0 $\pm$ 12.8	4.3 $\pm$ 3.1	1.8 $\pm$ 2.9	5.5 $\pm$ 12.0	0.5 $\pm$ 0.5	116 (86)
<i>Guana Island</i>								
Iguana Head	17	13.0 $\pm$ 15.0	13.0 $\pm$ 5.3	0.7 $\pm$ 1.2	1.5 $\pm$ 2.5	7.0 $\pm$ 7.5	2.5 $\pm$ 2.0	8 (8)
<i>St. Croix</i>								
Cane Bay	6	5.5 $\pm$ 4.0	14.5 $\pm$ 7.5	2.4 $\pm$ 1.9	9.5 $\pm$ 15.0	0.5 $\pm$ 1.0	5.5 $\pm$ 2.5	35 (70)
Long Reef	8	14.0 $\pm$ 6.5	19.5 $\pm$ 13.0	1.6 $\pm$ 1.4	3.5 $\pm$ 3.5	1.5 $\pm$ 1.5	3.5 $\pm$ 2.5	25 (19)
Salt River East	7	14.5 $\pm$ 8.5	10.0 $\pm$ 6.0	0.4 $\pm$ 1.0	2.0 $\pm$ 1.5	0.5 $\pm$ 1.0	3.0 $\pm$ 2.5	3 (2)
<i>Anegada</i>								
Herman's Reef	19	13.5 $\pm$ 6.0	27.7 $\pm$ 10.5	3.5 $\pm$ 2.1	3.2 $\pm$ 2.7	0	2.0 $\pm$ 2.5	104 (55)
Horseshoe Reef	20	40.5 $\pm$ 14.5	32.8 $\pm$ 32.4	1.2 $\pm$ 2.7	0	7.0 $\pm$ 21.0	0.5 $\pm$ 1.0	147 (70)
Jack Bay	21	26.0 $\pm$ 23.5	10.8 $\pm$ 7.6	9.0 $\pm$ 6.7	4.5 $\pm$ 5.2	7.5 $\pm$ 13.5	4.0 $\pm$ 3.0	101 (77)
W. Cow Wreck	22	32.5 $\pm$ 9.5	6.8 $\pm$ 4.5	0	5.5 $\pm$ 10.5	1.5 $\pm$ 3.5	1.5 $\pm$ 2.5	230 (150)

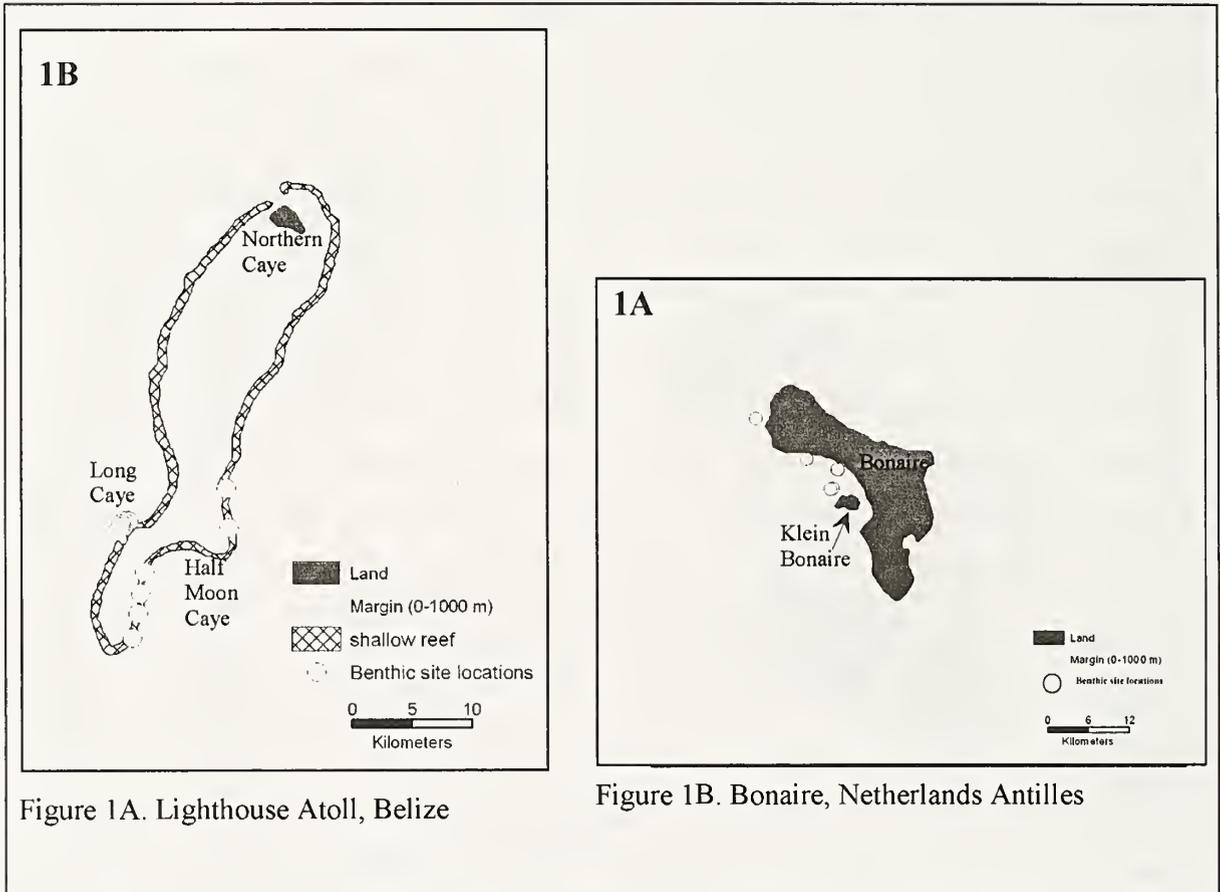
<sup>1</sup>*M. chrysurus* = *Microspathodon chrysurus*<sup>2</sup>*Epinephelus* spp. and *Mycteroperca* spp.<sup>3</sup>Rel = % relative macroalgal abundance x macroalgal height; Abs. = % absolute macroalgal abundance x macroalgal height; from Nemeth et al. (this volume).

Table 5. Length (mean  $\pm$  standard deviation) in cm of AGRRA fishes by site in the Virgin Islands.

Site name	Site code	Acanthuridae	Scaridae ( $\geq 5$ cm)	Haemulidae ( $\geq 5$ cm)	Lutjanidae	Serranidae <sup>1</sup>	Pomacanthidae	Rank <sup>2</sup>
<b>Northern Virgin Islands</b>								
<i>St. John (shallow)</i>								
Great Lameshur Donkey	12	10.5 $\pm$ 3.5	19.7 $\pm$ 13.5	16.1 $\pm$ 1.5	22.5 $\pm$ 4.5	24.5 $\pm$ 18.0		++
Great Lameshur VIERS	13	11.0 $\pm$ 4.0	16.0 $\pm$ 1.0	10.0 $\pm$ 2.0	14.0 $\pm$ 7.5	13.0 $\pm$ 0		---- +
Fish Bay East Inner	11	4.5 $\pm$ 2.5	17.9 $\pm$ 8.0	11.5 $\pm$ 6.0	16.5 $\pm$ 8.0	35.5		- +
Fish Bay West Inner	14	9.5 $\pm$ 6.0	18.0 $\pm$ 4.0	16.9 $\pm$ 4.0	20.5 $\pm$ 9.5	19.0 $\pm$ 23.5	25.5 $\pm$ 1.0	+
<i>St. John (deep)</i>								
Great Lameshur Tektite	15	11.0 $\pm$ 3.5	18.0 $\pm$ 4.0	14.0 $\pm$ 4.0	18.0 $\pm$ 14.0	17.0 $\pm$ 6.0	9.0 $\pm$ 9.0	----
Great Lameshur Yawzi	16	11.4 $\pm$ 2.5	17.5 $\pm$ 1.5	14.5 $\pm$ 6.0	20.0 $\pm$ 6.5	23.5 $\pm$ 7.5	13.5 $\pm$ 3.0	
Fish Bay East Outer	9	11.5 $\pm$ 3.5	21 $\pm$ 3.0	17.5 $\pm$ 2.5	21.5 $\pm$ 3.5	30.5 $\pm$ 7.0	15.5 $\pm$ 0	+++++
Fish Bay West Outer	10	10.0 $\pm$ 2.2	20.5 $\pm$ 5.5	17.4 $\pm$ 3.0	20.0 $\pm$ 7.0	24.0 $\pm$ 10.5	9.0 $\pm$ 9.0	-
<i>St. Thomas</i>								
Brewer's Bay	2	4.5 $\pm$ 3.0	10.0 $\pm$ 4.0	15.5 $\pm$ 0	11.0 $\pm$ 4.0	15.5 $\pm$ 0.0		---
Buck Island	4	8.5 $\pm$ 4.5	17.0 $\pm$ 3.0	11.5 $\pm$ 6.5	27.0 $\pm$ 18.5	15 $\pm$ 11.0	12.0 $\pm$ 5.5	-- ++
Caret Bay	1	13.0 $\pm$ 2.0	16.5 $\pm$ 2.0	20.5 $\pm$ 7.0	15.5 $\pm$ 0	19.0 $\pm$ 8.0		+++
Flat Cay	3	11.0 $\pm$ 3.0	12.0 $\pm$ 2.5	16.0 $\pm$ 5.0	15.5 $\pm$ 0	17.0 $\pm$ 4.0	13.5 $\pm$ 2.5	
Sprat Bay	5	10.0 $\pm$ 2.5	18.0 $\pm$ 11.0	14.0 $\pm$ 3.5	22.0 $\pm$ 5.0	18.0 $\pm$ 10.5	25.5	
<i>Virgin Gorda</i>								
Eustacia Reef	18	16.5 $\pm$ 2.5	19.0 $\pm$ 5.0	18.0 $\pm$ 3.0	25.0 $\pm$ 2.5	20.5 $\pm$ 7.0	25.5 $\pm$ 0	
<i>Guana Island</i>								
Iguana Head	17	14.0 $\pm$ 4.0	19.0 $\pm$ 3.0	15.0 $\pm$ 9.0	23.0 $\pm$ 6.0	19.5 $\pm$ 4.5	35.5	
<i>St. Croix</i>								
Cane Bay	6	17.5 $\pm$ 3.5	21.0 $\pm$ 7.5	17.5 $\pm$ 2.5	15.5 $\pm$ 0	19.0 $\pm$ 4.0		--- +
Long Reef	8	15.5 $\pm$ 1.5	19.0 $\pm$ 3.5	18.5 $\pm$ 3.0	22.5 $\pm$ 5.0	20.0 $\pm$ 3.5	18.0 $\pm$ 6.0	-- +
Salt River East	7	16.0 $\pm$ 1.5	22.0 $\pm$ 4.5	21.0 $\pm$ 5.0	20.5 $\pm$ 7.0	21.0 $\pm$ 4.0	12.0 $\pm$ 5.5	+++
<i>Anegada</i>								
Herman's Reef	19	13.7 $\pm$ 2.0	16.8 $\pm$ 2.0	22.0 $\pm$ 5.0		26.0 $\pm$ 2.5	14.0 $\pm$ 16.0	-
Horseshoe Reef	20	10.5 $\pm$ 2.0	17.3 $\pm$ 2.0		33.5 $\pm$ 14.0	30.5 $\pm$ 18.0	15.5	++
Jack Bay	21	14.1 $\pm$ 2.5	20.0 $\pm$ 7.0	19.0 $\pm$ 4.0	25.5 $\pm$ 0	15.0 $\pm$ 5.5	2.5	--- +
W. Cow Wreck	22	8.0 $\pm$ 3.0	21.0 $\pm$ 5.0	23.0 $\pm$ 4.5	25.5 $\pm$ 0	20.5 $\pm$ 4.5	7.0 $\pm$ 6.0	-- ++

<sup>1</sup>*Epinephelus* spp. and *Mycteroperca* spp.

<sup>2</sup>Ranking = number of times a site within an island (i.e. St. Thomas) or group (e.g., St. John shallow) was represented by the smallest (-) or largest (+) value within the size range of a family. Angelfish were excluded due to the number of missing cells and Virgin Gorda and Guana Island were excluded since they only had one site per island.



**Figure 1.** AGRRA surey sites at (A) Lighthouse Atoll, Belize and (B) off Bonaire

**ASSESSMENT TABLES FOR ABACO, BAHAMAS (FISH), LIGHTHOUSE ATOLL, BELIZE (CORALS, ALGAE, FISHES), AND BONAIRE, NETHERLANDS ANTILLES (CORALS, ALGAE, FISHES)**

BY

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**INTRODUCTION**

Rapid assessments using the AGRRA protocol were conducted at three locations within the wider Caribbean during 1999. The results are presented here only as a series of data tables similar in format to other papers in this volume and were only used in the regional synthesis (Kramer, this volume). All data tables were generated from the AGRRA Access database using queries to generate site means based on either transect values (fish density, coral cover, coral density, *Diadema* density) or individual coral values (coral mortality, diameter, condition), or quadrat values (relative abundance of algae and canopy height of macroalgae).

The location of the sites for Lighthouse Atoll and Bonaire are shown in figures 1A and 1B (see Figure 1 in Feingold et al., this volume, for the Abaco site locations). Tables 1-2 show results of the fish assessments collected in Abaco, Bahamas (for benthic results, see Feingold et al., this volume). Tables 3-6 show stony corals, algae, and fish results from Lighthouse Atoll, Belize. Tables 7-10 show stony corals, algae, and fish results from Bonaire, Netherlands Antilles.

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Table 1. Site information for AGRRA fish surveys in Abaco, Bahamas.

Site name	Reef type	Latitude (o' N)	Longitude (o' W)	Survey date	Depth (m)	% live stony coral cover (mean $\pm$ sd) <sup>1</sup>	30 m fish transects (#)	AGRRA fishes	
								Density (#/100m <sup>2</sup> ) (mean $\pm$ sd)	Species (#)
Sandy Cay, Forereef	Fore-reef slope	26 23.870	76 59.310	Aug 9 1999	3.5	22.5 $\pm$ 11.5	11	113.7 $\pm$ 43.8	17
Sandy Cay, Backreef	Back reef	26 23.869	76 59.311	Aug 9 1999	4	14.0 $\pm$ 7.5	10	46.2 $\pm$ 18.1	16
Elbow Cay, South-inner	Outer reef crest	26 31.610	76 57.420	Aug 11 1999	6	14.5 $\pm$ 4.5	10	106.0 $\pm$ 23.0	ND
Elbow Cay, Middle	Outer reef crest	26 32.420	76 56.770	Aug 11 1999	5.5	13.0 $\pm$ 10.0	10	58.0 $\pm$ 12.8	13
Elbow Cay, North	Outer reef crest	26 35.900	76 56.970	Aug 16 1999	8.5	7.0 $\pm$ 3.5	10	79.2 $\pm$ 21.0	14
Fowl Cay-shallow	Outer reef crest	26 38.230	77 02.310	Aug 13 1999	4	12.0 $\pm$ 4.5	10	87.7 $\pm$ 39.1	14
Storr's Reef	Outer reef crest	26 34.650	76 57.540	Aug 16 1999	4.0	14.5 $\pm$ 7.5	10	93.8 $\pm$ 23.8	15
Man O' War Cay, N. of S. Channel	Outer reef crest	26 36.200	76 58.990	Aug 12 1999	5.5	10.5 $\pm$ 6.0	10	65.3 $\pm$ 35.8	22
Man O' War Cay, S. of S. Channel	Outer reef crest	26 35.840	76 58.580	Aug 12 1999	3.5	15.0 $\pm$ 5.5	10	71.7 $\pm$ 27.3	23
Lynyard Cay, North	Spur and groove	26 21.450	76 58.610	Aug 15 1999	8.0	14.0 $\pm$ 12.5	10	53.2 $\pm$ 13.1	15
Lynyard Cay, South	Spur and groove	26 21.210	76 58.680	Aug 15 1999	8.5	13.0 $\pm$ 6.0	10	68.7 $\pm$ 25.3	15
Elbow Cay, South-outer	Shallow pinnacles	26 31.940	76 57.020	Aug 10 1999	4	18.0 $\pm$ 5.5	9	103.9 $\pm$ 41.6	18
Fowl Cay-deep	Deep pinnacles	26 38.400	77 02.360	Aug 13 1999	12.5	17.0 $\pm$ 13.5	10	39.3 $\pm$ 20.8	18

<sup>1</sup>From Feingold et al. (this volume)

Table 2. Density (mean  $\pm$  standard deviation) of AGRRA fishes and macroalgal index, by site for Abaco, Bahamas.

Site name	Herbivores (#/100m <sup>2</sup> )			Carnivores (#/100m <sup>2</sup> )		Macroalgal index <sup>b</sup>
	Acanthuridae	Scaridae	Haemulidae	Lutjanidae	Serranidae	
Sandy Cay, Forereef	58.5 $\pm$ 30.1	27.8 $\pm$ 12.8	1.0 $\pm$ 1.8	0.2 $\pm$ 0.5	0	81
Sandy Cay, Backreef	27.8 $\pm$ 8.3	12.3 $\pm$ 7.9	0.7 $\pm$ 1.2	0.5 $\pm$ 1.1	0	64
Elbow Cay, South-inner	79.5 $\pm$ 21.1	14.2 $\pm$ 5.7	0.8 $\pm$ 1.6	0.7 $\pm$ 1.2	0	133
Elbow Cay, Middle	44.0 $\pm$ 8.2	9.5 $\pm$ 5.5	0	0.2 $\pm$ 0.5	0	114
Elbow Cay, North	50.3 $\pm$ 14.2	23.7 $\pm$ 12.4	0	0.5 $\pm$ 1.1	0	155
Fowl Cay-shallow	51.0 $\pm$ 21.8	22.8 $\pm$ 20.4	0.3 $\pm$ 1.1	4.8 $\pm$ 8.0	0.3 $\pm$ 0.7	184
Storr's Reef	62.8 $\pm$ 12.9	24.2 $\pm$ 16.5	0	0.7 $\pm$ 1.2	0.2 $\pm$ 0.5	72
Man O' War Cay, N. of S. Channel	37.7 $\pm$ 17.1	10.2 $\pm$ 9.3	1.7 $\pm$ 1.9	1.0 $\pm$ 1.2	0.8 $\pm$ 1.4	68
Man O' War Cay, S. of S. Channel	34.5 $\pm$ 11.5	16.5 $\pm$ 5.7	1.7 $\pm$ 2.2	5.7 $\pm$ 7.8	1.0 $\pm$ 1.6	106
Lynyard Cay, North	35.0 $\pm$ 9.4	13.2 $\pm$ 9.2	0	1.3 $\pm$ 1.5	2.0 $\pm$ 1.3	124
Lynyard Cay, South	46.0 $\pm$ 17.5	15.0 $\pm$ 13.1	0.3 $\pm$ 0.7	2.7 $\pm$ 3.1	1.3 $\pm$ 2.2	125
Elbow Cay, South-outer	84.8 $\pm$ 42.0	10.6 $\pm$ 4.6	0.4 $\pm$ 0.7	1.3 $\pm$ 2.2	0.2 $\pm$ 0.6	130
Fowl Cay-deep	18.8 $\pm$ 11.2	11.5 $\pm$ 9.9	0	3.8 $\pm$ 3.3	1.5 $\pm$ 1.8	141

<sup>b</sup> Macroalgae Index = relative percent abundance x height; data from Feingold et al., this volume

Table 3. Site information for AGRRA stony coral and algal surveys in Lighthouse Atoll, Belize.

Site name	Reef type	Latitude (°'N)	Longitude (°'W)	Survey date	Depth (m)	Benthic transects (#)	≥25cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)
Aquarium	Fore reef	17 13.558	87 36.300	9/6/1999	7.5	14	7.5 ± 1.5	16.0 ± 3.5
Babylon's Gardens	Fore reef	17 10.312	87 35.563	10/10/1999	9.5	8	9.0 ± 2.0	19.0 ± 6.0
Black Durgon	Fore reef	17 13.260	87 31.590	8/4/1999	8	12	8.5 ± 2.0	23.0 ± 8.0
Cathedral	Fore reef	17 13.311	87 36.483	9/11/1999	7	12	8.5 ± 2.0	22.0 ± 4.5
Driftwood Reef	Fore reef	17 15.035	87 31.693	8/5/1999	9	14	7.0 ± 1.5	13.5 ± 6.5
Gotham City	Fore reef	17 11.253	87 35.371	10/12/1999	8.5	8	10.5 ± 2.0	20.5 ± 5.5
Long Caye Ridge	Fore reef	17 13.100	87 36.585	9/13/1999	7	11	9.2 ± 2.0	17.5 ± 3.5
Marie's Promenade	Fore reef	17 08.305	87 35.936	10/6/1999	9	8	8.5 ± 1.5	17.0 ± 6.0
Painted Wall	Fore reef	17 13.558	87 36.099	7/29/1999	4.5	10	10.5 ± 3.5	36.5 ± 16.0
Silver Caves	Fore reef	17 13.441	87 36.408	9/14/1999	7	10	10 ± 3.0	18.5 ± 4.0
Surge City	Fore reef	17 9.4069	87 35.661	9/14/1999	7	7	10.0 ± 1.5	16.5 ± 5.0

Table 4. Size and condition (mean ± standard deviation) of all stony corals (≥25 cm diameter), by site in Lighthouse Atoll.

Site name	Stony corals		Partial colony surface mortality (%)				Stony corals (%)		
	#	Diameter (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased	
Aquarium	108	39.0 ± 14.0	1.0 ± 6.0	22.0 ± 21.5	23.5 ± 22.5	0	9.5	2.5	
Babylon's Gardens	73	39.5 ± 14.5	3.5 ± 12.0	22.5 ± 23.5	25.5 ± 26.0	0	27.5	2.5	
Black Durgon	102	49.0 ± 36.5	5.0 ± 16.5	29.5 ± 27.5	34.5 ± 30.0	0	13.0	0	
Cathedral	103	54.5 ± 29.5	2.5 ± 4.5	25.5 ± 24.5	28.0 ± 26.0	0	9.5	10.5	
Driftwood Reef	98	40.0 ± 12.0	2.0 ± 7.0	23.0 ± 24.0	25.0 ± 22.5	1.0	8.5	1.5	
Gotham City	84	40.5 ± 12.0	0.5 ± 2.5	25.5 ± 28.5	26.0 ± 29.0	1.0	12.0	2.5	
Long Caye Ridge	101	46.0 ± 26.5	2.0 ± 5.5	24.5 ± 24.0	26.5 ± 25.5	0	5.5	15.0	
Marie's Promenade	69	55.5 ± 59.5	0.5 ± 2.0	32.0 ± 29.0	32.5 ± 29.5	1.5	16.0	0	
Painted Wall	105	51.0 ± 26.5	7.5 ± 12.0	24.0 ± 22.5	31.5 ± 26.0	0	7.5	1.0	
Silver Caves	100	47.5 ± 25.0	3.5 ± 9.5	21.5 ± 24.5	25.0 ± 26.0	0	2.0	1.0	
Surge City	70	44.5 ± 17.5	1.0 ± 2.0	24.0 ± 24.0	25.5 ± 24.0	1.5	4.0	0	

\*ND = no data

Table 5. Algal and density of stony coral recruits and *Diadema antillarum* (mean  $\pm$  st. dev.), by site in Lighthouse Atoll.

Site name	Quadrats		Relative abundance (%)		Macroalgae		Coral recruits (#/0625m <sup>2</sup> )	<i>Diadema</i> (#/10m)
	(#)	Macroalgae	Turf algae	Crustose coralline algae	Height (cm)	Index <sup>1</sup>		
Aquarium	60	21.0 $\pm$ 15.5	52.0 $\pm$ 11.0	27.0 $\pm$ 10.5	2.5 $\pm$ 1.0	46.5 $\pm$ 20.5	0.5 $\pm$ 0.3	0
Babylon's Gardens	56	20.5 $\pm$ 17.0	45.0 $\pm$ 16.0	34.5 $\pm$ 17.0	1.5 $\pm$ 1.0	29.0 $\pm$ 11.0	0.2 $\pm$ 0.3	0
Black Durgon	60	18.0 $\pm$ 10.5	49.5 $\pm$ 8.0	32.5 $\pm$ 10.0	2.0 $\pm$ 0.5	31.5 $\pm$ 19.0	0.4 $\pm$ 0.2	0
Cathedral	50	12.0 $\pm$ 7.0	55.0 $\pm$ 3.0	32.5 $\pm$ 7.0	1.5 $\pm$ 0.5	19.0 $\pm$ 7.0	0.6 $\pm$ 0.5	0
Driftwood Reef	50	15.0 $\pm$ 12.0	57.0 $\pm$ 7.5	28.0 $\pm$ 11.0	2.0 $\pm$ 1.0	31.5 $\pm$ 16.5	0.4 $\pm$ 0.3	0
Gotham City	56	20.0 $\pm$ 15.0	47.0 $\pm$ 21.5	33.5 $\pm$ 21.5	1.5 $\pm$ 1.0	28.0 $\pm$ 7.5	0.3 $\pm$ 0.2	0
Long Caye Ridge	55	11.0 $\pm$ 7.0	57.0 $\pm$ 3.5	32.0 $\pm$ 7.5	2.0 $\pm$ 1.0	22.5 $\pm$ 7.5	0.9 $\pm$ 0.3	0
Marie's Promenade	49	10.5 $\pm$ 4.0	63.5 $\pm$ 7.0	26.0 $\pm$ 8.0	1.5 $\pm$ 0.5	15.5 $\pm$ 2.0	0.1 $\pm$ 0.1	0
Painted Wall	40	33.0 $\pm$ 14.0	39.5 $\pm$ 10.5	28.0 $\pm$ 13.5	3.0 $\pm$ 1.0	97.0 $\pm$ 32.5	0.4 $\pm$ 0.3	0
Silver Caves	50	9.0 $\pm$ 4.0	57.5 $\pm$ 3.0	33.5 $\pm$ 5.5	1.5 $\pm$ 0.5	14.0 $\pm$ 2.0	0.5 $\pm$ 0.4	0
Surge City	49	15.0 $\pm$ 10.0	54.0 $\pm$ 13.5	31.0 $\pm$ 15.0	1.5 $\pm$ 0.5	22.5 $\pm$ 7.5	0.1 $\pm$ 0.1	0

Table 6. Density (mean  $\pm$  standard deviation) of AGRRA fishes by site in Lighthouse Atoll, Belize.

Site name	AGRRA fish		Herbivores (#/100m <sup>2</sup> $\pm$ sd)			Carnivores (#/100m <sup>2</sup> $\pm$ sd)		
	30 m fish transects (#)	Total Density #/100m <sup>2</sup>	fish species (#)	Acanthurida <sup>e</sup>	Scaridae	Haemulidae	Lutjanidae	Serranidae
Aquarium	10	32.5 $\pm$ 11.5	24	9.5 $\pm$ 12.8	9.2 $\pm$ 5.1	2.7 $\pm$ 1.6	4.0 $\pm$ 3.6	2.0 $\pm$ 1.7
Babylon's Gardens	10	29.0 $\pm$ 10.5	24	7.8 $\pm$ 1.9	7.2 $\pm$ 5.1	2.2 $\pm$ 3.3	1.2 $\pm$ 1.6	2.5 $\pm$ 1.4
Black Durgon	10	64.0 $\pm$ 29.5	22	3.0 $\pm$ 3.4	5.8 $\pm$ 5.8	3.2 $\pm$ 2.7	3.0 $\pm$ 3.4	1.2 $\pm$ 1.6
Cathedral	10	27.5 $\pm$ 13.0	24	4.2 $\pm$ 3.4	9.3 $\pm$ 7.3	3.5 $\pm$ 4.0	2.5 $\pm$ 3.6	3.5 $\pm$ 2.1
Driftwood Reef	10	41.0 $\pm$ 17.0	23	18.8 $\pm$ 9.7	6.0 $\pm$ 5.9	2.3 $\pm$ 2.4	0.0 $\pm$ 0.0	1.2 $\pm$ 2.1
Gotham City	10	20.5 $\pm$ 11.0	23	7.7 $\pm$ 3.9	4.2 $\pm$ 2.7	2.5 $\pm$ 4.1	0.8 $\pm$ 1.4	1.7 $\pm$ 1.8
Long Caye Ridge	10	27.5 $\pm$ 7.5	27	2.0 $\pm$ 1.9	9.8 $\pm$ 4.5	2.8 $\pm$ 2.5	3.0 $\pm$ 2.6	3.8 $\pm$ 2.9
Marie's Promenade	10	32.5 $\pm$ 10.5	23	9.0 $\pm$ 3.4	4.3 $\pm$ 4.3	1.8 $\pm$ 1.2	1.3 $\pm$ 3.2	0.5 $\pm$ 0.8
Painted Wall	10	43.5 $\pm$ 14.0	25	8.8 $\pm$ 14.6	14.7 $\pm$ 4.8	3.5 $\pm$ 1.7	5.0 $\pm$ 6.9	4.3 $\pm$ 2.6
Silver Caves	10	25.0 $\pm$ 5.5	24	3.7 $\pm$ 2.5	8.2 $\pm$ 4.2	3.3 $\pm$ 1.4	1.5 $\pm$ 2.5	2.2 $\pm$ 1.4
Surge City	10	26.5 $\pm$ 14.0	22	10.3 $\pm$ 7.3	4.3 $\pm$ 3.5	1.7 $\pm$ 1.8	0.8 $\pm$ 1.2	1.0 $\pm$ 1.2

Table 7. Site information for AGRRA stony coral and algal surveys in Bonaire.

Site name	Reef type	Latitude (° N)	Longitude (° W)	Survey date	Depth (m)	Benthic transects (#)	≥25cm stony corals (#/10 m)	% live stony coral cover (mean ± sd)
Barcardera	Fore reef	12 12.030	68 18.500	Feb. 4 1999	9.5	11	11.5	42.5 ± 11.0
Carl's Jam	Fore reef	12 13.100	68 21.170	Feb. 5 1999	11.5	8	16.2	52.0 ± 14.0
Habitat	Fore reef	12 12.030	68 18.500	Feb. 2 1999	10	4	4.2	56.0 ± 47.5
Karpata	Fore reef	12 09.950	68 19.600	Feb. 4 1999	9	9	14	44.0 ± 8.5
Twin Peaks	Fore reef	12 15.500	68 25.650	Feb. 4 1999	9	11	11.8	47.0 ± 12.5
W. Klein	Fore reef	12 09.900	68 19.800	Feb. 3 1999	12	8	9.3	41.5 ± 9.5

Table 8. Size and condition (mean ± standard deviation) of all stony corals (≥25 cm diameter), by site in Bonaire.

Site name	Stony corals		Partial colony surface mortality (%)			Stony corals (%)		
	#	Diameter (cm)	Recent	Old	Total	Standing dead	Bleached	Diseased
Barcardera	130	63.5 ± 39.5	0.4 ± 1.2	38.1 ± 31.0	38.4 ± 31.2	1.5	15.4	4.8
Carl's Jam	130	65.0 ± 37.5	0.6 ± 2.1	30.0 ± 28.5	30.6 ± 28.5	1	21.5	2.3
Habitat	17	43.5 ± 19.5	0.4 ± 0.9	37.7 ± 23.9	38.1 ± 24.0	0	58.8	0
Karpata	127	70.5 ± 45.0	0.9 ± 2.2	30.4 ± 24.6	31.3 ± 25.3	2.5	31	7.1
Twin Peaks	127	95.0 ± 64.5	1.0 ± 4.3	32.0 ± 27.4	33.1 ± 27.8	3.0	33.9	8.5
W. Klein	83	54.0 ± 26.0	0.4 ± 1.5	22.7 ± 23.5	23.0 ± 23.7	3.5	21.3	4

Table 9. Algal characteristics, and density of stony coral recruits and of *Diadema antillarum* (mean  $\pm$  standard deviation), by site in Bonaire.

Site name	Quadrats		Relative abundance (%)			Macroalgae		Coral recruits (#/0625m <sup>2</sup> )	<i>Diadema</i> (#/10m)
	(#)	Macroalgae	Turf algae	Crustose coralline algae	Height (cm)	Index <sup>1</sup>			
Barcadera	51	15.0 $\pm$ 21.0	65.5 $\pm$ 29.0	19.0 $\pm$ 23.5	0.6 $\pm$ 0.2	8.5 $\pm$ 6.0	0.2 $\pm$ 0.2	0	
Carl's Jam	37	18.5 $\pm$ 17.0	54.0 $\pm$ 22.5	27.5 $\pm$ 14.5	1.1 $\pm$ 1.1	13.5 $\pm$ 5.5	0.2 $\pm$ 0.6	0	
Habitat	12	1.0 $\pm$ 3.0	76.0 $\pm$ 16.0	23.0 $\pm$ 14.0	ND*	ND*	0.3 $\pm$ 0.6	0	
Karpata	42	15.0 $\pm$ 21.0	61.0 $\pm$ 24.0	24.0 $\pm$ 18.5	0.7 $\pm$ 0.3	11 $\pm$ 11.5	0.1 $\pm$ 0.2	0	
Twin Peaks	49	12.5 $\pm$ 20.0	43.5 $\pm$ 20.5	43.5 $\pm$ 20.5	0.8 $\pm$ 0.3	10 $\pm$ 11	0.1 $\pm$ 0.2	0	
W. Klein	15	15.5 $\pm$ 13.0	64.5 $\pm$ 20.0	20.0 $\pm$ 11.5	0.9 $\pm$ 0.2	14 $\pm$ 13	0.3 $\pm$ 0.3	0	

<sup>1</sup> Macroalgae Index = relative percent abundance x height

Table 10. Density (mean  $\pm$  standard deviation) of AGRRA fishes by site in Bonaire.

Site name	AGRRA Fishes			Herbivores (#/100m <sup>2</sup> )			Carnivores (#/100m <sup>2</sup> )		
	30 m transects (#)	Total density (#/100m <sup>2</sup> )	Species (#)	Acanthuridae	Scaridae	Haemulidae	Lutjanidae	Serranidae	
Barcadera	12	43.8 $\pm$ 12.5	27	1.8 $\pm$ 2.4	27.0 $\pm$ 8.0	2.0 $\pm$ 2.8	3.7 $\pm$ 3.5	3.2 $\pm$ 2.5	
Carl's Jam	12	18.8 $\pm$ 12.8	20	1.5 $\pm$ 1.8	10.8 $\pm$ 8.1	0.2 $\pm$ 0.5	2.0 $\pm$ 3.5	1.2 $\pm$ 1.8	
Karpata	10	55.6 $\pm$ 70	23	22.0 $\pm$ 63.7	21.0 $\pm$ 7.9	2.5 $\pm$ 2.1	4.2 $\pm$ 3.8	1.0 $\pm$ 1.2	
Twin Peaks	9	42.5 $\pm$ 9.0	21	9.1 $\pm$ 2.5	19.1 $\pm$ 3.6	0.2 $\pm$ 0.6	0.6 $\pm$ 1.2	0.7 $\pm$ 0.9	

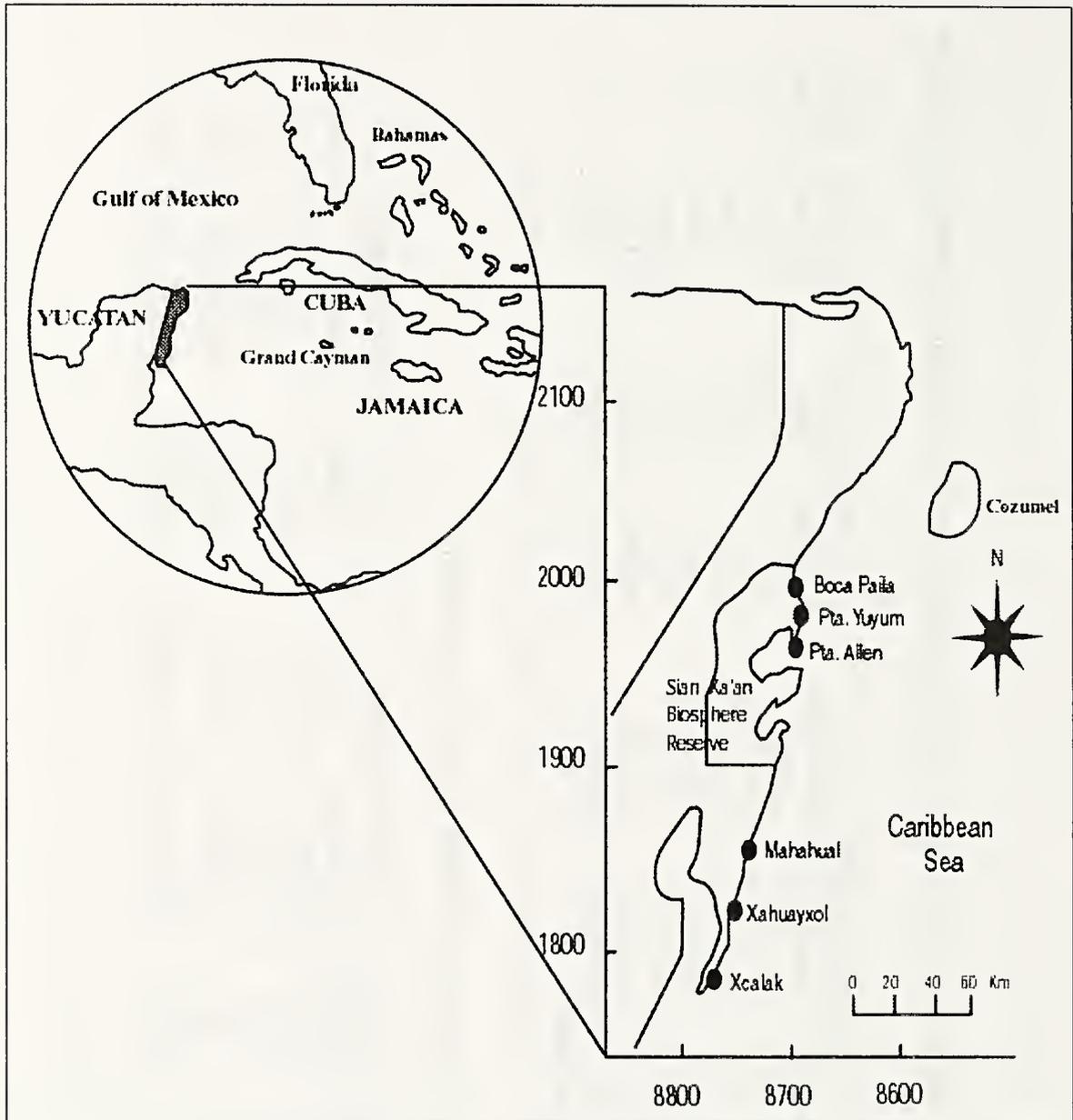


Figure 1. AGRRA survey sites (boldface) in central-southern Quintana Roo, México.

# CONDITION OF CORAL REEF ECOSYSTEMS IN CENTRAL-SOUTHERN QUINTANA ROO (PART 3: JUVENILE REEF FISHES)

BY

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## ABSTRACT

The spatial patterns of coral reef fish recruits were assessed using a visual census method at three scales (subreefs, reefs, and areas) between June and August, 1999 in the spur-and-groove habitat of six fore reefs in the Mexican Caribbean. Six thousand three hundred twenty-seven fish recruits belonging to 54 species in 30 genera and 18 families were counted. Slight differences were found in composition and density of all species at all three spatial scales. A multiple regression analysis indicated statistically significant relationships with recruit density that were positive for mean diameter and negative for the live/dead ratio of “large” ( $\geq 25$  cm diameter) stony corals. Recruit density may depend largely on the intrinsic behavior of each species in direct relation to food and refuge availability, rather than on live coral coverage.

## INTRODUCTION

The fringing coral reefs of the Mexican Caribbean form the northernmost part of the Mesoamerican Barrier Reef. They extend 350 km along the eastern coast of Quintana Roo state from the northeasterly corner of the Yucatán Peninsula to its southern frontier with Belize (Fig. 1). The Sian Ka'an Biosphere Reserve is located in the central-eastern portion of Quintana Roo from just south of Tulum ( $20^{\circ} 06' N$ , the northern boundary of Sian Ka'an), to Punta Punticub ( $19^{\circ} 05' N$ ). The reserve consists of 528,147 hectares with limited access and includes Ascensión and Espíritu Santo Bays. About 120,000 hectares are coastal and marine environments encompassing the coral reef zone (Gutiérrez-Carbonel et al., 1993).

Six reefs were sampled as part of the present study, three within the Sian Ka'an Biosphere Reserve and three outside the Reserve in the southern portion of the Mexican Caribbean (Fig. 1). Two of the reserve's reefs are remote from population centers (Boca

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Paila, Punta Yuyum), which has allowed their natural conditions to be largely conserved. The third, Punta Allen, is situated near the settlement of Rojo Gómez where the principal activity is a small-scale lobster fishery. Two of the reefs outside of the reserve (Mahahual, Xcalak) are located near human settlements where small-scale fishing is the predominant activity although most of the fleet's capture is focused on the open sea or at Chinchorro Bank rather than in the coastal reef zone. The reef at Xahuayxol has been relatively undisturbed anthropogenically. The principal fishing methods used are fixed traps, trotlines, gill nets and harpoons and these are mostly directed toward capture of commercially important species such as grouper (serranids), barracuda (sphyranids), snapper (lutjanids) and grunts (haemulids).

Relatively little research has been done on the coral reefs in the Mexican Caribbean and even less on its reef fishes. Earlier studies, predominantly focused on adult fish community structure, have been summarized by Arias-González et al. (1997) and Salazar et al. (1997). Recently Castro (1998) has characterized the fish community structure of Mahahual Reef. Núñez-Lara (1998) and Núñez-Lara and Arias-González (1998) have demonstrated the importance of environmental factors, particularly topographic complexity, in determining reef fish community structure. Arias-González (1998) has created trophic models for a protected and an unprotected zone, finding important differences reflected in fish biomass. Díaz-Ruiz et al. (1999) presented evidence that variations in species diversity and trophic structure are associated with sequential habitat use during the life cycle.

Recruitment is widely considered a key structuring process in reef fish communities (Doherty and Williams, 1988; Doherty and Fowler, 1994). Previous studies in the Western Atlantic have focused on the factors that regulate temporal and spatial variability in recruitment such as predation, refuge availability, and habitat use (Shulman, 1985a, 1985b; Shultz and Cowen, 1994; Booth and Beretta, 1994; Caselle and Warner, 1996). Enhanced understanding of natural fluctuations in recruitment and its relationship to coral reef conditions would allow more effective management of fishery resources. This study is the first assessment of coral reef fish recruitment patterns at different spatial scales in the Mexican Caribbean. Relationships between recruitment and descriptors of benthic reef condition are also examined.

## METHODS

Reef fish recruitment patterns were visually assessed between June 25 and August 31, 1999 on three spatial scales: area, reef and subreef. The central and southern areas of Quintana Roo have coastlines that are approximately 100 km long. Each of these areas was represented by three, strategically chosen reefs (see Ruiz et al., this volume), which were separated from one another by about 25-30 km. Every reef was partitioned into three representative subreefs at approximately 1 km intervals. Ten belt transects, each 30 m long by 1 m wide, were swum parallel to the coast at a depth of 12 m in the spur-and-groove habitat of the fore-reef zone by one diver (González-Salas) in every subreef. The spacing between adjacent transects was about 50 m. To assess the diurnal community of juvenile reef fishes, all surveys were made between 09:00 and 14:00 hours. Recruits were identified to species, or to the lowest possible taxonomic level. Their size was estimated

with a T-bar. Fish identifications were based on the descriptions of Randall (1983), Lindeman (1986), Humman (1994), and Lieske and Myers (1995).

Recruitment patterns were analyzed on the three spatial scales. Multiple regression analysis was used to estimate the relationship between juvenile density and benthic variables [total live stony coral cover; total, old, and recent partial-colony mortality, live/dead ratio, mean diameter, and percent bleached or diseased colonies for “large” ( $\geq 25$  cm diameter) stony corals; relative abundance of macroalgae and turf], which were measured in the same sites by Ruiz et al. (this volume). To evaluate the affinity of the sampling stations, a multivariate classification analysis based on the density of the fish species was made using the Bray-Curtis distance index (Bray and Curtis, 1957), complemented by the Unweighted Pair Grouping Method Average (UPGMA) cluster method. The Atlantic and Gulf Rapid Reef Assessment (AGRRA) fishes constitute a subset of the “all species” data: in this paper, “serranids” are species of *Epinephelus* (excluding *E. cruentatus* and *E. fulvus*, here considered to be *Cephalopholis cruentata* and *C. fulva*, respectively) and *Mycteroperca*.

## RESULTS

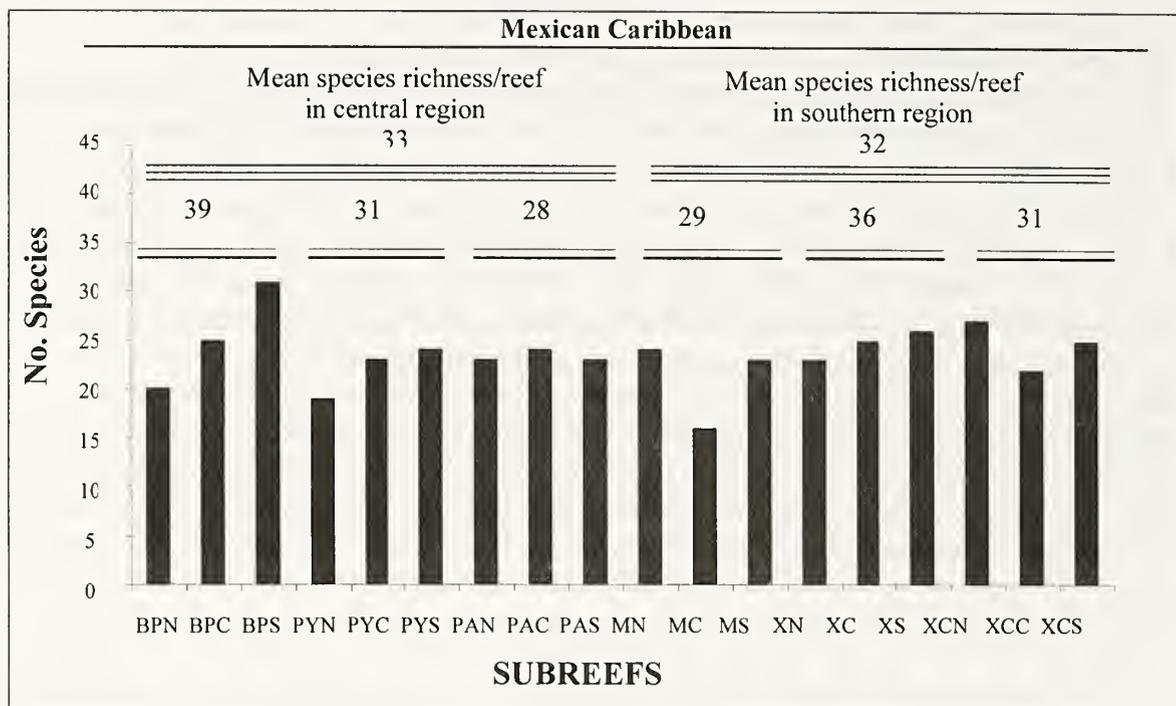
A total of 54 species of reef fish recruits belonging to 30 different genera and 18 families were identified in the surveys. No difference was noted in species richness at the largest spatial scale (Fig. 2), but the central area had reefs with both the highest (Boca Paila) and lowest (Punta Allen) numbers of species (Table 1). A total of 6,327 reef fish recruits were counted: 3,337 in the central area and 2,990 in the southern area. Hence, the “all species” recruit density was somewhat higher in the central area (Fig. 3). The highest densities, found in Boca Paila (central area) and Xcalak (southern area), were about twice that of the lowest in Xahuayxol (southern area). The mean “all species” recruit density overall was 117 individuals/100m<sup>2</sup>.

The 25 most frequently sighted species (Table 2) were a mixture of herbivores (*Stegastes*, *Sparisoma*, *Acanthurus*, *Scarus*) and carnivores that feed primarily on the benthos (e.g., *Halichoeres*, many other genera).

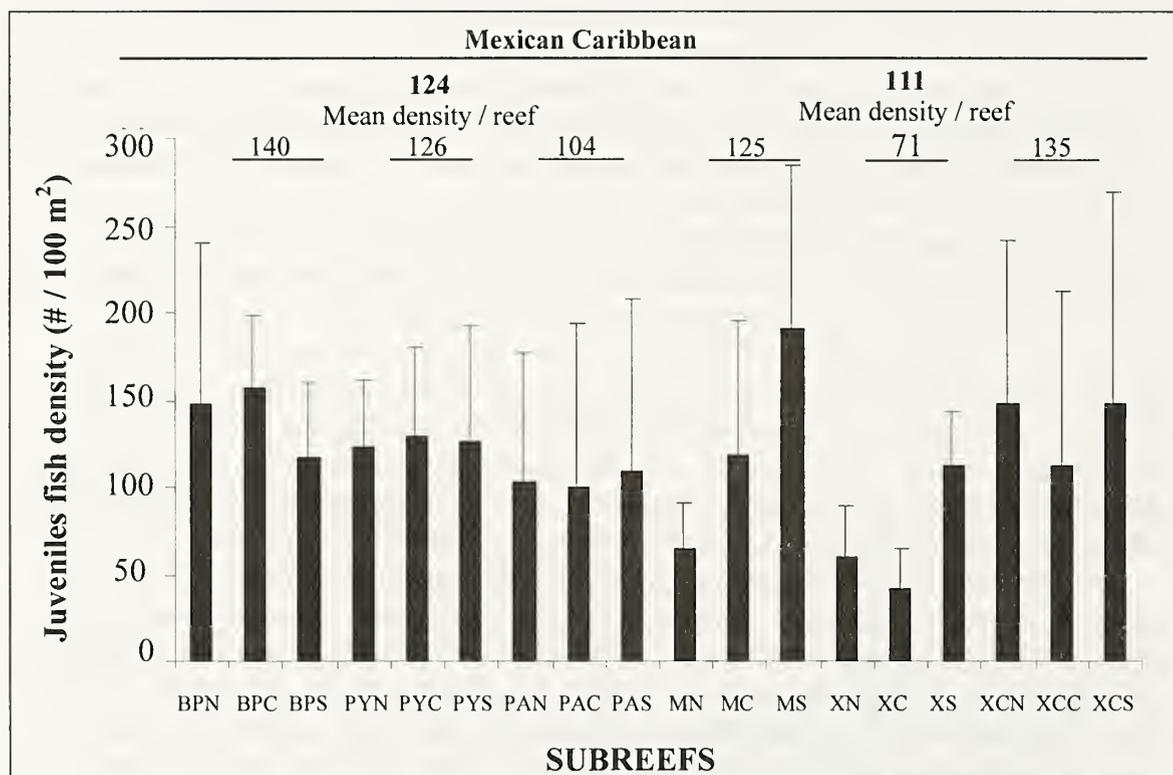
Overall, recruits of the AGRRA herbivores (scarids, acanthurids) were more abundant than the AGRRA carnivores (lutjanids>haemulids>serranids as defined above) (Table 3). Mean densities of parrotfish (scarid) recruits were highest in the two most northerly reefs of the central area (Boca Paila, Punta Yuyum) and in Mahahual, the most northerly reef of the southern area. Surgeonfish (acanthurid) recruits were most common in the most southerly reef (Xcalak) and least abundant in Mahahual.

The multiple regression analysis indicated that the mean diameter and live/dead ratio of the  $\geq 25$  cm in diameter stony corals together explained more than 60% of the variability in recruit density. The relationship with mean coral diameter was positive ( $r^2 = 0.276$ ,  $p = 0.02$ ), whereas that with the live/dead ratio was negative ( $r^2 = 0.379$ ,  $p < 0.05$ ) (Fig. 4). The remaining variables had no significant relationship with recruit density.

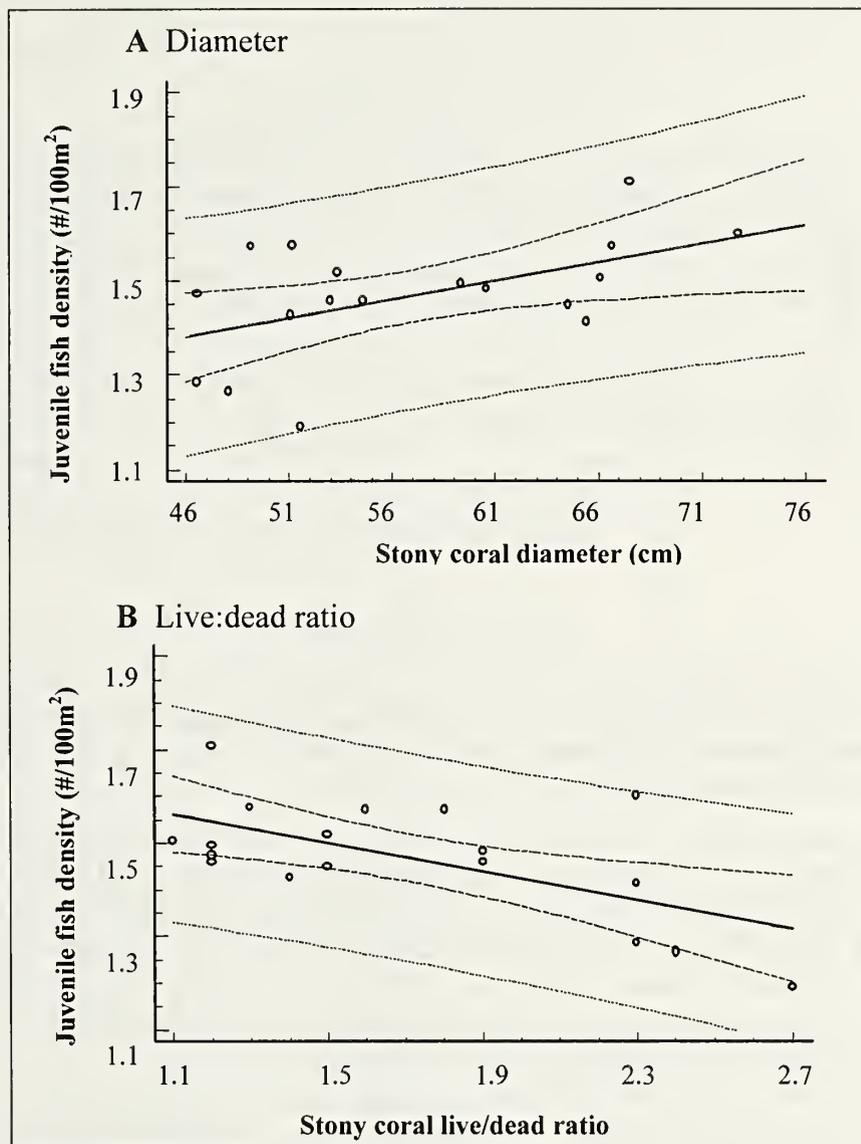
A numerical classification analysis in Q mode identified five groups of fish recruits (Fig. 5). The first includes four subreefs from the southern area and one from the central area. The second and fifth groups are restricted to the central area whereas the third and fourth are located in the southern area.



**Figure 2.** Species richness of all species of juvenile coral reef fishes, by subreef in central-southern Quintana Roo, México. See Table 1 for site codes.



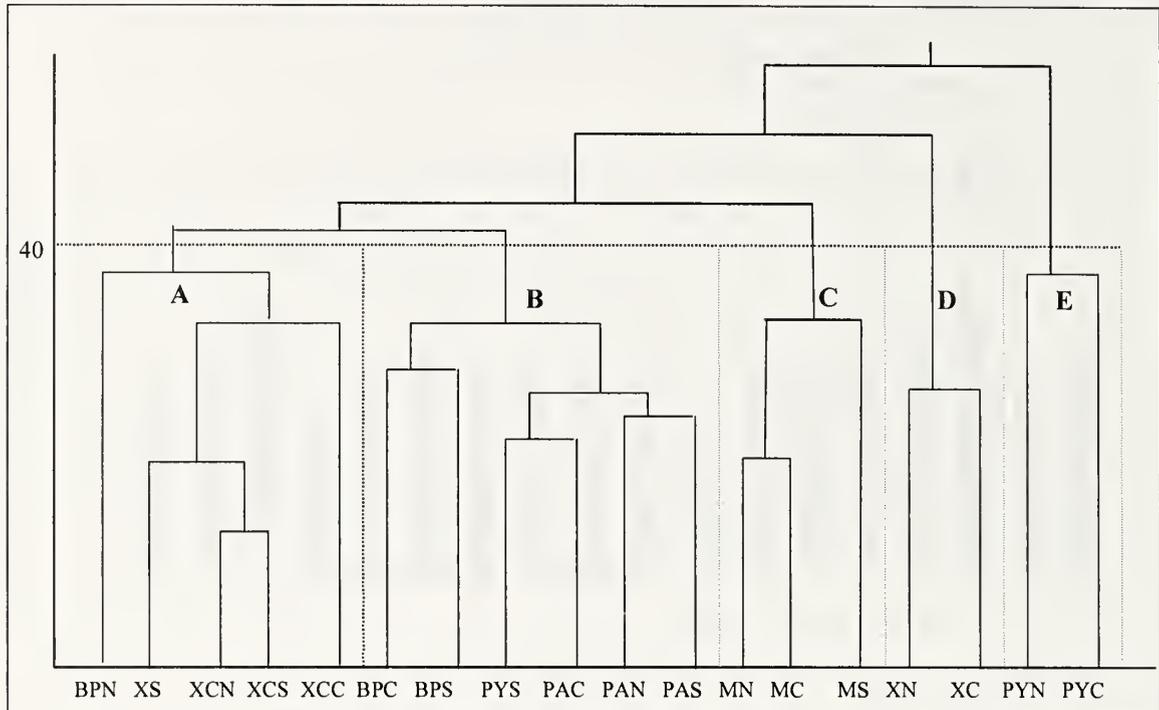
**Figure 3.** Mean density (no. individuals/100 m<sup>2</sup>) of all species of juvenile coral reef fishes by subreef in central-southern Quintana Roo, México. See Table 1 for site codes.



**Figure 4.** Regression plot between mean juvenile reef fish density (no. individuals/100 m<sup>2</sup>) and (A) mean stony coral diameter ( $y=1.041+0.00795374x$ ,  $r^2=0.276$ ,  $f(1,16)=6.13$ ,  $p=.002$ ) and (B) mean stony coral live/dead ratio ( $y=1.73079-0.15325x$ ,  $r^2=0.379$ ,  $f(1,16)=9.79$ ,  $p=.006$ ), by subreef in central-southern Quintana Roo, México.

## DISCUSSION

The similarity in the species richness of recruits in the central and southern areas is probably attributable in part to the great similarity in their reef structures although large-scale physical processes (such as current patterns) should also be taken into account. Much of the subreef- and reef-scale variability in reef fish recruitment could be related to local, coastal environmental factors. For example, the relatively high values found off Boca Baila (northern area) and off Xahuayxol and Xcalak (southern area) were all in reefs that are close to lagoons or bays that may serve as sources of recruits.



**Figure 5.** Hierarchical classification analysis of the juvenile fish transect data by subreef in central-southern Quintana Roo, México. See Table 1 for site codes. Labels A-E represent groups of sites with similar juvenile fish community structure.

Whereas adult herbivores (parrotfishes and surgeonfishes) were relatively abundant in the more heavily fished reefs where their natural predators are less common (Nuñez-Lara et al., this volume), the mean density of their recruits showed relatively little between-reef variation (Table 3). The relative paucity of juvenile snappers and grunts compared to adult densities found by Nuñez-Lara et al. (this volume) is probably related to the strong association of the juveniles with lagoonal systems, particularly in Boca Paila, Punta Yuyum and Punta Allen.

Recruitment variation at any of the three scales is likely to be influenced by one or more of the following factors: differential larval availability; differential settlement during habitat selection; mortality differences in early, post-settlement stages; or post-recruitment movement towards different preferred habitats (Caselle, 1996). The positive relationship between recruit density and stony coral diameter suggests that the survival of reef fish recruits is enhanced by the greater structural complexity of large corals. Indeed, Nemeth (1998) has recently documented the positive effects of hole and crevice density on recruit abundance. The overall negative relationship with the live/dead coral ratio, as previously found in St. Croix, U.S. Virgin Islands (Booth and Beretta, 1994; Caselle and Warner, 1996) is an indication that recruit density is largely independent of the condition of these stony corals. In other words, the dominant factors affecting reef fish recruitment apparently are not associated with the benthos condition indicators recorded by Ruiz et al. (this volume) but rather with larval input and behavior, as well as the availability of refuges and food. If the scales at which recruitment is being carried out are smaller than those employed in this study, it will be necessary to measure other variables, such as refuge type, or the heterogeneity, size and density of refuges in reef substrata.

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Table 1. Site information for AGRRA juvenile fish surveys in central-southern Quintana Roo, México

Site name	Site code	Reef type	Latitude (°'N)	Longitude (°'W)	Survey Date	Depth (m)	≥25 cm stony corals (#/10 m) <sup>1</sup>	% live stony coral cover (mean ± sd) <sup>1</sup>	30 m fish transects (#)	Species in transects (#)	Total
<b>Central</b>											
Boca Paila North	BPN	Fringing	20 06 51	87 27 23	Aug. 30 99	12	6	15.5 ± 4.5	10	9	20
Boca Paila Center	BPC	Fringing	20 06 21	87 27 34	Aug. 31 99	12	6	19.0 ± 9.5	10	9	25
Boca Paila South	BPS	Fringing	20 05 51	87 27 47	Aug. 31 99	12	2	8.5 ± 2.5	10	11	31
Punta Yuyum North	PYN	Fringing	19 58 30	87 27 10	Aug. 27 99	12	4.5	10.5 ± 5.5	10	7	19
Punta Yuyum Center	PYC	Fringing	19 58 00	87 27 06	Aug. 26 99	12	4	12.5 ± 6.0	10	10	23
Punta Yuyum South	PYS	Fringing	19 57 30	87 26 52	Aug. 25 99	12	4.5	10.5 ± 6.0	10	8	24
Punta Allen North	PAN	Fringing	19 50 30	87 26 15	Aug. 20 99	12	4.5	11.5 ± 3.5	10	8	23
Punta Allen Center	PAC	Fringing	19 50 00	87 26 36	Aug. 21 99	12	4.5	14.5 ± 7.5	10	9	24
Punta Allen South	PAS	Fringing	19 49 30	87 26 52	Aug. 22 99	12	3.5	10.5 ± 4.5	10	7	23
<b>Southern</b>											
Mahahual North	MN	Fringing	18 43 24	87 41 56	June 27 99	12	5	17.0 ± 7.0	10	12	24
Mahahual Center	MC	Fringing	18 43 01	87 42 09	June 26 99	12	4.5	17.0 ± 6.0	10	6	16
Mahahual South	MS	Fringing	18 42 30	87 42 20	June 25 99	12	6	16.5 ± 5.5	10	8	23
Xahuayxol North	XN	Fringing	18 30 55	87 45 02	July 20 99	12	3.5	11.5 ± 3.0	10	10	23
Xahuayxol Center	XC	Fringing	18 30 25	87 45 13	July 21 99	12	2.5	12.0 ± 5.5	10	11	25
Xahuayxol South	XS	Fringing	18 29 55	87 45 22	July 22 99	12	3.5	11.5 ± 5.5	10	9	26
Xcalak North	XCN	Fringing	18 13 43	87 49 51	July 13 99	12	3.5	9.0 ± 4.5	10	11	27
Xcalak Center	XCC	Fringing	18 13 09	87 49 54	July 14 99	12	3.5	9.5 ± 3.0	10	8	22
Xcalak South	XCS	Fringing	18 12 39	87 49 47	July 15 99	12	3	7.5 ± 4.5	10	10	25

<sup>1</sup>From Ruiz et al. (this volume)<sup>2</sup>Excluding any *Epinephelus cruentatus* and *E. fubius*.

Table 2. Sighting frequency and mean density of the 25 most frequently sighted juvenile fish species in the "all species" belt transect surveys in central-southern Quintana Roo, México. \* = AGRRA species.

Species name	Sighting frequency (%) <sup>1</sup>	Density (#/100m <sup>2</sup> )
<i>Halichoeres garnoti</i>	83	13.20
<i>Stegastes partitus</i>	76	13.37
* <i>Sparisoma aurofrenatum</i>	64	5.28
<i>Thalassoma bifasciatum</i>	57	13.54
<i>Chromis cyanea</i>	52	26.07
* <i>Acanthurus coeruleus</i>	51	3.04
* <i>Acanthurus bahianus</i>	48	3.31
* <i>Sparisoma viride</i>	35	1.69
* <i>Cephalopholis fulva</i> (= <i>Epinephelus fulvus</i> ) <sup>2</sup>	32	1.44
* <i>Holacanthus tricolor</i>	32	1.33
<i>Cantigaster rostrata</i>	29	1.30
<i>Stegastes dorsopunicans</i>	28	1.50
* <i>Scarus iserti</i> (= <i>S. croicensis</i> ) <sup>2</sup>	28	3.09
<i>Stegastes planifrons</i>	28	1.93
* <i>Bodianus rufus</i>	22	1.00
* <i>Cephalopholis cruentata</i> (= <i>Epinephelus cruentatus</i> ) <sup>2</sup>	22	0.96
<i>Clepticus parrae</i>	19	17.50
<i>Halichoeres maculipinna</i>	14	0.57
* <i>Chaetodon capistratus</i>	12	0.63
<i>Stegastes variabilis</i>	11	0.43
<i>Pseudopeneus maculatus</i>	11	0.39
<i>Gramma loreto</i>	11	0.72
* <i>Sparisoma radians</i>	10	0.76
* <i>Sparisoma atomarium</i>	10	0.48
* <i>Lutjanus apodus</i>	9	1.00

<sup>1</sup>Sighting frequency (%) = percentage of transects in which the species was recorded.

<sup>2</sup>Genus and/or species names according to Eschmeyer's (1998) revision.

Table 3. Density (mean  $\pm$  standard deviation) of juvenile AGRRA fishes by site in central-southern Quintana Roo, México.

Site name	Herbivores (#/100m <sup>2</sup> )		Carnivores (#/100m <sup>2</sup> )			Macroalgal Index <sup>2</sup>
	Acanthuridae	Scaridae	Haemulidae	Lutjanidae	Serranidae <sup>1</sup>	
<b>Central</b>						
Boca Paila N	2.7 $\pm$ 3.2	4.1 $\pm$ 5.0		0.2 $\pm$ 4.0		90
Boca Paila C	3.6 $\pm$ 3.0	3.5 $\pm$ 3.0				130
Boca Paila S	1.8 $\pm$ 1.0	2.6 $\pm$ 3.0				113
Punta Yuyum N	2.7 $\pm$ 2.0	4.6 $\pm$ 4.0		0.003		96
Punta Yuyum C	1.6 $\pm$ 4.0	5.8 $\pm$ 3.0		7.3 $\pm$ 0.7		81
Punta Yuyum S	2.5 $\pm$ 0.7	2.5 $\pm$ 3.0				81
Punta Allen N	2.5 $\pm$ 0.7	2.9 $\pm$ 2.0				142
Punta Allen C	4.0 $\pm$ 2.0	3.5 $\pm$ 1.0			0.003	116
Punta Allen S	3.0 $\pm$ 0.9	2.7 $\pm$ 2.0				126
<b>Southern</b>						
Mahahual N	0.3	1.4 $\pm$ 0.7		0.007		64
Mahahual C	0.3	4.6 $\pm$ 3.0				108
Mahahual S	1.3	5.9 $\pm$ 8.0				74
Xahuayxol N	3.6 $\pm$ 0.9	1.5 $\pm$ 1.0	0.003	0.003		75
Xahuayxol C	3.7 $\pm$ 3.0	1.4 $\pm$ 1.0	0.003	0.003		83
Xahuayxol S	1.6 $\pm$ 8.0	1.0 $\pm$ 0.4		0.007		71
Xcalak N	4.7 $\pm$ 3.0	1.9 $\pm$ 2.0		0.007		52
Xcalak C	4.6 $\pm$ 0.9	3.0 $\pm$ 3.0				49
Xcalak S	4.2 $\pm$ 2.0	2.6 $\pm$ 3.0	0.013	0.017		53

<sup>1</sup> *Epinephelus* spp. (excluding any *E. cruentatus* and *E. fulvus*) and *Mycteroperca* spp.<sup>2</sup> From Ruiz et al. (this volume)

## APPENDIX ONE

### THE ATLANTIC AND GULF RAPID REEF ASSESSMENT (AGRRA) PROTOCOLS: FORMER VERSION 2.2

BY

PATRICIA RICHARDS KRAMER<sup>1</sup> and JUDITH C. LANG<sup>2</sup>

#### INTRODUCTION

The AGRRA methodology is the result of an on-going international collaboration of reef scientists and managers. Since its initiation in 1995, the methods have undergone a series of iterations (Table 1) in response to recommendations from the organizing committee and our colleagues. Version 2.2 of the AGRRA method is briefly summarized below as it was the version used for many of the assessments reported in this volume. Sections that have been changed in subsequent versions are underlined. Specific deviations from this or other versions of the protocols are detailed in the individual papers of this volume. For more information on *the current version of the methodology*, data-sheet templates, survey equipment, and the AGRRA approach, see the AGRRA website (<http://coral.aoml.noaa.gov/agra/method/methodhome.htm>).

One of the main objectives of the AGRRA approach is to provide a standardized methodology enabling teams working in different areas to collect and compare data on a regional scale. With visual censuses, it is particularly important to minimize individual bias among observers. Hence, it is essential to carefully standardize methods prior to data collection. Suggestions for consistency training and calibration can also be found at the AGRRA website.

#### SELECTION OF REEFS AND SITES

For the purposes of AGRRA, a region is defined as the coarsest scale category (~100-1000 km scale); followed by an area (~10-100 km scale); a reef (~1-10 km scale); and a site (0.2 km scale).

The method for selection of reefs to assess will be influenced in part by their local abundance and distribution and by the sampling effort to be undertaken. Whenever the extent and/or number are too large for complete sampling, the reefs should be subdivided or “stratified” and examples selected from within each subdivision. All good sources of information (benthic maps, aerial photographs, remote images, charts, local knowledge, reconnaissance by Manta tow-board, etc.) should be employed to separate the reefs into

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subdivisions on the basis of geographic distribution and by secondary characteristics such as size, depth range, and position relative to land. If there are no clear bases for making subdivisions (e.g., a continuous bank barrier or fringing reef several kilometers long), a numbered grid 200 m long can be superimposed along a chart or image of the reef, with each number representing a potential survey site. Hardgrounds, pavements and other habitats that lack a framework constructed of reef-building corals should be avoided.

Once reefs are stratified, depending on the methods and resources available for use, the ones that are selected for survey will generally fall into one of three categories:

1. Unbiased—chosen on the basis of a random sampling strategy;
2. Representative—chosen with local knowledge to be representative of reefs in the area;
3. Strategic—chosen with local knowledge and for a purpose (e.g., they are considered threatened, degraded or in particularly good condition).

For regional comparisons, it is best to have reefs that are chosen by either category (1) or (2). Reefs chosen by (3) should be clearly flagged as such.

A site is defined as an area of habitat of 200 m x 200 m (or less) that is more or less homogeneous and accessible from a boat anchored or moored in one place. Two habitats of maximum reef development of particular interest are the ~1-5 m depth interval (shallow *Acropora palmata* zone) and the ~8-15 m depth interval (shallow fore reef or equivalent). Whenever possible, one site within each of these depth intervals should be surveyed on each reef chosen for an AGRRA assessment. The site description for each survey site should include an explanation of how the site was selected.

### **BENTHOS (STONY CORALS, ALGAE AND *DIADEMA*)**

1. At each site, haphazardly lay a 10-m transect line (marked in 2 m intervals) just above the reef surface (= taut) in a direction that is approximately parallel to the long axis of the reef. Be sure to avoid or cross any other transects. Stay away from the edges of the reef and try to avoid areas with abrupt changes in slope, deep grooves, large patches of sand or unconsolidated coral rubble. Unusual reef features should be included only to the extent appropriate to their relative abundance at the site.
2. Record total live stony coral (scleractinians + *Millepora*) cover under the transect line (to the nearest 10 cm) with a measuring tool (e.g., a 1-m long PVC pipe marked in 10 cm intervals). If the reefs are too small to avoid sandy patches, record how much of the line crosses sand (for later calculation of live stony coral cover/m of reef hard reef substratum).
3. Swim along the transect line, stopping at the first stony coral (Table 2) located directly beneath the line that is at least 25 cm (or, if preferred, at least 10 cm) long and in original growth position or, if fallen, has either become reattached to the substratum or is too large to move. Record each of the following:
  - A. Name (genus and species)
  - B. The water depth at the top of the colony at the beginning and end of each transect and at any major change in depth (>1m).

- C. After identifying the colony boundaries based on common skeletal connections among the living polyps and similarities in their size and color, record maximum colony projected diameter (live + dead areas) perpendicular to the axis of growth and maximum height (live + dead areas) above the substratum parallel to the axis of growth.
- D. Percentage estimates of the coral that are “recently dead” (corallite structures are white and either still intact or covered by a layer of algae or fine mud; include fish bites) and “old dead” (corallite structures are either gone or covered by organisms that are not easily removed, including brown encrusting clionid sponges) as viewed from above at an angle that is parallel to the axis of growth. If the entire coral has been dead a long time and can still be identified, at least to generic level based on gross (e.g., *Acropora palmata*) or skeletal (e.g., *Diploria*) morphology, score as 100% “old dead.”
- E. Evidence anywhere on the entire colony of—
- Any diseases, using the following color categories:
    - BB = Black band
    - WB = White band
    - WS = White spots, patches or pox
    - WP = White plague
    - YB = Yellow blotch (sometimes called yellow band)
    - RB = Red band
    - UK = Unknown

Underline any disease visible on the upper colony surface that contributed to the estimate of “% recent dead.”

- Bleached tissues as approximate severity of discoloration:
  - P = Pale (discoloration of coral tissue)
  - PB = Partly bleached (patches of fully bleached or white tissue)
  - BL = Bleached (tissue is totally white, no zooxanthellae visible)
- All other sources of recent mortality that can still be unambiguously identified, such as sediment, storm damage, parrotfish bites, damselfish bites and/or algal gardens, predation (e.g., by *Corallophilia abbreviata* or *Hermodice carunculata*), and spatial competitors (e.g., benthic algae, invertebrates like *Erythropodium caribaeorum* or other stony corals). Underline any that contributed to the estimate of “% recent dead.”
- Algal gardens established by damselfish (in live parts of the coral as numbers of resident fishes and/or the presence of their gardens).

For large clusters or thickets in which colony boundaries are not distinguishable, use a standard point-count method to identify recent death, old death, or living coral tissue every 25 cm along the line. The maximum diameter and height should be determined for the entire cluster or thicket.

4. Go to the next appropriately sized coral and repeat step 3 above. Continue evaluating each coral until reaching the other end of the transect line.

5. Reswim the transect with the 25 x 25 cm algal quadrat. Starting at 1 m, place the quadrat every two meters directly below the transect line (i.e., at 1,3,5,7,9 m). If any of these area are unsuitable (i.e., <80% is covered by algae of any functional group), place the quadrat on the nearest available space within a 1 m radius of the mark. If no suitable areas are present, draw a line through the space on the data sheet. For each quadrat that is in a suitable area, remove any thin layers of sediment that could cover crustose coralline algae and record each of the following:
  - A. Substratum type;
  - B. % of macroalgae (all larger erect fleshy algae >1 cm in height; both fleshy and calcareous);
  - C. % of algal turfs (mostly tiny filaments, <1 cm in height), including any below the canopies of macroalgae;
  - D. % living crustose coralline algae (solid, calcareous encrusters that are pink or reddish in color, include any that are clearly visible below turf algae or a thin layer of sediment);
  - E. By using the plastic ruler for scale, the average canopy height of the macroalgae in the quadrat.
  - F. Optional—the number of all small (up to 2 cm maximum diameter) stony corals (scleractinians and *Millepora*) in the quadrat and, whenever possible, their identity to the genus level.
6. Swim along the transect, counting every *Diadema antillarum* (juveniles and adults) that can be seen within a belt extending 1/2 m on each side of the line.
7. Collect the line and haphazardly reset it at least 1 m laterally away from its previous position, following previous guidelines. Try to distribute transects around the survey site.
8. Repeat steps 2-6 for each transect. Continue to reset transects until a minimum of 100 corals and 50 quadrats have been assessed.
9. Enter transect data into the AGRRA benthos spreadsheet. After checking for accuracy, submit an electronic copy to [agrra@rsmas.miami.edu](mailto:agrra@rsmas.miami.edu). Back up data regularly and store in a safe place.

## FISH

The following two distinct assessment methods should be applied at each site. Whenever possible, they should be conducted between 1000 and 1400 hours. As many fishes are wary of humans, it is important to try to keep away from other divers.

## Method I. Belt Transects for AGRRA Species.

All belt transects used for fish assessments should be located within the same general habitats and depth intervals (i.e., at 1-5 m and 8-15 m) as the benthic transects, but they will tend to be further apart and may range into deeper and shallower water.

1. Lay a 30 m transect by first placing the weighted end of the tape on the bottom and then swimming in a straight line (by periodically fixing on a distant object) while releasing the tape from the reel, which can be clipped to the weight belt for easy release.
2. Swim at a more or less constant rate holding a 1-m wide T-bar (with 5 cm increments for scale) angled downward at about 45 degrees while looking consistently about 2 m ahead of the T-bar. Within a belt visually estimated to be 2 m wide, count only the fish listed in Table 3 (also listed on the data sheet), while using the T-bar to estimate the size of each fish within one of the following size categories (<5, 5-10, 10-20, 20-30, 30-40, >40cm). If necessary, pause while recording data and then resume swimming. Do not count juvenile parrotfishes and grunts <5 cm in total length. Large, intraspecific groups can be classified into one-or-more size categories as necessary. By keeping an equivalent effort on all segments of the transect, the tendency to count all members of a school crossing the transect, instead of just those members which happen to be within the belt as counting of its segment takes place, can be avoided.
3. Recoil the tape and haphazardly redeploy at a distance of least 5 m laterally away from the previous position.
4. Repeat steps 1-3 for each transect. Continue to reset transects until a minimum of 10 transects have been completed.
5. Enter belt-transect data into the AGRRA fish spreadsheet. After checking for accuracy, submit an electronic copy to [agrra@rsmas.miami.edu](mailto:agrra@rsmas.miami.edu). Back up data regularly and store in a safe place.

Modifications. Researchers wanting to census other species of fish are encouraged to do so on a separate pass over the transect, after the completing counts of AGRRA fishes.

## Method II. Roving Diver Technique

After finishing the belt transects (or concurrently depending on the number of surveyors), a census of all species of fishes should be conducted in the same general depth and habitat as the belt transects, following the methodology of the Reef Environmental Education Foundation (REEF) (see <http://www.reef.org/>) and briefly explained below.

1. Swim around the site for approximately 30 minutes (preferably 45-60 minutes), searching under overhangs, in caves and so on to find as many fish species as possible.
2. By the end of the dive, estimate the density of each species by using logarithmic categories: Single (1 fish), Few (2-10 fishes), Many (11-100 fishes), or Abundant (>100 fishes) and summarize these observations on a standardized REEF data entry sheet.
3. Transcribe the data on a standardized REEF Scantron sheet and send to REEF HQ, at P.O. Box 246, Key Largo, FL 33037, USA.

### OPTIONAL COMPONENTS

#### Stationary Plot Fish Survey

The Stationary Plot Technique (Bohnsack and Bannerot, 1986), outlined below, has been used extensively in the Caribbean and the Florida Keys to provide data on abundance and size for a wide range of fish species. Its use is encouraged as a third way to quantify fishes at each site, but not as a replacement for Method I or II.

1. Count the number of fish observed in a visually estimated cylinder (radius of 7.5 m) for a period of five 5 minutes.
2. Estimate and record the length of each fish counted.

#### Herbivory

The Fish Bite Method (Steneck, 1985) can be used to gauge the effect of certain herbivorous fishes on algal composition in the survey sites. Herbivorous fish guilds are categorized as:

Scrapers = Scaridae (parrotfish)

Browsers = Acanthuridae (surgeonfish), *Microspathodon chrysurus* (yellowtail damselfish)

Non-denuders = other Pomacentridae (damselfish)

1. Use a metric scale in conjunction with natural landmarks on the reef surface (e.g., a small coral or gorgonian) to haphazardly delineate an area that is approximately 1 m square and representative of the benthic cover at the site. (Do not place a meter quadrat on the substratum as some fish are particularly prone to biting novel objects placed within their feeding territories).

Back off as far as possible while still being able to see the meter-square area. Watch for five minutes. Record the depth, time of day, and number of bites from all species of fishes in the three guilds listed above, whenever possible identifying them to

species. It is necessary to be able to distinguish (a) juvenile scarids from other fishes with similar stripes, such as acanthurids and labrids (wrasses which only look as though they are biting algae as they search for amphipods to eat) and (b) yellowtail damselfish (which are browsers) from the species of damselfish that cultivate algal gardens.

2. Repeat for a total of five quadrats (and ~25 minutes of observation).

### ACKNOWLEDGMENTS

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Table 1. Timeline of the development of the AGRRA methodology.

Version	Date	Major contributors	Highlights
Prototype 1	1995	Ginsburg, Bak, Kiene, Gischler and Kosmynin. (1996)	Large stony corals–mortality and size quantified in Florida Keys
Prototype 2	1996-1997	Steneck and Lang	Initial draft of AGRRA benthos and fish protocols prepared and field-tested in the Bahamas and Mexico
Prototype 3	Summer 1997	Kramer, Kosmynin, Marks, and Kramer	First extensive field test and consistency training in Florida Keys and Andros; Benthos protocol–damselfish and sources of coral mortality added; Fish protocol–Roving Diver Technique used in addition to belt transects
Version 1.0	Fall 1997	Steneck, Lang, Kramer & Ginsburg	First version posted at AGRRA web site
Version 2.0	June 1998	Ginsburg, Kramer, Lang, Sale and Steneck	Revised on basis of Miami June 1998 Workshop: Benthos protocol–estimates of live stony coral cover added, bleaching categories defined, placement of algal quadrats and substratum type standardized, macroalgal height changed from maximum to average; Fish protocol–belt transects and Roving Diver Technique methods adopted, belt transect length and number stabilized at 30 m and 10, respectively
Version 2.1	March 1999	Bonaire Training Workshop	Benthos protocol–minimum size for individual stony coral assessment informally changed from 25 cm to 10 cm, coral disease terminology standardized; Fish protocol–parrotfish and grunts <5 cm in total length removed; UW data sheets standardized; Excel data spreadsheets introduced
Version 2.2	May 1999	Akumal Training Workshop	Benthos protocol–informal clarification that thin layers of sediment only should be removed from substratum in algal quadrats; Fish protocol–species in “Other” category of belt transects standardized; First Spanish translation of workshop training manual

Table 1 Continued.

Version	Date	Major contributors	Highlights
Version 3.0	May 2000		<p>Revised on basis of Miami May 2000 Workshop:</p> <p>Benthos protocol—minimum number of transects set at 6, minimum size for individual stony corals changed from 25 cm to 10 cm and minimum number of colonies assessed changed from 100 to 50, minimum number of quadrats changed from 50 to 30, turf algae removed and macroalgae partitioned into fleshy and calcareous with separate measurements of their average heights, relative abundance of remaining algal functional groups redefined to be assessed without removal of canopy layers or sediment, coral recruit counts changed from optional to required, a measure of maximum reef relief added;</p> <p>Fish protocol—species of balistids and serranids in belt transects standardized</p>

Table 2. List of stony corals surveyed in the AGRRA belt transects during 1997-2000.

Suborder Family <sup>1</sup>	Species <sup>1</sup>
<b>Milleporina</b> Milleporidae	<i>Millepora alcicornis</i> <i>M. brazilienses</i> <sup>2</sup> <i>M. complanata</i>
<b>Scleractinia</b>	
Acroporidae	<i>Acropora cervicornis</i> <i>A. palmata</i> <i>A. prolifera</i>
Agaricidae	<i>Agaricia agaricites</i> <i>A. fragilis</i> <i>A. grahamae</i> <i>A. humilis</i> <i>A. lamarcki</i> <i>A. tenuifolia</i> <i>A. undata</i> <i>Leptoseris cucullata</i>
Astrocoeniidae	<i>Stephanocoenia intersepta</i>
Caryophyllidae	<i>Eusmilia fastigiata</i>
Faviidae	<i>Colpophyllia natans</i> (OR <i>C. amaranthus</i> , <i>C. breviserialis</i> and <i>C. natans</i> ) <i>Cladocora arbuscula</i> <i>Diploria clivosa</i> <i>D. labyrinthiformis</i> <i>D. strigosa</i> <i>Favia fragum</i> <i>F. leptophylla</i> <sup>2</sup> <i>Manicina areolata</i> <i>Montastraea annularis</i> (OR <i>Montastraea annularis f. annularis</i> ) <i>M. cavernosa</i> <i>M. faveolata</i> (OR <i>Montastraea annularis f. faveolata</i> ) <i>M. franksi</i> (OR <i>Montastraea annularis f. franksi</i> ) <i>Solenastrea bournoni</i> <i>S. hyades</i>
Meandrinidae	<i>Dendrogyra cylindrus</i> <i>Dichocoenia stokesi</i> <i>Meandrina meandrites</i>
Mussidae	<i>Isophyllastrea rigida</i> <i>Isophyllia sinuosa</i> <i>Mussa angulosa</i> <i>Mussismilia braziliensis</i> <sup>2</sup> <i>M. hartii</i> <sup>2</sup> <i>M. hispida</i> <sup>2</sup> <i>Mycetophyllia aliciae</i> <i>M. ferox</i> <i>M. lamarckiana</i> (OR <i>M. danaana</i> and <i>M. lamarckiana</i> )

Table 2 Continued.

Suborder Family <sup>1</sup>	Species <sup>1</sup>
	<i>Scolymia cubensis</i>
	<i>S. lacera</i>
Oculinidae	<i>Oculina</i> sp.
Pocilloporidae	<i>Madracis decactis</i>
	<i>M. formosa</i>
	<i>M. mirabilis</i>
	<i>M. pharensis</i>
Poritidae	<i>Porites astreoides</i>
	<i>P. branneri</i>
	<i>P. furcata</i> (OR <i>P. porites</i> f. <i>furcata</i> )
	<i>P. porites</i> (OR <i>P. porites</i> f. <i>porites</i> )
Siderastreidae	<i>Siderastrea radians</i>
	<i>S. siderea</i> (OR <i>Siderastrea radians</i> f. <i>siderea</i> )
	<i>S. stellata</i> <sup>2</sup>

<sup>1</sup> Family and species names as in Cairns et al. (1991), except for Foster's (1987) revision of *Stephanocoenia*, and Weil and Knowlton's (1994) revision of the *Montastraea annularis* species complex.

<sup>2</sup> in Brazil

Table 3. List of fishes surveyed in the AGRRA belt transects (Version 2.2).

Family names <sup>1</sup>		Species names <sup>1</sup>	
Scientific	Common	Scientific	Common
Acanthuridae	surgeonfish	<i>Acanthurus bahianus</i>	ocean surgeonfish
		<i>A. chirurgus</i>	doctorfish
		<i>A. coeruleus</i>	blue tang
Balistidae	leatherjackets	<i>Aluterus scriptus</i>	scrawled filefish
		<i>B. vetula</i>	queen triggerfish
		<i>Cantherhines macrocerus</i>	whitespotted filefish
		<i>C. pullus</i>	orangespotted filefish
		<i>C. sufflamen</i>	ocean triggerfish
		<i>Melichthys niger</i>	black durgon
		<i>Xanthichthys ringens</i>	Sargassum Triggerfish
Chaetodontidae	butterflyfish	<i>Chaetodon aculeatus</i>	longsnout butterflyfish
		<i>C. capistratus</i>	foureye butterflyfish
		<i>Chaetodon ocellatus</i>	spotfin butterflyfish
		<i>Chaetodon sedentarius</i>	reef butterflyfish
		<i>Chaetodon striatus</i>	banded butterflyfish
Haemulidae <sup>2</sup>	grunt	<i>Anisotremus surinamensis</i>	black margate
		<i>A. virginicus</i>	porkfish
		<i>Haemulon album</i>	margate
		<i>H. aurolineatum</i>	tomtate
		<i>H. carbonarium</i>	caesar grunt
		<i>H. chrysargyreum</i>	smallmouth grunt
		<i>H. flavolineatum</i>	French grunt
		<i>H. macrostomum</i>	Spanish grunt
		<i>H. melanurum</i>	cottonwick
		<i>H. parra</i>	sailors choice
		<i>H. plumieri</i>	white grunt
		<i>H. sciurus</i>	bluestriped grunt
		<i>H. striatum</i>	striped grunt
Lutjanidae	snapper	<i>Lutjanus analis</i>	mutton snapper
		<i>L. apodus</i>	schoolmaster
		<i>L. cyanopterus</i>	cupera snapper
		<i>L. griseus</i>	gray snapper
		<i>L. jocu</i>	dog snapper
		<i>L. mahogoni</i>	mahogany snapper
		<i>L. synagris</i>	lane snapper
		<i>Ocyurus chrysurus</i>	yellowtail snapper
Pomacanthidae	angelfish	<i>Centropyge argi</i>	cherubfish
		<i>Holacanthus bermudensis</i>	blue angelfish
		<i>H. ciliaris</i>	queen angelfish
		<i>H. tricolor</i>	rock beauty
		<i>Pomacanthus arcuatus</i>	gray angelfish
		<i>P. paru</i>	French angelfish

Table 3 Continued.

Family names <sup>1</sup>		Species names <sup>1</sup>			
Scientific	Common	Scientific	Common		
Scaridae <sup>2</sup>	parrotfish	<i>Scarus coelestinus</i>	midnight parrotfish		
		<i>S.coeruleus</i>	blue parrotfish		
		<i>S. croicensis</i>	striped parrotfish		
		<i>S. guacamaia</i>	rainbow parrotfish		
		<i>S. taeniopterus</i>	princess parrotfish		
		<i>S. trispinosus</i> <sup>3</sup>	greenlip parrotfish		
		<i>S. vetula</i>	queen parrotfish		
		<i>Sparisoma atomarium</i>	greenblotch parrotfish		
		<i>S. aurofrenatum</i>	redband parrotfish		
		<i>S.chrysopterus</i>	redtail parrotfish		
		<i>S. radians</i>	bucktooth parrotfish		
		<i>S. rubripinne</i>	redfin parrotfish		
		<i>S. viride</i>	stoplight parrotfish		
		Serranidae	sea basses (grouper)	<i>Epinephelus adscensionis</i>	rock hind
				<i>E. cruentatus</i>	graysby
<i>E. fulvus</i>	coney				
<i>E. guttatus</i>	red hind				
<i>E. marginatus</i> <sup>3</sup>	dusky grouper				
<i>E. striatus</i>	Nassau grouper				
<i>Mycteroperca bonaci</i>	black grouper				
<i>M interstitialis</i>	yellowmouth grouper				
<i>M. rubra</i>	comb grouper				
<i>M. tigris</i>	tiger grouper				
Others		<i>M. venenosa</i>	yellowfin grouper		
		<i>Bodianus rufus</i>	spanish hogfish		
		<i>Caranx ruber</i>	bar jack		
		<i>Lachnolaimus maximus</i>	hogfish		
		<i>Microspathodon chrysurus</i>	yellowtail damselfish		
		<i>Sphyraena barracuda</i>	great barracuda		

<sup>1</sup>Nomenclature as in Robbins et al. (1991); see Appendix Two for the corresponding generic and specific names in Eschmeyer et al. (1998).

<sup>2</sup>Excluding Haemulidae and Scaridae <5 cm in total length.

<sup>3</sup>in Brazil

## APPENDIX TWO

### FISH BIOMASS CONVERSION EQUATIONS

BY

KENNETH W. MARKS<sup>1</sup> and KRISTI D. KLOMP<sup>2</sup>

### INTRODUCTION

Fish abundance determined from the AGRRA belt transects has been reported throughout this volume as density estimates (number of individuals/100m<sup>2</sup>). Most investigators using the AGRRA protocol have utilized a consistent methodology for censusing fish so that abundance estimates reported as density are especially useful in making regional comparisons of fish abundance. In some instances, however, where individual species within the reported taxa have widely varying morphologies, biomass (weight) may be a more representative measure of fish abundance. Biomass is an important attribute of populations that may be of interest to ecologists and resource managers since it provides insight into the trophic structure of the community and the production capacity of a reef (Bohnsack and Harper, 1988; Anderson and Neumann, 1996). Some of the authors in this volume have chosen to present their fish abundance results as biomass (grams/100m<sup>2</sup>). Fish weight was estimated using previously established length-weight relationships for Caribbean fishes (Table 1).

### METHODS

Fish weight was calculated using the power function:  $W = aL^b$ , where  $W$  is the weight (grams),  $L$  is the length (cm), and  $a$  and  $b$  are parameters estimated by linear regression of logarithmically transformed length-weight data. The parameters  $a$  and  $b$  shown in Table 1 were adjusted for unit length from linear regressions performed on fish lengths (mm) reported previously in the literature. Most of the length-weight relationships were determined from southern Florida specimens (Bohnsack and Harper, 1988), with exceptions as noted from Bohnsack and Harper (1988), Bullock et al. (1992), Claro and Garcia-Arteaga (1994), and Letourneur et al. (1998).

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Table 1. Length-weight relationships for the AGRRA fishes.

Scientific name	Common name	a	b	Comments <sup>2</sup>
<i>Acanthurus bahianus</i>	Ocean Surgeonfish	0.0237	2.9752	
<i>Acanthurus chirurgus</i>	Doctofish	0.0040	3.5328	
<i>Acanthurus coeruleus</i>	Blue Tang	0.0415	2.8346	
<i>Aluterus scriptus</i>	Scrawled Filefish	0.8230	1.8136	
<i>Anisotremus surinamensis</i>	Black Margate	0.0059	3.3916	
<i>Anisotremus virginicus</i>	Porkfish	0.0148	3.1674	
<i>Balistes vetula</i>	Queen Triggerfish	0.0267	2.9903	
<i>Bodianus rufus</i>	Spanish Hogfish	0.0144	3.0532	
<i>Cantherhines macrocerus</i>	Whitespotted Filefish	0.0562	2.6534	
<i>Cantherhines pullus</i>	Orangespotted Filefish	0.0684	2.5632	
<i>Canthidermis sufflamen</i>	Ocean Triggerfish	0.0176	3.0554	
<i>Caranx ruber</i>	Bar Jack	0.0074	3.2370	
<i>Centropyge argi</i>	Cherubfish	0.0601	2.6920	* No data available for species; used <i>Centropyge tibicen</i> as model (4)
<i>Cephalopholis cruentata</i> <sup>1</sup>	Graysby	0.0135	3.0439	(was <i>Epinephelus cruentatus</i> )
<i>Cephalopholis fulva</i> <sup>1</sup>	Coney	0.0175	3.0000	(was <i>Epinephelus fulvus</i> ); * St. Croix data used: n=1644, a=0.0175, b=3.0 (1)
<i>Chaetodon aculeatus</i>	Longsnout Butterflyfish	0.0220	3.1897	* No data available for species; used <i>Chaetodon capistratus</i> as model
<i>Chaetodon capistratus</i>	Four-eye Butterflyfish	0.0220	3.1897	
<i>Chaetodon ocellatus</i>	Spotfin Butterflyfish	0.0318	2.9838	
<i>Chaetodon sedentarius</i>	Reef Butterflyfish	0.0252	3.0760	
<i>Chaetodon striatus</i>	Banded Butterflyfish	0.0222	3.1395	
<i>Epinephelus adscensionis</i>	Rock Hind	0.0111	3.1124	* No data available for species; used <i>Epinephelus guttatus</i> as model
<i>Epinephelus guttatus</i>	Red Hind	0.0111	3.1124	
<i>Epinephelus itajara</i>	Jewfish	0.0131	3.0560	* Gulf of Mexico data used: n=66, a=0.0131, b=3.056 (2)
<i>Epinephelus marginatus</i>	Dusky Grouper	0.0065	3.2292	* No data available for species; used <i>Epinephelus striatus</i> as model
<i>Epinephelus morio</i>	Red Grouper	0.0123	3.0350	
<i>Epinephelus striatus</i>	Nassau Grouper	0.0065	3.2292	
<i>Haemulon album</i>	White Margate	0.0167	3.0423	
<i>Haemulon aurolineatum</i>	Tomtate	0.0100	3.2077	
<i>Haemulon carbonarium</i>	Caesar Grunt	0.0147	3.0559	
<i>Haemulon chrysargyreum</i>	Smallmouth Grunt	0.3971	2.1567	
<i>Haemulon flavolineatum</i>	French Grunt	0.0127	3.1581	
<i>Haemulon macrostomum</i>	Spanish Grunt	0.0244	3.0295	
<i>Haemulon melanurum</i>	Cottonwick	0.0226	2.9527	
<i>Haemulon parra</i>	Sailors Choice	0.0199	2.9932	
<i>Haemulon plumieri</i>	White Grunt	0.0121	3.1612	
<i>Haemulon sciurus</i>	Bluestriped Grunt	0.0194	2.9996	
<i>Holacanthus bermudensis</i>	Striped Grunt	0.0175	3.0990	
<i>Holacanthus ciliaris</i>	Blue Angelfish	0.0319	2.8994	
<i>Holacanthus tricolor</i>	Queen Angelfish	0.0337	2.9004	
<i>Lachnolaimus maximus</i>	Rock Beauty	0.0428	2.8577	
<i>Lutjanus analis</i>	Hogfish	0.0203	2.9880	
<i>Lutjanus apodus</i>	Mutton Snapper	0.0162	3.0112	
	Schoolmaster	0.0194	2.9779	

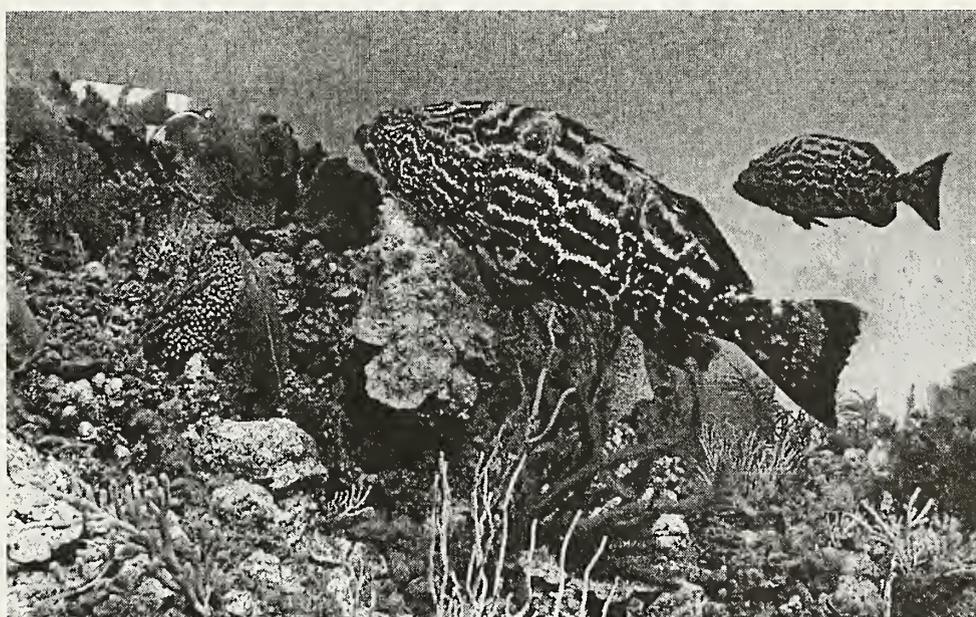
<i>Lutjanus cyanopterus</i>	Cubera Snapper	0.0151	3.0601	
<i>Lutjanus griseus</i>	Gray Snapper	0.0232	2.8809	
<i>Lutjanus jocu</i>	Dog Snapper	0.0308	2.8574	
<i>Lutjanus mahogoni</i>	Mahogany Snapper	0.0429	2.7190	
<i>Lutjanus synagris</i>	Lane Snapper	0.0295	2.8146	
<i>Melichthys niger</i>	Black Durgon	0.0562	2.6534	* No data available for species; used <i>Cantherhines macrocerus</i> as model
<i>Microspathodon chrysurus</i>	Yellowtail Damselfish	0.0239	3.0825	
<i>Mycteroperca acutirostris</i>	Comb Grouper	0.0068	3.2051	* No data available for species; used <i>Mycteroperca bonaci</i> as model
<i>Mycteroperca bonaci</i>	Black Grouper	0.0068	3.2051	
<i>Mycteroperca interstitialis</i>	Yellowmouth Grouper	0.0068	3.2051	* No data available for species; used <i>Mycteroperca bonaci</i> as model
<i>Mycteroperca microlepis</i>	Gag	0.0130	3.0305	
<i>Mycteroperca phenax</i>	Scamp	0.0068	3.2051	* No data available for species; used <i>Mycteroperca bonaci</i> as model
<i>Mycteroperca tigris</i>	Tiger Grouper	0.0094	3.1200	* Cuban data used: n=145, a=0.0094, b=3.12 (3)
<i>Mycteroperca venenosa</i>	Yellowfin Grouper	0.0095	3.1400	* St. Thomas/St. John data used: n=103, a=0.0069, b=3.14 (1)
<i>Ocyurus chrysurus</i>	Yellowtail Snapper	0.0405	2.7180	
<i>Pomacanthus arcuatus</i>	Gray Angelfish	0.0344	2.9680	
<i>Pomacanthus paru</i>	French Angelfish	0.0203	3.1264	
<i>Scarus coelestinus</i>	Midnight Parrotfish	0.0153	3.0618	
<i>Scarus coeruleus</i>	Blue Parrotfish	0.0124	3.1109	
<i>Scarus guacamala</i>	Rainbow Parrotfish	0.0155	3.0626	
<i>Scarus inserti</i> <sup>1</sup>	Striped Parrotfish	0.0147	3.0548	(was <i>S. croicensis</i> )
<i>Scarus sp.</i>	Unidentified Scarus	0.0250	2.9214	* Used <i>Sparisoma viride</i> as model
<i>Scarus taeniopterus</i>	Princess Parrotfish	0.0335	2.7086	
<i>Scarus trispinosus</i>	Greenlip Parrotfish	0.0153	3.0618	* No data available for species; used <i>Scarus coelestinus</i> as model
<i>Scarus vetula</i>	Queen Parrotfish	0.0250	2.9214	* No data available for species; used <i>Sparisoma viride</i> as model
<i>Sparisoma atomarium</i>	Greenblotch Parrotfish	0.0121	3.0275	
<i>Sparisoma aurofrenatum</i>	Redband Parrotfish	0.0046	3.4291	
<i>Sparisoma chrysopterum</i>	Redtail Parrotfish	0.0099	3.1708	
<i>Sparisoma radians</i>	Bucktooth Parrotfish	0.0121	3.0275	* No data available for species; used <i>Sparisoma atomarium</i> as model
<i>Sparisoma rubripinne</i>	Redfin Parrotfish	0.0156	3.0641	
<i>Sparisoma spp.</i>	Unidentified Sparisoma	0.0250	2.9214	* Used <i>Sparisoma viride</i> as model
<i>Sparisoma viride</i>	Stoptight Parrotfish	0.0250	2.9214	
<i>Sphyræna barracuda</i>	Great Barracuda	0.0050	3.0825	
<i>Xanithichthys ringens</i>	Sargassum Triggerfish	0.0267	2.9903	* No data available for species; used <i>Balistes vetula</i> as model

<sup>1</sup>Genus and species names according to the classification in Eschmeyer et al. (1998).

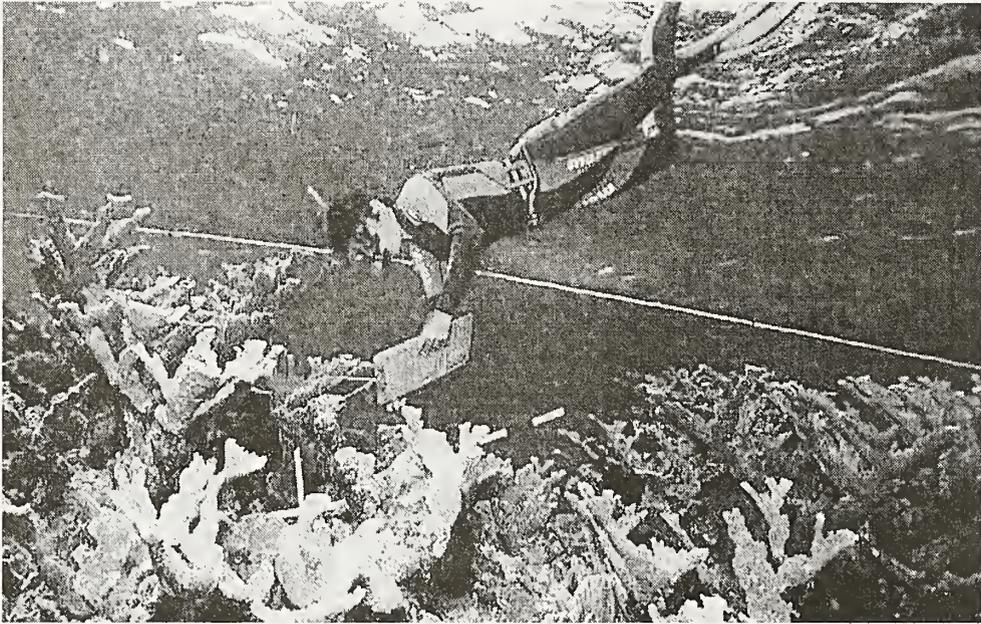
<sup>2</sup>(1) = Bohnsack and Harper (1988); (2) = Bullock et al. (1992); (3) = Claro and Garcia-Arteaga (1994); (4) = Letourneur et al. (1998)



**Plate 13A.** Reef fishes play many important roles in coral reef community dynamics by their trophic interactions as herbivores and as predators. Schooling acanthurids, as shown here, overwhelm damselfish to raid their gardens. In the AGRRA belt transect method, sampling biases are minimized by restricting the width of the belt transect and the number of species recorded which help to maintain a relatively consistent search image. (Photo Kenneth W. Marks)



**Plate 13B.** Belt transects are used to estimate the abundance and size (used for biomass estimations) of ecologically and/or commercially significant fishes, such as these black groupers (*Mycteroperca bonaci*) which have been overharvested in much of the wider Caribbean. REEF's Roving diver surveys are used to assess fish species richness and relative abundance and complement the belt transect method. (Photo Robert W. Steneck)



**Plate 14A.** One of the main objectives of the AGRRA approach is to provide a standardized methodology enabling teams working in different areas to collect and compare data on a regional scale. The transect-based benthos protocol is focused on several indicators of the condition of stony corals and the abundance of reef algae and *Diadema*. (Photo Kenneth W. Marks)



**Plate 14B.** Two distinct methods, belt transects as shown here, and REEF roving diving surveys, provide complementary “snapshots” of fishes at AGRRA assessment sites. (Photo Kenneth W. Marks)













**STATUS OF CORAL REEFS IN THE WESTERN ATLANTIC:  
RESULTS OF INITIAL SURVEYS,  
ATLANTIC AND GULF RAPID REEF ASSESSMENT (AGRR) PROGRAM**

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