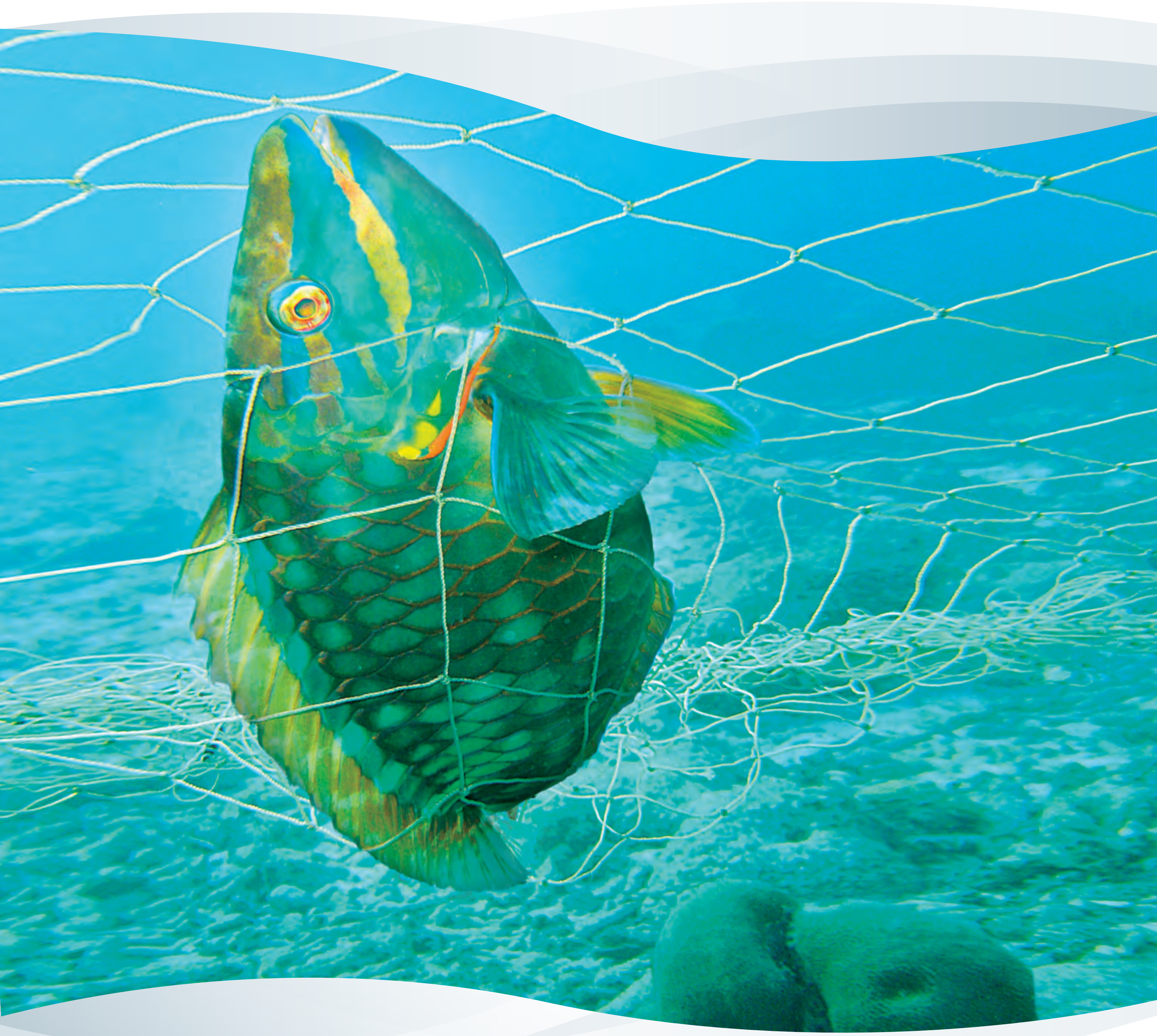
GCRMN
GLOBAL CORAL REEF MONITORING NETWORK



Status and Trends of Caribbean Coral Reefs: 1970-2012

EDITED BY
JEREMY JACKSON · MARY DONOVAN · KATIE CRAMER · VIVIAN LAM

STATUS AND TRENDS OF CARIBBEAN CORAL REEFS: 1970-2012

EDITED BY JEREMY JACKSON, MARY DONOVAN, KATIE CRAMER, AND VIVIAN LAM

Dedication: This book is dedicated to the many people who have worked on coral reefs to understand them, to protect them, and to appreciate their beauty and meaning for humanity and the natural world. We also recognize the International Coral Reef Initiative and partners, and particularly the people of all nations throughout the wider Caribbean region who continue to strive for the existence of healthy Caribbean reefs for future generations.

Note: The conclusions and recommendations of this volume are solely the opinions of the authors and contributors and do not constitute a statement of policy, decision, or position on behalf of the participating organizations.

Front Cover: Dead parrotfish (*Sparisoma viride*) caught in gillnet in front of a completely destroyed reef (Photo by Ayana Elizabeth Johnson)

Back Cover: School of the stoplight parrotfish *Sparisoma viride* on the south shore of Bermuda. (Photo by Philipp Rouja)

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Global Coral Reef Monitoring Network

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FOREWORD

The Caribbean is a sprawling sea of deep nutrient-poor waters punctuated by great oases of biomass production and diversity of species, otherwise known as coral reefs. These reef systems circumscribe the shallow seafloor surrounding islands and delimit the continental shelf edge abutting contiguous landmasses: They also populate sunken and receding sub-marine banks.

The reef systems of the Caribbean provide a wide range of services for almost 40 million people, which affect livelihood, economic progress, food security, cultural expressions and communion with nature. They are the basis of the tourism and fishing industries in the insular Caribbean and most of Central America, Mexico and the south-eastern United States. Both tourism and fisheries development are major contributors to GDP and employment in the region.

The interactions of the peoples of the Caribbean with the reef ecosystems do carry a cost in terms of pollution, mechanical destruction and degradation as well as the effects of climate change. These ravages impair and erode the functionality of the reef and thus their facility to deliver useful service.

If the impacts to the reef are to be avoided, or diminished or ameliorated and if the vitality and vigour of the system is to be retrieved and sustained over time - there would be the need to improve our knowledge and understanding of the extent of the impacts and how the reef ecosystems respond to these and what measures are needed to rescue and improve the situation. In this regard the intervention of science and specifically the genesis of the publication: '*Status and Trends of Caribbean Coral Reefs: 1970 – 2012*' becomes highly relevant. This seminal work by Professor Jeremy Jackson and his editorial team is the most comprehensive analysis and compilation of information on coral reef in the Caribbean over the past 40 years. Although the report clearly shows that there has been an ongoing decline in coral cover and reef health, there is also a strong message of hope that with the appropriate management interventions we can affect the desired outcome of a better balance between man and the reef environment.

Lisel Alamilla
Minister of Forestry, Fisheries
and Sustainable Development
Belize

INTRODUCTION

This is the 9th status report since the Global Coral Reef Monitoring Network (GCRMN) was founded in 1995 as the data arm of the International Coral Reef Initiative (ICRI) to document the ecological condition of coral reefs, strengthen monitoring efforts, and link existing organizations and people working on reefs worldwide. The US Government provided the initial funding to help set up a global network of coral reef workers and has continued to provide core support. Since then, the series of reports have aimed to present the current status of coral reefs of the world or particular regions, the major threats to reefs and their consequences, and any initiatives undertaken under the auspices of ICRI or other bodies to arrest or reverse the decline of coral reefs.

IUCN assumed responsibility for hosting the global coordination of the GCRMN in 2010 under the scientific direction of Jeremy Jackson with the following objectives:

1. Document quantitatively the global status and trends for corals, macroalgae, sea urchins, and fishes based on available data from individual scientists as well as the peer reviewed scientific literature, monitoring programs, and reports.
2. Bring together regional experts in a series of workshops to involve them in data compilation, analysis, and synthesis.
3. Integrate coral reef status and trends with independent environmental, management, and socioeconomic data to better understand the primary factors responsible for coral reef decline, the possible synergies among factors that may further magnify their impacts, and how these stresses may be more effectively alleviated.
4. Work with GCRMN partners to establish simple and practical standardized protocols for future monitoring and assessment.
5. Disseminate information and results to help guide member state policy and actions.

The overarching objective is to understand why some reefs are much healthier than others, to identify what kinds of actions have been particularly beneficial or harmful, and to vigorously communicate results in simple and straightforward terms to foster more effective conservation and management.

This and subsequent reports will focus on separate biogeographic regions in a stepwise fashion and combine all of the results for a global synthesis in the coming years. We began in the wider Caribbean region because the historical data are so extensive and to refine methods of analysis before moving on to other regions. This report documents quantitative trends for Caribbean reef corals, macroalgae, sea urchins, and fishes based on data from 90 reef locations over the past 43 years. This is the first report to combine all these disparate kinds of data in a single place to explore how the different major components of coral reef ecosystems interact on a broadly regional oceanic scale.

We obtained data from more than 35,000 ecological surveys carried out by 78 principal investigators (PIs) and some 200 colleagues working in 34 countries, states, and territories throughout the wider Caribbean region. We conducted two workshops in Panama and Brisbane, Australia to bring together people who provided the data to assist in data quality control, analysis, and synthesis. The first workshop at the Smithsonian Tropical Research Institute (STRI) in the Republic of Panama 29 April to 5 May, 2012 included scientists from 18 countries and territories to verify and expand the database and to conduct exploratory analyses of status and trends. Preliminary results based on the Panama Workshop were presented to the DC Marine Community and Smithsonian Institution Senate of Scientists in May 2012 and at the International Coral Reef Symposium (ICRS) and annual ICRI meeting in Cairns, Australia in July 2012. The second workshop in Brisbane, Australia in December 2012 brought together eight coral reef scientists for more detailed data analysis and organization of results for this report and subsequent publications. Subsequent presentations to solicit comments while the report was being finalized were made in 2013-2014 at the ICRI General meeting in Belize, the biennial meeting of the Association of Island Marine Laboratories in Jamaica, the Panamerican Coral Reef Congress in Merida, Mexico, the annual meeting of the Western Society of Naturalists, and numerous universities in Costa Rica, the USA and Europe.

The main body of the report is in two sections. Part I provides an overview of overall status and trends and detailed analyses of the multiple factors responsible for the decline of reef corals throughout the entire wider Caribbean region. The editors are grateful to all the people who have so generously provided data and expertise, but we assume responsibility for the many statements, conclusions and recommendations and final wording of the text. Part II provides a more detailed analysis of the status and trends of coral reef ecosystems in the 32 countries, states, and territories for which we have data. The format includes maps indicating all locations sampled, a detailed table of data sources and sites surveyed, timelines of ecologically important events, and relevant references. Each of these reports was compiled in consultation with local experts and all those who provided data and advice are listed as authors of each country report.

ACKNOWLEDGMENTS, CO-SPONSORS, AND SUPPORTERS OF GCRMN

Producing this report would have been impossible without the voluntary contributions of many people who are working to study, monitor, and conserve the coral reefs of the greater Caribbean region. We wish to specifically thank Carl Gustaf Lundin for his steadfast support of our vision to strengthen the underlying science for coral reef management and conservation. James Oliver at IUCN headquarters and Anne Caillaud at ICRI Australia provided essential administrative support and Sylvie Rockel at IUCN provided invaluable assistance in the final editing. We also thank the following for their generous assistance in helping us to connect with others, providing references or photos, and gathering crucial metadata: Ameer Abdulla, Octavio Aburto, Alejandro Acosta, Lorenzo Alvarez Filip, Nilda Aponte, Alejandro Arrivillaga, Jerry Ault, James Azueta, Julio Baisre, Brian Beck, Juan Eduardo Bezaury Creel, Kate Brown, Lauretta Burke, Georgina Bustamante, Celso Cawich, Leandra Cho-Ricketts, Rachel Collin, Roberto Colon, Martha Davis, Christine Dawson, Russell Day, Owen Day, Mark Eakin, Nicola Foster, Helen Fox, David Freestone, Graciela Garcia-Moliner, Jaime Garzon Ferreira, Janet Gibson, Bob Glazer, David Guggenheim, Scott Hajost, Marea Hatziolos, Jane Hawkridge, Rob Hedges, Sarah Hile, Zandy Hillis-Starr, Eric Hochberg, Miriam Huitric, AG Jordán-Garza , Ruy Kikuchi, Judy Lang, Thomas Laughlin, Ken Lindeman, Kathryn Lohr, MA Maldonado, Nyawira Muthiga, David Obura, John Ogden, Adrian Oviedo, Beatrice Padovani Ferreira, Francisco Pagan, Matt Patterson, Shari Sant Plummer, Yves Renard, Lionel Reynal, Katie, Reytar, Laura Richardson, Kimberly Roberson, Callum Roberts, Marisol Rueda, Carlos Saavedra, Yvonne Sadovy de Mitcheson, Héctor Salvat Torres, Elizabeth Selig, Omar Shamir Reynoso, Hélène Souan, Craig Starger, Jerker Tamelander, John Thompson, Abel Valdivia, Neil van Niekerk, Tserk van Rooij, Alessandra Vanzella-Khouri, L Vázquez-Vera, Beverly Wade, Clive Wilkinson, Stacey Williams, and Fernando Zapata. We'd also like to thank our interns and internal staff for their help developing this project and report: Craig Beatty, Nour Elshabassi, Amanda Feuerstein, Lauren Franck, Andy Hooten, Whitney Goodell, Patrick Curry, and Jamie Pratt. We would also like to thank Joseph Lecky for contributing his cartographic skills, and Anne Caillaud and Aurea Jaime-Sanchez for translating the executive summary. The following programs provided generous support through collaboration and data sharing: Atlantic and Gulf Rapid Reef Assessment (AGRRA), Bahamas Reef Environment Educational Foundation (BREEF), Bermuda Zoological Society, Caribbean Adaptation to Climate Change (CPACC), Caribbean Coastal Marine Productivity Programme (CARICOMP), Caribbean Environment Programme (CEP), Corporación para el Desarrollo Sostenible del Archipiélago de San Andrés, Providencia y Santa Catalina (CORALINA), Florida Fish and Wildlife Conservation Commission (FWC), Healthy Reefs Initiative (HRI), ICRI Secretariat, Instituto de Investigaciones Marinas y Costeras (INVEMAR), IUCN Global Marine and Polar Programme (IUCN-GMPP), Mainstreaming Adaptation to Climate Change Project (MACC), National Park Service-South Florida/Caribbean Network (NPS-SFCN), NOAA Biogeography Branch, Reef Check, Stichting Nationale Parken Bonaire (STINAPA), The Khaled bin Sultan Living Oceans Foundation (LOF), and World Resource Institute (WRI). The above is an inevitably incomplete list, and thanks are due to the countless others who have contributed to the project as it developed over the past three years.

Support for the GCRMN comes from the ARC Centre of Excellence for Coral Reef Studies, Caribbean Environment Program, Global Marine and Polar Programme of the International Union for the Conservation of Nature, McQuown Foundation, French Ministère de l'Écologie du Développement durable et de l'Énergie, Ministry of Economic Affairs of the Netherlands, Smithsonian Tropical Research Institute, SPAW Protocol, United States State Department, Summit Foundation, and United Nations Environment Programme (UNEP).

Data from SCREMP were funded in part by a grant agreement from the Florida Department of Environmental Protection (EDP) through National Oceanic and Atmospheric Administration (NOAA) award No. NA08NOS426037 to FLDEP.

Status and Trends of Caribbean Coral Reefs: 1970-2012

EXECUTIVE SUMMARY

Jeremy Jackson

"Perhaps the most striking aspect of plant life on a coral reef is the general lack of it. It seems anomalous to even the casual observer that tropical reefs, notable for their dazzling profusion of animal life, are almost devoid of conspicuous plants."

Sylvia Earle, 1972

INTRODUCTION

Sylvia Earle's early observations upon Caribbean reefs describe a forgotten world. Caribbean coral reefs have suffered massive losses of corals since the early 1980s due to a wide range of human impacts including explosive human population growth, overfishing, coastal pollution, global warming, and invasive species. The consequences include widespread collapse of coral populations, increases in large seaweeds (macroalgae), outbreaks of coral bleaching and disease, and failure of corals to recover from natural disturbances such as hurricanes. Alarm bells were set off by the 2003 publication in the journal *Science* that live coral cover had been reduced from more than 50% in the 1970s to just 10% today. This dramatic decline was closely followed by widespread and severe coral bleaching in 2005, which was in turn followed by high coral mortality due to disease at many reef locations. Healthy corals are increasingly rare on the intensively studied reefs of the Florida reef tract, US Virgin Islands, and Jamaica. Moreover, two of the formerly most abundant species, the elkhorn coral *Acropora palmata* and staghorn coral *Acropora cervicornis*, have been added to the United States Endangered Species List. Concerns have mounted to the point that many NGOs have given up on Caribbean reefs and moved their attentions elsewhere.

It was against this gloomy backdrop that this study was undertaken to assess more rigorously than before the extent to which coral reef ecosystems throughout the wider Caribbean may have suffered the same fate, and if they have not, to determine what were the factors responsible. Various reports suggested that reefs in the southern Caribbean were in better ecological condition than elsewhere, with more live coral and reef fish. If this were true, understanding why some reefs are healthier than others would provide an essential first step for more effective management to improve the condition of coral reefs throughout the entire Caribbean region.

STRATEGY AND SCOPE OF THE PRESENT REPORT

Previous Caribbean assessments lumped data together into a single database regardless of geographic location, reef environment, depth, oceanographic conditions, etc. Data from shallow lagoons and back reef environments were combined with data from deep fore-reef environments

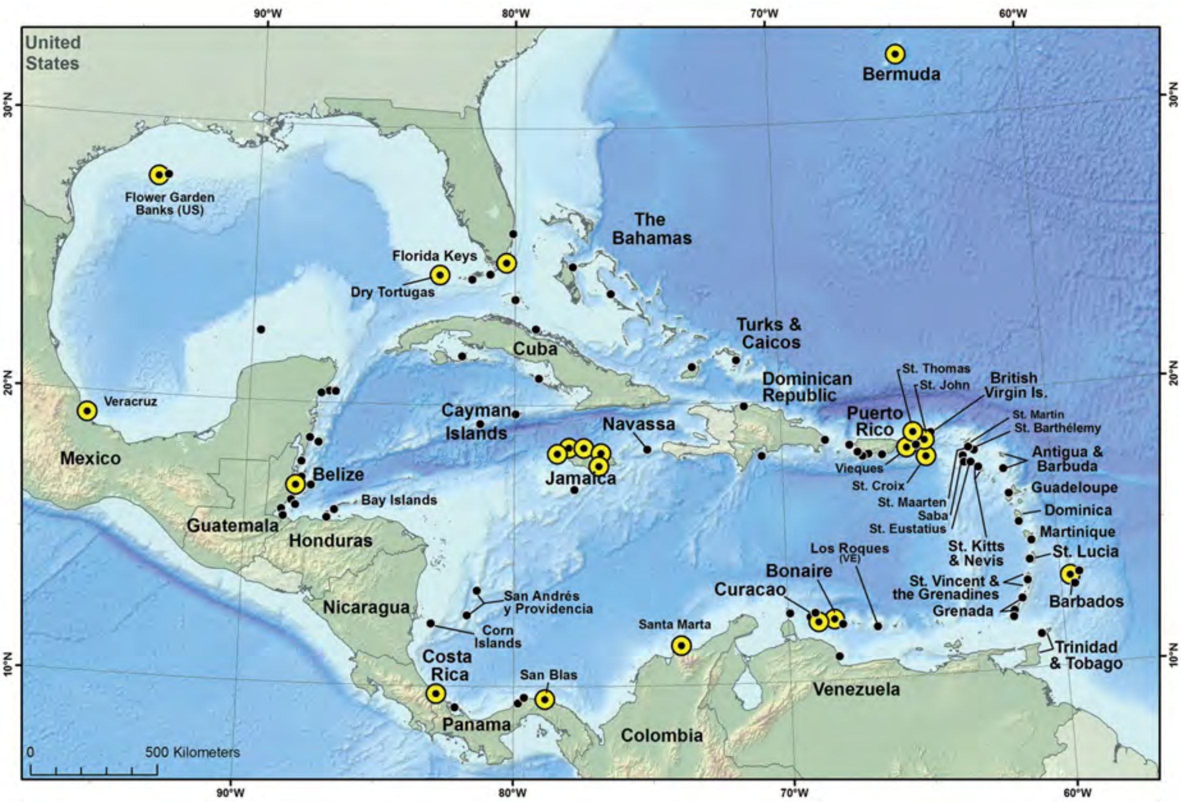

FIGURE 1. Distribution of 90 reef locations analyzed for this study. The large circles indicate 21 locations with the most complete time series data for analysis of long-term trends in coral cover.

and atolls. Geographic coverage was uneven, reflecting primarily the most studied sites with the most easily accessible data. Only total coral cover was recorded, with no attempt to assess the fates of different coral species. Nor was there any attempt to compile records of macroalgae, sea urchins, and fishes that are well known to have significant ecological interactions with corals.

We addressed these methodological problems by a detailed analysis of the status and trends of reef communities at distinct reef locations throughout the wider Caribbean. We also compiled essential metadata on the nature of the reef environment, depth, and history of human population growth, fishing, hurricanes, coral bleaching, and disease at each location. The quality of biological information varied among locations, but wherever possible data were obtained for coral and macroalgal cover, abundance of the critically important grazing sea urchin *Diadema antillarum*, and biomass of fishes, most importantly large grazing parrotfish.

Most of the quantitative data for Caribbean reefs is unpublished or buried in gray literature and government reports. To obtain these data, we contacted hundreds of people in all the countries of the Caribbean via several thousand emails, requests for data posted on relevant websites, and through presentations and interviews at international conferences. We also corresponded with managers of all the large monitoring programs in the region. In the end, we obtained data for corals, macroalgae, sea urchins, and fishes from a total of more than 35,000 quantitative reef surveys from 1969 to 2012. This is the largest amount of quantitative coral reef survey data ever compiled and exceeds by several fold that used for earlier Caribbean assessments.

Data are distributed among 90 reef locations in 34 countries (Fig. 1). Most of the data are from fore-reef and patch-reef environments in depths between 1-20 meters that are the focus of this study. Data are sparse up until the mass mortality of the formerly ubiquitous sea urchin *Diadema antillarum* in 1983-1984 when several monitoring programs first began. Data for corals are extensive

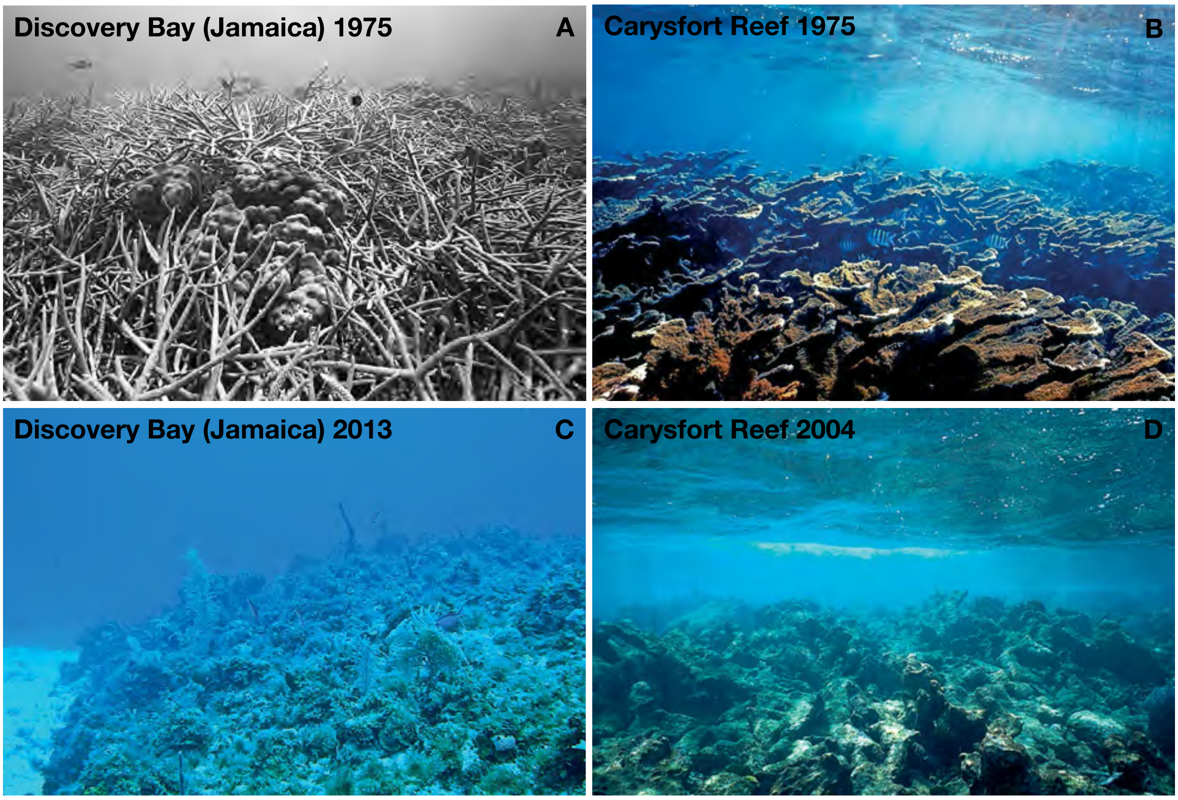

FIGURE 2. Phase shift from dominance by corals to dominance by macroalgae on the shallow fore-reefs in the northern Florida Keys and north coast of Jamaica. (A) Discovery Bay, Jamaica in 1975 and (C) the same location in 2013. (B) Carysfort Reef within the Florida Keys National Marine Sanctuary in 1975 and (D) in 2004 ((A, B, D by Phillip Dustan, and C by Robert Steneck).

and range from 1970 to the present. *Diadema* data are more limited up until mass mortality reduced its abundance to near zero and scientists realized what they had lost. Data for macroalgae are the most problematic because of inconsistent monitoring and taxonomy so that much of the data had to be discarded from our analysis. Quantitative data for both size and abundance of reef fishes needed to estimate fish biomass are unavailable until 1989 but are extensive after that.

The longest time series from the same reefs are large photo quadrats from 1973 to the present for fixed sites at Curaçao and Bonaire, with newer time series from the same islands beginning in the 1990s. Comparably long time series extending back into the early 1970s to early 1980s are available from the northern Florida Keys, Jamaica, St. John and St. Croix in the US Virgin Islands, and Panama. However, these records were compiled by different workers at different times and are therefore not as consistent or complete as data from the Dutch Caribbean.

Intensity of sampling varied greatly in time and space. We therefore partitioned the data into three time intervals of 12-14 years each based on major ecological events that extended throughout the wider Caribbean. These are:

1. 1970-1983: Interval from the oldest data up until and including the mass mortality of the formerly abundant sea urchin *Diadema antillarum* in 1983, as well as the first reports of White Band Disease (WBD) in the mid 1970s and early 1980s.
2. 1984-1998: From just after the *Diadema* die-off up to and including the widely reported 1998 extreme heating event.
3. 1999-2011: The modern era of massively degraded coral reefs.

PATTERNS OF CHANGE FROM 1970 TO 2012

Average coral cover for the wider Caribbean based on the most recent data for all the locations with coral data is 16.8% (range 2.8–53.1%). Taking into account the great variation among locations and data sets reduces this estimate to 14.3% (+2.0, -1.8). Even this more rigorously refined mean is 43% higher than the 2003 regional estimate of 10% cover. Nevertheless, coral cover declined at three quarters of the locations with the greatest losses for locations that were surveyed earliest and for the longest time.

Average coral cover for all 88 locations with coral data declined from 34.8% to 19.1% to 16.3% over the three successive time intervals, but the disparity among locations was great. In contrast, macroalgal cover increased from 7% to 23.6% between 1984-1998 and held steady but with even greater disparity among locations since 1998. The patterns were similar for the 21 locations with coral data from all three intervals highlighted by circles in Fig. 1. These opposite trends in coral and macroalgal cover constitute a large and persistent Caribbean phase shift from coral dominated to macroalgal dominated communities that has persisted for 25 years (Figs. 2 and 3), a pattern also strongly supported by ordination analyses of benthic community composition.

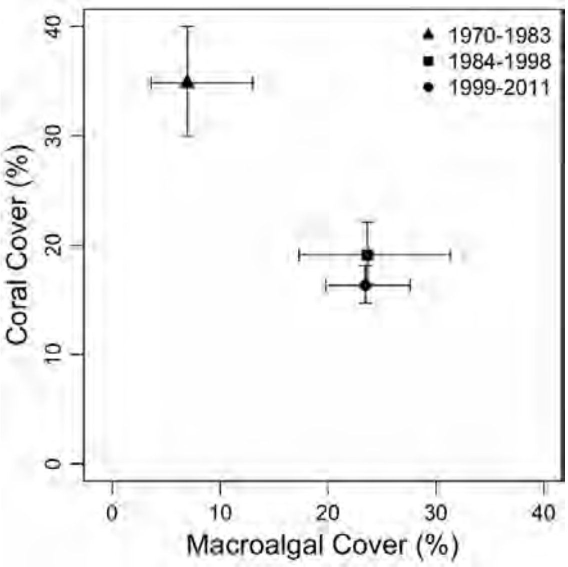

FIGURE 3. Large-scale shifts from coral to macroalgal community dominance since the early 1970s. Symbols and confidence intervals represent means and standard deviations for 3 time intervals that take into account variability due to location, and datasets using a mixed modeling framework.

The greatest overall changes in coral and macroalgal cover occurred between 1984 and 1998, after which there was little overall change at the great majority of locations except for places most strongly affected by the extreme warming events of 2005 and 2010. The same was true for formerly abundant elkhorn and staghorn *Acropora* that began to decline in the 1960s, the mass mortality of the sea urchin *Diadema antillarum* in 1983-1984, and the extreme overfishing of large parrotfish at most locations in the early to middle 20th century. Thus the largest and most damaging changes on Caribbean reefs occurred before most coral reef scientists and managers had begun to work on reefs, a classic example of the Shifting Baselines Syndrome and a harsh reminder that the problems of today are just the latest chapter in a much longer story of decline.

Looking beyond this general picture, however, long-term trends at the 21 highlighted locations in Fig. 1 exhibit three strikingly contrasting patterns of change in coral cover (Fig. 4). Trajectories for nine of the locations resemble a hockey stick with precipitous declines of 58-95% between intervals 1 and 2 followed by no change (Fig. 4A). In contrast, five other locations exhibited comparable decline that was spread out approximately equally between intervals 1 and 2 and between intervals 2 and 3 (Fig. 4B). The third group of seven locations exhibited much greater stability with overall changes (increase or decrease) of just 4-35% (Fig. 4C).

DRIVERS OF CHANGE

The drivers of the ecological degradation of Caribbean reefs need to be understood in the context of the highly unique situation of the Caribbean compared to other tropical seas. The Caribbean is effectively a Mediterranean sea that is the most geographically and oceanographically isolated tropical ocean on the planet. Isolation began tens of millions of years ago with the gradual break-up on the once circumtropical Tethys Seaway, the widening of the Atlantic Ocean, and ultimately isolation from the Eastern Pacific by the closure of the Panamanian Seaway 5.4 to 3.5 million years ago.

Consequently, Caribbean reef biotas are also highly distinctive. Many coral genera once combined with Pacific taxa have proven to belong to uniquely Atlantic evolutionary lineages based on molecular genetics. Moreover, acroporid corals that make up more than a third of Indo-Pacific coral diversity are represented by only two

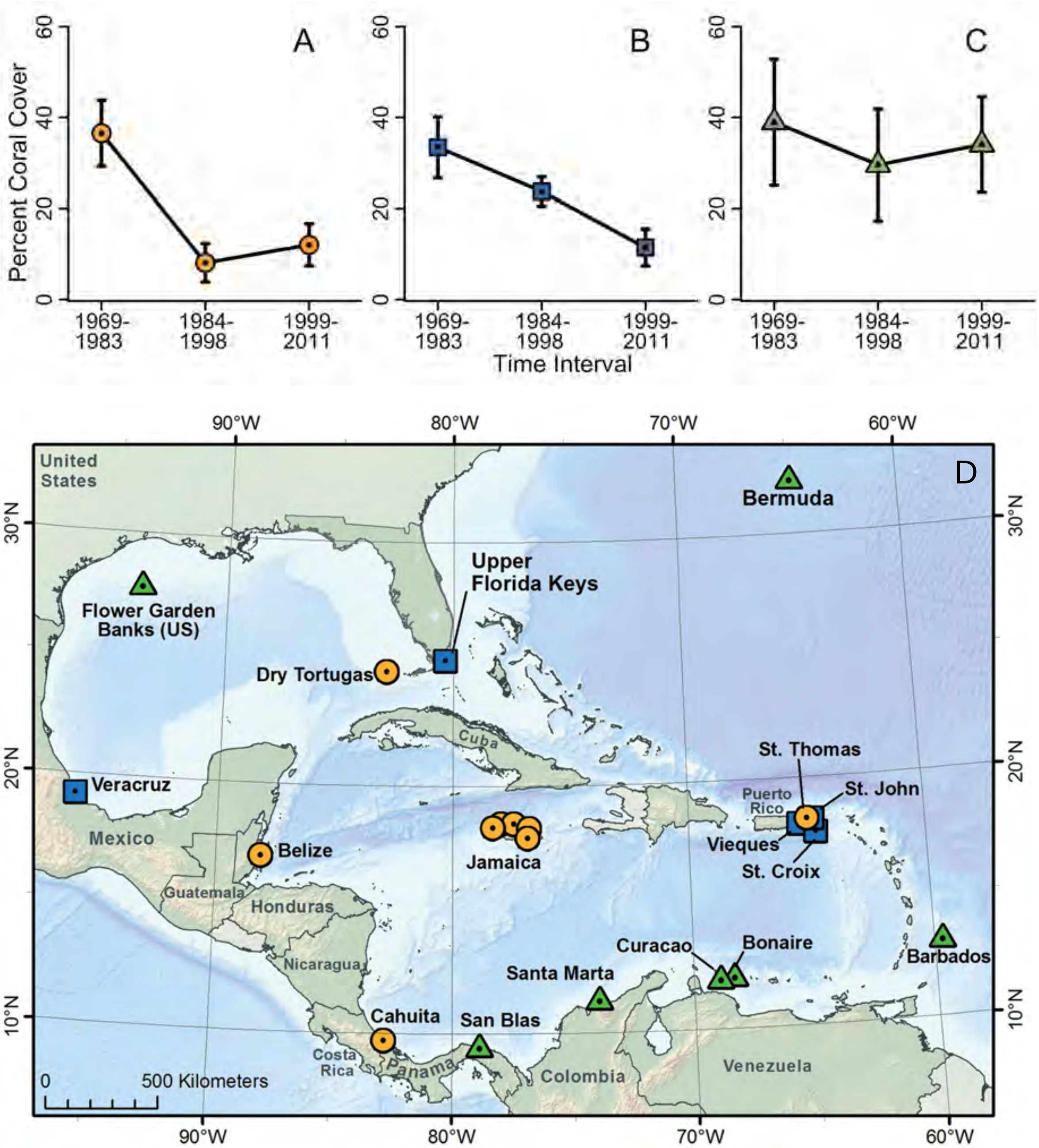

FIGURE 4. Trajectories of coral cover at 21 reef locations, grouped on the basis of the total amount of change over all three intervals and the tempo of change. (A) Hockey stick pattern with a steep decline between the first two intervals followed by little or no change. (B) Approximately continuous decline over all three intervals. (C) Comparative stability with smaller net changes in cover.

Caribbean species. Taxonomic diversity and ecological redundancy are low and the potential for rejuvenation from other regions is essentially nil. Caribbean species also had no evolutionary experience for dealing with exotic species and disease before the advent of people.

We focused on potential anthropogenic drivers of decline for which there were data for meaningful comparisons. Drivers were treated separately for ease of analysis and discussion, but they are inextricably linked. In particular, coral disease is a complex and poorly understood symptom of several forms of human disturbance rather than a direct driver of change. Thus disease is treated in relation to several different drivers including introductions of alien species, ocean warming, coastal pollution, and overfishing. Overall, results are stronger for evaluating effects of human population increase, overfishing, and ocean warming because there are more data, and less so for coastal pollution and invasive species.

FIGURE 5. Examples of mass tourism in the Caribbean. (A) Large cruise ships with thousands of passengers arrive every day in the Caribbean, shown here is St. Thomas, the US Virgin Islands (Source: Calyponte, Wikipedia). (B) Numerous hotel resorts offer ever more tourists the opportunity to stay in the Caribbean Sea, as here at Cancún Island, Mexico (Source: Foto Propia, Photo by Mauro I. Barea G., Wikipedia). (C) High density of tourists line South Beach, Miami, Florida (Source: Photo by Marc Averette, Wikipedia).

Too many people

Tourism is the lifeblood of many Caribbean nations (Fig. 5). However, our evidence demonstrates that extremely high densities of both tourists and residents are harmful to reefs unless environmental protections are comprehensive and effectively enforced. Unfortunately, this is only rarely the case. Numbers of visitors per square kilometer per year range from a low of 110 in the Bahamas to an astounding 25,000 at St. Thomas. All locations with more than the median value of 1,500 visitors per square kilometer per year have less than the median value of 14% coral cover except for Bermuda with 39% cover and Grand Cayman with 31%. The exceptional situation at Bermuda most likely reflects progressive environmental regulations in place since the 1990s and the infrastructure required to make them work. Otherwise, the harmful environmental costs of runaway tourism seem to be inevitable.

Overfishing

Artisanal fishing for subsistence is crucial to most Caribbean economies but the consequences have been catastrophic for coral reefs. Overfishing caused steep reductions in herbivores, especially large parrotfishes, which are the most effective grazers on Caribbean reefs but vulnerable to all gear types except hook and line.

Nevertheless, the consequences of overfishing parrotfish for coral survival were little understood until the abrupt demise of the sea urchin *Diadema antillarum* due to an unidentified disease in 1983-1984. Until then, *Diadema* had increasingly become the last important macro-herbivore on Caribbean reefs due to overfishing. *Diadema* and parrotfish strongly compete for food, and variations in their abundance were inversely proportional until 1983. This inverse relationship provides a rigorous proxy to assess the consequences of historical overfishing of parrotfish for coral cover in the absence of quantitative data for parrotfish biomass before 1989.

Our analysis of overfishing focused primarily on 16 of the 21 highlighted reefs in Fig. 1 for which quantitative data on *Diadema* abundance were available before the die-off in 1983/84, in addition to coral cover for all three of the time intervals in Fig. 3. Nine of these reefs were classified as overfished for parrotfishes by 1983, with *Diadema* densities ranging from 6.9-12.4 per square meter, whereas the other seven reefs were classified as less fished with *Diadema* densities of just 0.5-3.8 per square meter. This ranking agreed well with the qualitative literature. Reefs where parrotfishes had been overfished before 1984 suffered greater subsequent decreases in coral cover and increases in macroalgae than reefs that still had moderately intact populations of parrotfish. Coral and macroalgal cover were independent of *Diadema* densities before 1984 when either the sea urchin or parrotfish grazed down macroalgae to extremely low levels. All that changed, however, after the *Diadema* die-off when coral cover declined in proportion to historical *Diadema* abundance, a trend that has continued to the present day.

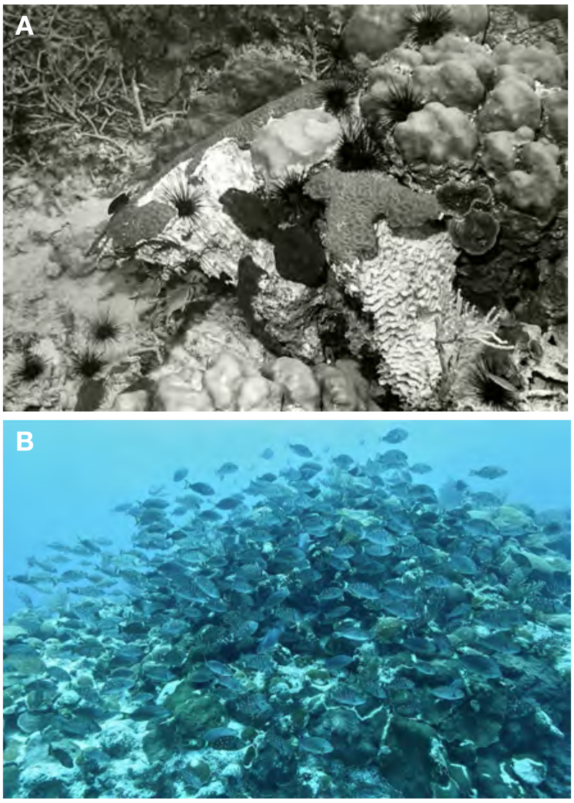

FIGURE 6. Formerly abundant grazers on Caribbean reefs. (A) Dense aggregation of the sea urchin *Diadema antillarum* on the west forereef at Discovery Bay, Jamaica in about 10 meters a year before the massive die-off in 1983/1984 (Photo by Jeremy Jackson). (B) Large school of Stoplight Parrotfish *Sparisoma viride* on the south shore of Bermuda where fishing on parrotfish is banned (Photo by Philippe Rouja). Such large numbers of parrotfish are rare to absent today on the great majority of Caribbean reefs.

There is also strong field and experimental evidence for persistent indirect effects of the increase in macroalgae, including decreased larval recruitment and survival of juvenile corals and increased coral disease. Coral recruitment sharply declined after 1984, at least in part due to a decline in the parental brood stock. But there is also strong evidence for active interference by macroalgae. Larval settlement onto the tops of experimental panels in Curaçao declined five-fold between identical experiments in 1979-1981 and 1998-2004. Crustose coralline algae, that are a preferred substrate for larval settlement, covered the entire upper surfaces of the panels in the earlier experiment and macroalgae were absent. In contrast, upper surfaces in the later experiment were entirely covered by macroalgae.

Other experiments demonstrate that coral larvae actively avoid substrates where macroalgae are present and larval recruits suffer increased mortality and growth inhibition due to physical interference by macroalgae. But the strongest evidence for macroalgal interference comes from recent large increases in coral recruitment and juvenile survival on reefs where *Diadema* have partially recovered or parrotfish have increased in marine protected areas. Experiments also demonstrate that macroalgae induce a wide variety of pathological responses in corals including virulent diseases. Release of toxic allelochemicals by macroalgae also disrupts microbial communities associated with corals sometimes causing bleaching or death.

FIGURE 7. Dense growths of macroalgae with surviving branch tips of *Porites* protruding through the algal canopy in the top right corner and previously overgrown dead branches of *Porites* and *Acropora cervicornis* in the bottom left (Dry Tortugas, 2000, Photo by Mark Chiappone).

Overfishing may have also indirectly affected the capacity of reefs to recover from damage by hurricanes; something they have routinely done for millions of years before or reefs would not exist. Over the past few decades, however, corals have increasingly failed to become reestablished on many reefs after major storms. We investigated this apparent shift using data for the 16 reefs with coral and *Diadema* data from before 1984. Coral cover was independent of the long-term probability of hurricanes before 1984 but not afterwards. Overfishing of parrotfish may have decreased the ability of corals to recover after hurricanes. Reefs protected from overfishing at Bermuda experienced four hurricanes since 1984 with no loss in average coral cover, whereas recently overfished reefs on the Central Barrier in Belize declined by 49% after 3 hurricanes.

FIGURE 8. Overfishing severely reduced fish biomass and diversity in the Caribbean. (A – C) Decline in the composition and size of coral reef trophy fish in the Florida Keys since the 1950s (modified from McClenachan 2008). (D – F) Parrotfish were the most important grazers on Caribbean reefs: (D) Stoplight parrotfish (*Sparisoma viride*) caught in a gill net. (E) A typical day of spearfishing off southeast Curaçao. (F) Fishing boats at Barbuda's Coco Point (Photos by Ayana Elizabeth Johnson).

Coastal pollution

Limited comparative data for water transparency based on secchi disk observations at three CARICOMP sites (Caribbean Coastal Marine Productivity Program by UNESCO) show that water quality is declining in areas of unregulated agricultural and coastal development. In particular, water transparency steeply declined over 20 years at Carrie Bow Cay in Belize due to huge increases in agriculture and coastal development from Guatemala to Honduras such as illustrated in Fig. 9C. A similar pattern was observed at La Parguera on the west coast of Puerto Rico. In contrast, water quality improved in Bermuda.

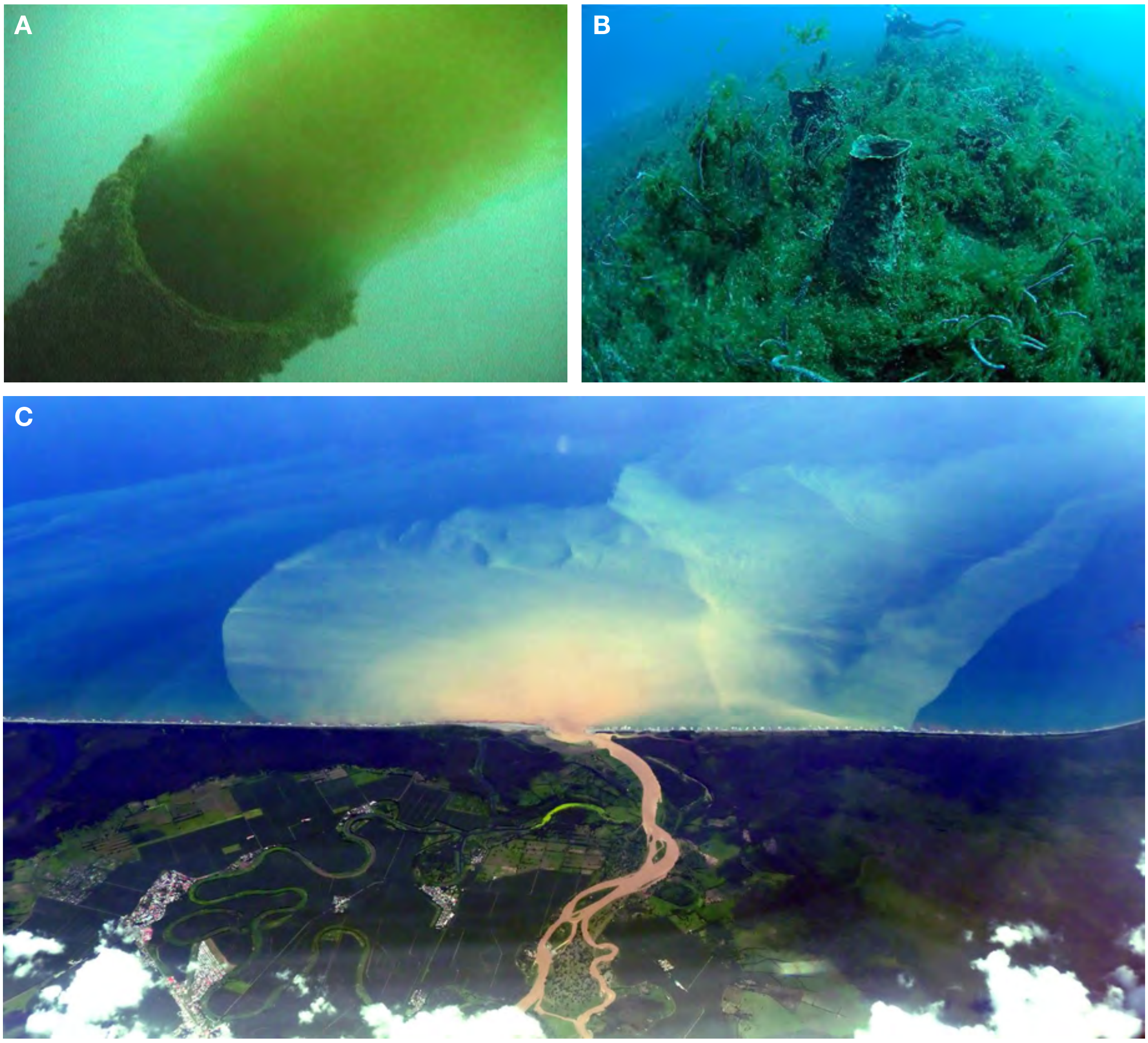

FIGURE 9. Impacts of coastal pollution on Caribbean reefs. (A) Sewage outfall in Delray Beach, Florida that discharges 13 million gallons per day of treated sewage up-current of a coral reef. (B) Macroalgae carpeting dead corals near the sewage outfall (Photos by Steve Spring, Marine Photobank). (C) Massive discharge of sediment loads by a river entering the Caribbean Sea off the Meso-American Coast (Photo by Malik Naumann, Marine Photobank).

Coral disease has been linked to excessive organic pollution but the data are spotty and limited in scope. In general there is a pressing need for more systematic and extensive monitoring of water quality throughout the wider Caribbean.

Ocean warming

Our first analyses were based on the Reefbase compilation of extreme bleaching events that showed no significant relationship between the numbers of extreme events per locality and coral cover at locations across the wider Caribbean, Gulf of Mexico and Bermuda. Because of the subjectivity of such bleaching assessments, however, we obtained data for degree heating weeks (DHWs) for all 88 localities with coral cover from NOAA Coral Reef Watch.

We then used these data to assess the effects of the 1998, 2005, and 2010 extreme warming events on coral cover by calculating the proportional changes in coral cover for the two years following each event in relation to the two years before the event, and then plotting the proportional change in relation to the numbers of degree heating weeks (DHWs) experienced at each locality. There is a weak but insignificant negative correlation between changes in coral cover and numbers of DHWs, regardless of whether the data were analyzed for each warming event or combined, or whether we included all the localities or restricted

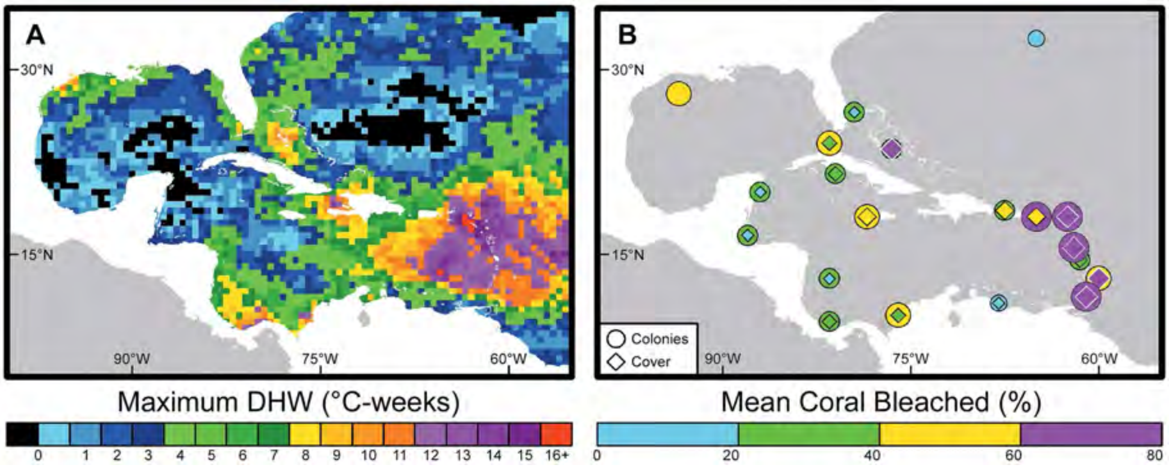

FIGURE 10. Extreme heating event and associated coral bleaching that most severely impacted the eastern Caribbean in 2005. (A) Degree heating weeks from Pathfinder Satellite observations. (B) Reports of the intensity of coral bleaching compiled from field observations (Courtesy Mark Eakin and colleagues).

the analysis to include only localities that experienced at least 8 DHWs. Moreover, the greatest losses in coral cover occurred at reef locations with less than 8 DHWs.

We caution that our results do not mean that extreme heating events are unimportant drivers of coral mortality due to coral bleaching and disease, as they clearly have been in the USVI, Puerto Rico, Florida Keys, and elsewhere. Moreover, increasingly severe extreme heating events will pose an even greater threat to coral survival in future decades. But our results do belie any regionally consistent effects of extreme heating events up to now and strongly imply that local stressors have been the predominant drivers of Caribbean coral decline to date.

Potentially deleterious effects of ocean acidification have not been treated here because of the lack of comparative data. If present trends of decreased pH continue, however, the ability of corals and other calcareous reef species to deposit skeletons will be increasingly compromised.

Invasive species

The explosion of exotic Pacific lionfish throughout the wider Caribbean (Fig. 12) has wreaked havoc in Caribbean fish communities. But as serious as the potential long-term consequences may be, they pale in comparison to the introduction of the unidentified pathogen that caused the die-off of *Diadema antillarum* or the effects of “White-band disease” (WBD) on acroporid corals. *Diadema* mass mortality began only a few km from the Caribbean entrance of the Panama Canal. That, coupled with orders of magnitude increases in bulk carrier shipping in the 1960s and 1970s, strongly suggests that *Diadema* disease was introduced by shipping. The same may be true of coral diseases although their earliest occurrences were widespread throughout the Caribbean.

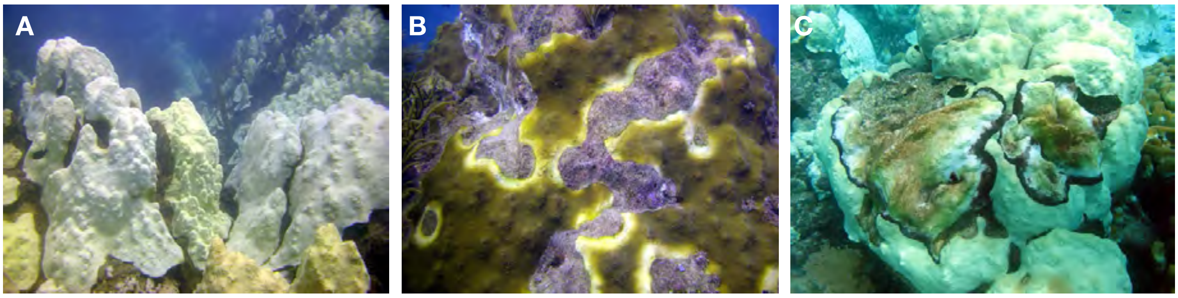

FIGURE 11. Effects of coral bleaching and disease on the formerly abundant coral *Orbicella faveolata*. (A) Bleached corals (Turrumote, Puerto Rico, 2005). Extensive partial colony mortality due to infection by (B) Yellow Band Disease (Turrumote, Puerto Rico, 2005) and (C) Black Band Disease (Los Roques Venezuela, 2010). (Photo A by Ernesto Weil; B & C by Aldo Cróquer).

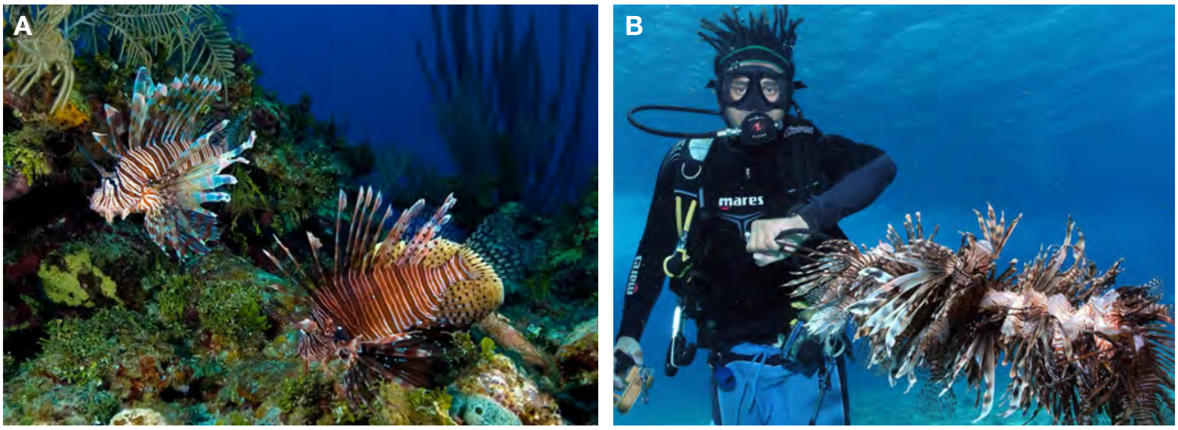

FIGURE 12. Population explosion of the highly successful Pacific lionfish (*Pterois volitans*) introduced into the Caribbean sometime between the 1980s and the early 1990s. (A) Abundant invasive lionfish on the reefs in the Cayman Islands (Photo courtesy of Niel Van Niekerk, with permission from IFAS, University of Florida). (B) Lionfish speared as part of a widespread effort to control the populations in the Dry Tortugas (Photo courtesy of ICRI).

Because of their isolation for millions of years, and by analogy to the fates of Native Americans after their first contact with Europeans, Caribbean species should be exceptionally prone to the impact of introduced diseases. And this appears to be the case. We know of no examples of the virtual elimination due to disease of any marine species throughout the entire extent of the Indian or Pacific oceans comparable to the demise of Caribbean *Diadema* and *Acropora*. This interpretation is also consistent with the apparent lack of any major environmental shift in the 1970s that might have triggered the outbreak of disease. Most importantly, the emergence of these diseases occurred many years before the first reported extreme heating events.

It would be possible to test this introduced species hypothesis for WBD since the pathogen is known and available for DNA-sequencing. It may also be possible for *Diadema* even though the pathogen is unknown by genetic analysis of entire frozen specimens of *Diadema* that died from the disease. This is not an entirely academic exercise: the two pivotal events in the demise of most Caribbean reefs are as much a mystery today as they were when they first occurred 30 or more years ago.

SUMMARY

Outbreaks of *Acropora* and *Diadema* diseases in the 1970s and early 1980s, overpopulation in the form of too many tourists, and overfishing are the three best predictors of the decline in Caribbean coral cover over the past 30 or more years based on the data available. Coastal pollution is undoubtedly increasingly significant but there are still too little data to tell. Increasingly warming seas pose an ominous threat but so far extreme heating events have had only localized effects and could not have been responsible for the greatest losses of Caribbean corals that had occurred throughout most of the wider Caribbean region by the early to mid 1990s.

In summary, the degradation of Caribbean reefs has unfolded in three distinct phases:

1. Massive losses of *Acropora* since the mid 1970s to early 1980s due to WBD. These losses are unrelated to any obvious global environmental change and may have been due to introduced pathogens associated with enormous increases in ballast water discharge from bulk carrier shipping since the 1960s.
2. Very large increase in macroalgal cover and decrease in coral cover at most overfished locations following the 1983 mass mortality of *Diadema* due to an unidentified and probably exotic pathogen. The phase shift in coral to macroalgal dominance reached a peak at most locations by the mid 1990s and has persisted throughout most of the Caribbean for 25 years. Numerous experiments provide a link between macroalgal increase and coral decline. Macroalgae reduce coral recruitment and growth, are commonly toxic, and can induce coral disease.
3. Continuation of the patterns established in Phase 2 exacerbated by even greater

overfishing, coastal pollution, explosions in tourism, and extreme warming events that in combination have been particularly severe in the northeastern Caribbean and Florida Keys where extreme bleaching followed by outbreaks of coral disease have caused the greatest declines.

IMPLICATIONS FOR MANAGEMENT

Our results contradict much of the rhetoric about the importance of ocean warming, disease, and hurricanes on coral reefs and emphasize the critical importance of historical perspective for coral reef management and conservation. The threats of climate change and ocean acidification loom increasingly ominously for the future, but local stressors including an explosion in tourism, overfishing, and the resulting increase in macroalgae have been the major drivers of the catastrophic decline of Caribbean corals up until today.

What this means is that smart decisions and actions on a local basis could make an enormous difference for increased resilience and wellbeing of Caribbean coral reefs and the people and enterprises that depend upon them. Thus, four major recommendations emerge from this report:

1. **Adopt robust conservation and fisheries management strategies** that lead to the restoration of parrotfish populations, including the listing of the parrotfish in relevant annexes of the Protocol concerning Specially Protected Areas and Wildlife (SPAW protocol) of the UNEP Caribbean Environment Programme. A recommendation to this effect was passed unanimously at the October 2013 International Coral Reef Initiative Meeting in Belize (see Box).
2. **Simplify and standardize monitoring** of Caribbean reefs and make the results available on an annual basis to facilitate adaptive management.
3. **Foster communication and exchange of information** so that local authorities can benefit from the experiences of others elsewhere.
4. **Develop and implement adaptive legislation and regulations** to ensure that threats to coral reefs are systematically addressed, particularly threats posed by fisheries, tourism and coastal development as determined by established indicators of reef health.

We understand that action upon these recommendations will be a matter of local and national socioeconomic and political debate. But the implications of our scientific results are unmistakable: *Caribbean coral reefs and their associated resources will virtually disappear within just a few decades unless all of these measures are promptly adopted and enforced.*

RECOMMENDATION

on addressing the decline in coral reef health throughout the wider Caribbean: the taking of parrotfish and similar herbivores

Adopted on 17 October 2013, at the 28th ICRI General Meeting (Belize City)

Background

The latest report of the Global Coral Reef Monitoring Network (GCRMN), entitled: *Status and Trends of Caribbean Coral Reefs: 1970-2012* is the first report to document quantitative trends of coral reef health based on data collected over the past 43 years throughout the wider Caribbean region.

The results of the study clearly show:

- Coral reef health requires an ecological balance of corals and algae in which herbivory is a key element;
- Populations of parrotfish are a critical component of that herbivory, particularly since the decline of *Diadema* sea urchins in the early 1980s;
- The main causes of mortality of parrotfish are the use of fishing techniques such as spearfishing and, particularly, the use of fish traps.

The Report further identifies that overfishing of herbivores, particularly parrotfish, has been the major drivers of reef decline in the Caribbean to date, concluding that management action to address overfishing at the national and local levels can have a direct positive impact on reef health now and for the future. *In some areas of the wider Caribbean (for example Bermuda and the Exuma Cays Land and Sea Park in the Bahamas, and more lately in Belize and Bonaire), active management including bans on fish traps, has led to increases in parrotfish numbers and consequent improvement in reef health and resilience to perturbations including hurricanes. This is in contrast to other areas within the Caribbean, where heavily fished reefs lacked the resilience to recover from storm damage.*

Positive impacts on reef health demonstrably have spill over effects on local economies, including the potential for alternative livelihoods to fishing, thanks to increased tourism revenues, replenishment of fish stocks and restoration of ecosystem services such as shoreline protection.

It is recognised that in the Caribbean there are varying levels of community reliance on fishing in general and the taking of parrotfish in particular. However, in light of the evidence now available, and in accordance with ICRI's Framework for Action cornerstone of 'integrated management' (which includes fisheries management), the International Coral Reef Initiative would like to highlight the benefits of strong management to protect reefs from overfishing, and urges immediate action to effectively protect parrotfish and similar herbivores.

Accordingly, the International Coral Reef Initiative urges Nations and multi-lateral groupings of the wider Caribbean to:

1. ***Adopt*** conservation and fisheries management strategies that lead to the restoration of parrotfish populations and so restore the balance between algae and coral that characterises healthy coral reefs;
2. ***Maximise*** the effect of those management strategies by incorporating necessary resources for outreach, compliance, enforcement and the examination of alternative livelihoods for those that may be affected by restrictions on the take of parrotfish;
3. ***Consider*** listing the parrotfish in the Annexes of the SPAW Protocol (Annex II or III) in addition to highlighting the issue of reef herbivory in relevant Caribbean fisheries fora;
4. ***Engage*** with indigenous and local communities and other stakeholders to communicate the benefits of such strategies for coral reef ecosystems, the replenishment of fisheries stocks and communities' economy.

SUMARIO EJECUTIVO

JEREMY JACKSON

“Quizás el aspecto más sorprendente del mundo vegetal en los arrecifes es su ausencia. Incluso al observador fortuito le parece fuera de lugar que los arrecifes tropicales conocidos por su extraordinaria profusión de vida animal estén casi desprovistos de plantas.”

Sylvia Earle, 1972

INTRODUCCION

Las observaciones iniciales de Sylvia Earle describen los arrecifes del Caribe como un mundo hoy olvidado. Los arrecifes coralinos del Caribe han sufrido una destrucción masiva de corales desde principios de los años 80 debido a una extensa variedad de impactos humanos que incluyen el crecimiento explosivo de la población, la sobrepesca, la contaminación de las zonas costeras, el calentamiento global y las especies invasoras.

Las consecuencias de estos impactos incluyen el general colapso de las poblaciones coralinas, el incremento the grandes algas (macroalgas), brotes the blanqueamiento y enfermedades, así como la incapacidad de recuperación de los corales frente a fenómenos naturales como los huracanes.

Un artículo publicado en 2003 en la revista *Science* hizo sonar la alarma cuando anunciaba que la cobertura de corales vivos había sido reducida de una media del 50% en los años 70 a tan sólo 10% hoy en día. Este declive espectacular fue seguido en el 2005 por un brote generalizado e intenso de blanqueamiento de corales, este fenómeno fue seguido a su vez de una epidemia que produjo una alta mortalidad en numerosos arrecifes de la región.

Corales saludables son una ocurrencia cada vez más fortuita en los arrecifes de coral de Florida Keys, Islas Vírgenes de los Estados Unidos y Jamaica. Por el contrario dos de las especies que un día fueron las especies más abundantes, el cuerno de alce *Acropora* palmata y cuerno de ciervo *Acropora* cervicornis, han sido inscritas en las lista de especies en peligro de extinción en los Estados Unidos. La preocupación ha llegado a tal nivel que muchas ONG han decidido desistir en la causa y dirigir sus esfuerzos a otras áreas de interés.

En este sombrío marco se realizó este estudio con la finalidad de evaluar con más rigor si los sistemas coralinos del Gran Caribe habían sufrido la misma suerte o en caso contrario determinar cuáles fueron los factores responsables. Varios estudios sugieren que los arrecifes en el Sur del Caribe se encuentran en mejores condiciones que el resto, con más coral vivo y más peces en los arrecifes. Si esto es cierto comprender porque algunos arrecifes se mantienen con más cantidad de corales vivos y peces, en mejores condiciones ecológicas que otros, podría ser el primer paso para una gestión más efectiva que mejore la condición de los corales en toda la Gran Región del Caribe.

ESTRATEGIA Y CONTENIDO DE ESTE ESTUDIO

Los resultados de previos estudios del Caribe se compilaron en un banco de datos común sin tener consideración de su situación geográfica, tipo de arrecife, profundidad o condiciones oceanográficas, etc. Los datos provenientes de lagunas y arrecifes posteriores poco profundos se mezclaron con datos de profundos atolones y arrecifes frontales. Las zonas geográficas que fueron estudiadas son discontinuas y reflejan primordialmente las áreas más estudiadas con los datos más fácilmente accesibles. Sólo se anotó la cobertura de coral sin ninguna intención de describir el destino de los diferentes tipos de corales. Tampoco se tomaron notas de las macroalgas, erizos de mar y peces cuyas vitales interacciones ecológicas con los corales son bien conocidas.

Hemos intentado remediar estos problemas de metodología analizando en detalle el estado y la tendencia de las comunidades del arrecife y distintas áreas de los arrecifes del Gran Caribe. También hemos intentado compilar en distintas áreas de los arrecifes meta data esencial sobre la naturaleza del medio ambiente del arrecife, la profundidad, la historia del crecimiento de la población humana, la pesca, los huracanes, el blanqueamiento del coral y sus enfermedades. La calidad de la información biológica varía entre los distintos lugares, pero siempre que fue posible obtuvimos datos sobre la cobertura de coral y macroalga, la crítica abundancia del herbívoro erizo de mar *Diadema antillarum* así como la biomasa de peces predominantemente los Peces Loro herbívoros de gran tamaño.

La mayoría de los datos cuantitativos sobre el Caribe no han sido publicados o están sumergidos en literatura gris y en los informes gubernamentales. Para obtener estos datos contactamos centenas de personas de todos los países del Caribe a través de miles de correos electrónicos y solicitamos datos de las redes de internet, de debates y entrevistas en conferencias internacionales. También contactamos a los administradores de todos los programas de monitoreo de gran escala en la región. Finalmente, obtuvimos más de 35,000 estudios cuantitativos con datos sobre corales, macroalgas, erizos de mar y peces desde 1969 hasta 2012. Este es el mayor número de estudios compilados hasta la fecha y supera con creces a cualquier estudio sobre el Caribe realizado previamente.

Los datos fueron recogidos en 90 arrecifes de 34 países (Fig.1) La mayoría de los datos provienen de los arrecifes frontales y de parche en aguas entre 1-20 metros de profundidad, siendo estas el foco de este estudio. Los datos fueron escasos hasta la mortalidad en masa del un día común erizo marino *Diadema antillarum* en 1983-84 cuando varios programas de monitoreo comenzaron. Los datos coralinos son extensivos desde 1970 hasta nuestros días. Los datos sobre la *Diadema* son más escasos hasta que la mortalidad en masa redujo su abundancia a casi cero y los científicos se dieron cuenta de lo que se había perdido. Los datos sobre macroalga son más problemáticos debido a la inconsistencia del monitoreo y la taxonomía hasta tal extremo que la mayor parte de los datos tuvieron que ser descartados. El tamaño y la abundancia, datos necesarios para estimar la biomasa del pez no comenzaron hasta 1989 pero son copiosos posteriormente.

Las series más antiguas desde 1973 hasta hoy tomadas en el mismo arrecife son cuadrados fotográficos de gran tamaño en áreas fijas de Curasao y Bonaire, con series más recientes de las mismas islas desde los años 90. Otras series disponibles desde los años 70 hasta los años 80 incluyen Florida Keys, Jamaica, St John y St Croix en las Islas Vírgenes de los Estados Unidos y Panamá. Sin embargo, estos datos fueron compilados por diferentes individuos en diferentes periodos y no son tan consistentes o completas como las del Caribe Holandés.

La intensidad del muestreo varía enormemente a través del tiempo y del espacio. Por este motivo dividimos los datos en tres intervalos de 12-14 años basados en acontecimientos ecológicos de gran escala que sucedieron el Gran Caribe.

1. 1970-1983: Este intervalo cubre desde los primeros datos hasta la mortalidad del erizo de mar *Diadema antillarum* en 1983 que una vez fue abundante, así como los primeros informes sobre la enfermedad de la banda blanca a mediados de los años 70 y principios de los 80.
2. 1984-1998: Desde la desaparición de la *Diadema* hasta el fenómeno de calentamiento extremo en 1998 inclusive.
3. 1999-2011: La era moderna de arrecifes coralinos en fase de severa degradación.

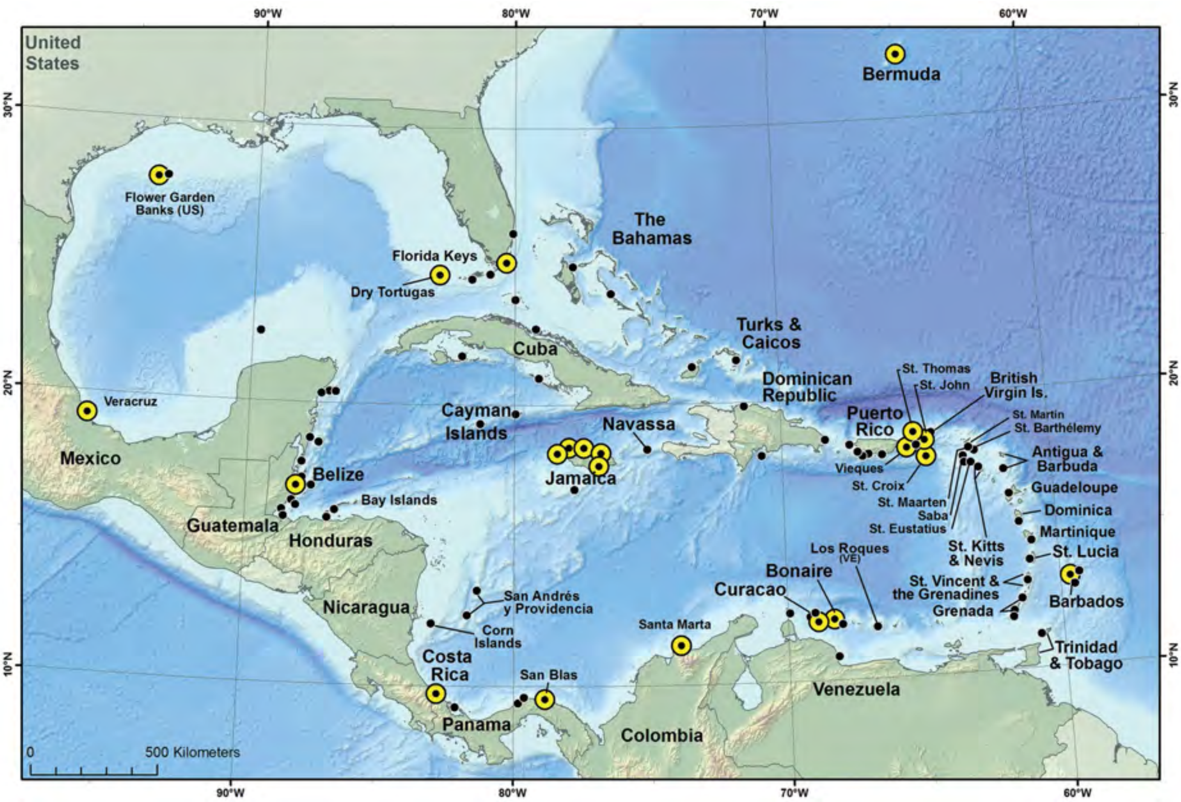

FIGURE 1. Distribución de 90 arrecifes analizados en este estudio. Los círculos amarillos indican 21 puntos de monitoreo con las series temporales más completas de análisis a largo plazo de las tendencias de la cobertura de coral.

MODELOS DE CAMBIO DE 1970 A 2012

La media de coral cobertura en el Gran Caribe basada en los datos más reciente en todas las áreas donde hay datos disponibles es 16.8% (entre 2.8-53.1%). Teniendo en cuenta la gran variedad entre las áreas y las series de datos las cifra se redujo al 14.3% (+2.0,-1.8). Incluso cuando esta media es el resultado de un riguroso proceso todavía es el 43% más elevada que el cálculo regional de 2003 de 10% de cobertura. No obstante, se observaron reducciones de la cobertura coralina en tres cuartas partes de las áreas observadas con las perdidas más importantes registradas en los lugares monitoreados desde hace más tiempo.

La cobertura coralina media de los 88 sitios en los que datos fueron obtenidos ha declinado del 34.8% al 19.1% al 16.3% durante estos tres intervalos aunque la disparidad entre las diversas áreas es considerable. En contraste, la cobertura de macroalga creció del 7% al 23.6% entre 1984 y 1998 y se mantuvo estable pero con una gran disparidad geográfica desde 1984. Los esquemas de las 21 localidades señalados en los círculos en la Figura 1 son semejantes en los tres intervalos. Las tendencias opuestas entre la cobertura de coral y macroalgal constituye una larga y persistente fase de cambio de dominación de las comunidades de macroalgas frente a los corales durante los últimos 25 años (Fig. 2 y 3). Este modelo es corroborado por los análisis de orden de composición de las comunidades bentónicas. En general, los cambios más significantes de cobertura de coral y alga ocurrieron entre 1984 y 1998, después de esta fecha no ha habido un gran cambio en términos generales excepto en lugares afectados por los extraordinarios fenómenos de calentamiento de 2005 y 2010. El mismo modelo se puede aplicar a los que una vez fueron abundantes cuerno de alce y cuerno de ciervo *Acropora* cuyo declive comenzó en los años 60; a la mortalidad en masa de *Diadema antillarum* en 1983-84, y a la amplia sobrepesca de pez loros grandes en la mayoría de las localidades, de principios a mediados del siglo 20. De esta manera los cambios más significantes y perjudiciales en los arrecifes del Caribe ocurrieron antes de que la mayoría de los científicos y administradores

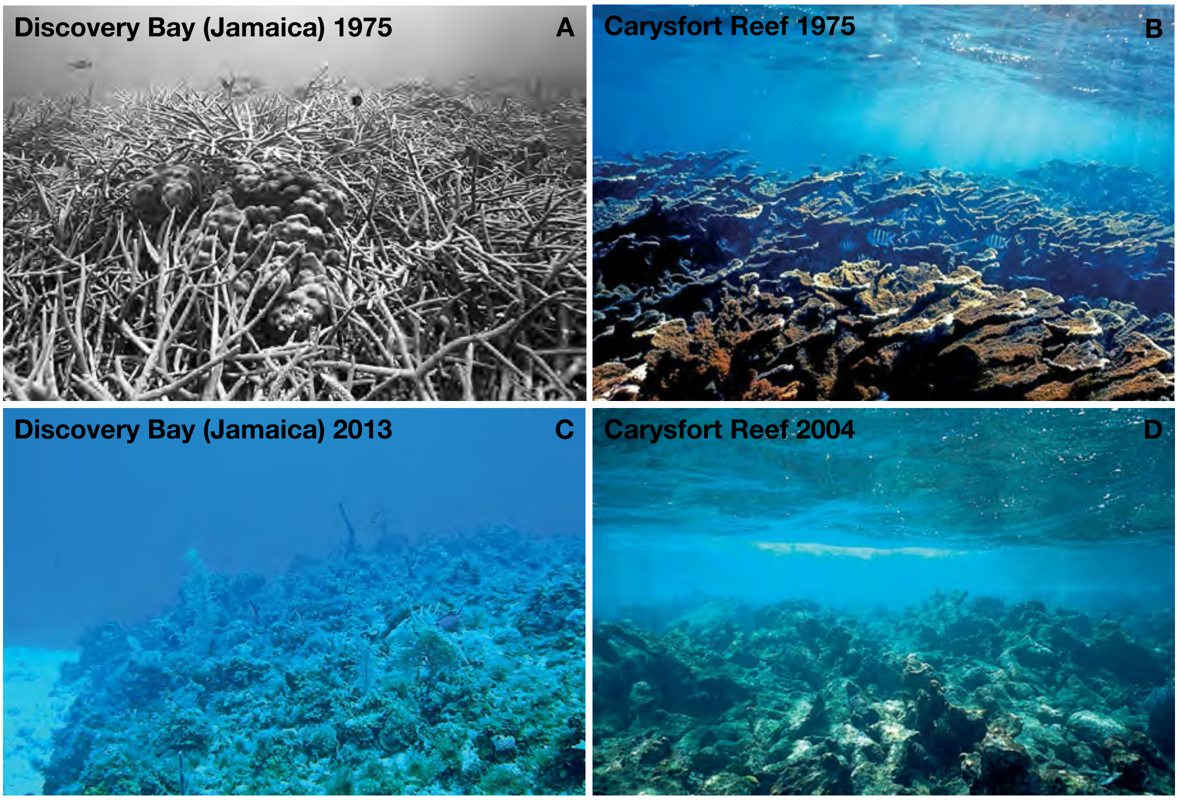

FIGURE 2. Transformación de la dominación de coral a macroalga en los arrecifes poco profundos del Norte de Florida Keys y la costa Norte de Jamaica. (A) Discovery,Bay, Jamaica en 1975 et (C) el mismo lugar en 2013. (B) Arrecife de Carysfort, en el corazón del Santuario Marino Nacional de Florida Keys en 1975 y (D) en 2004 (foto: Phillip Dustan).

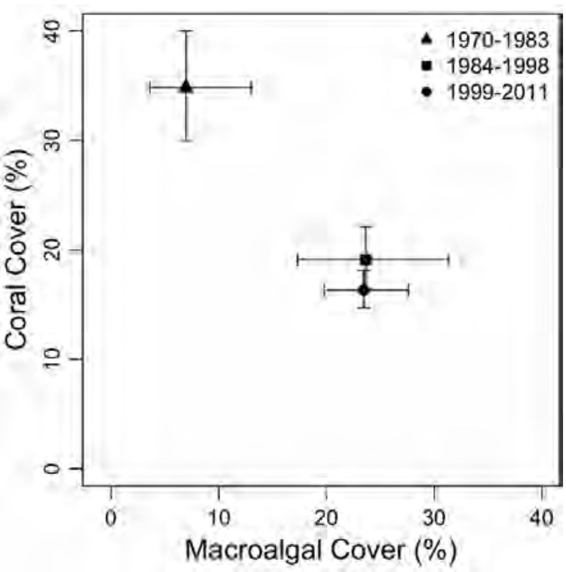

FIGURE 3. Permutaciones de la dominación de coral a macroalga comunidades a gran escala desde principios de los 70. Los símbolos e intervalos de confianza representan las medias y las desviaciones estándar de los tres intervalos, considerando la variabilidad de la localidad y los grupos de data mediante la utilización de un modelo mixto.

hubieran comenzado a trabajar en los arrecifes, un ejemplo clásico de cambio de punto de referencia (*shifting baseline* en inglés) y un aviso implacable de que los problemas de hoy son el último capítulo de una larga historia de declive.

Más allá de esta visión general, las tendencias a largo plazo en los 21 enclaves señalados anteriormente en la Fig. 1 muestran tres modelos de cambio de fuerte contraste en la cobertura de coral (Fig 3). Las trayectorias de nueve de estos enclaves se asemejan a un palo de hockey con descensos vertiginosos de 58 a 95% entre el primer y segundo intervalo seguidos de no aparente cambio (Fig. 4A). Por el contrario, otros 5 enclaves muestran un descenso similar pero extendido en igual medida entre el primer y segundo intervalo así como entre el segundo y el tercero (Fig.B) . El tercer grupo con siete enclaves muestra más estabilidad con cambios generales (ascenso o descenso) de tan sólo 4 al 35%(Fig. 4C).

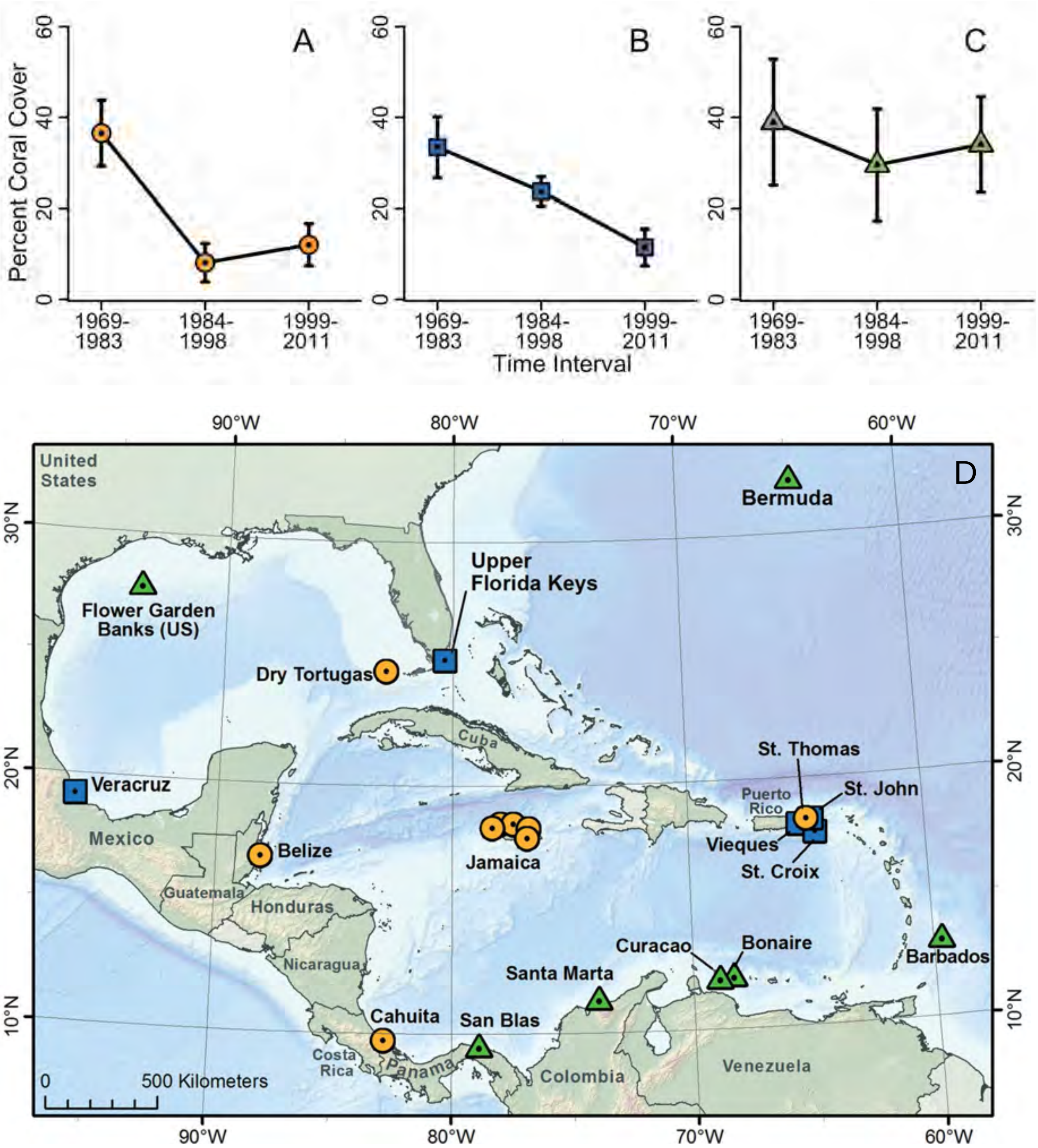

FIGURE 4.Trayectos del cambio de la cobertura coralina de los 21 puntos en el mapa, agrupados en función de la cantidad total de cambio durante los tres intervalos y el ritmo de cambio. (A) Trayectoria representada con palo de hockey mostrando un declive abrupto entre los dos intervalos primeros, seguido de un cambio mínimo o no cambio (B) Declive continúo durante los tres intervalos. (D) Estabilidad comparativa con cambios de cobertura netos mucho menores.

PROMOTORES DE CAMBIO

Las causas de la degradación de los arrecifes del Caribe han de ser entendidas en el contexto de la situación única del Caribe comparada con cualquier otro mar tropical. El Caribe se trata de un tipo de mar Mediterráneo tratándose del mar tropical más aislado del mundo geográfica y oceanográficamente hablando.

Este aislamiento se remonta a decenas de millones de años y comenzó con el desmembramiento gradual del que fuera el mar de Tethys que entonces circundaba la tierra, la apertura del Océano Atlántico y finalmente el aislamiento del Pacífico del este mediante el cierre del Ismo de Panamá desde hace unos 5.4 a 3.5 millones de años.

Como consecuencia la biota del Caribe muestra grandes singularidades. Estudias de genética molecular se han demostrado que numerosos géneros de corales, una vez combinados con taxa del Pacífico, pertenecen a líneas únicas evolucionarias del Atlántico. Además los acroporidos que representan más de una tercera parte de la diversidad coralina del Indo Pacífico sólo están presentes en dos especies del Caribe. La diversidad taxonómica y la redundancia ecológica son escasas, así como el potencial de rejuvenecimiento a través de otras regiones es esencialmente nulo. Las especies del Caribe no tienen la experiencia evolucionaria para competir con especies exóticas y enfermedades traídas por los seres humanos.

Este estudio se concentra en potencial causas antropogénicas de declive cuyos datos disponibles son suficientes para hacer comparaciones que tengan algún significado.

Las causas han sido tratadas en separado a fin de facilitar el análisis y la discusión pero estas están inextricablemente vinculadas. Así las enfermedades se interrelacionan con otras causas como la introducción de especies, el calentamiento del océano, la contaminación de las zonas costeras y la sobrepesca. En general, los resultados más significativos provienen de la evaluación de los efectos del incremento de la población, la sobrepesca y el calentamiento debido al mayor número de datos disponibles, y menores por la contaminación costera y las especies invasoras.

Una población excesiva

El turismo es la principal fuente económica de muchas naciones en el Caribe (Fig. 5). Sin embargo, nuestra evidencia demuestra que altas densidades ambas de turistas y residentes son perjudiciales para los arrecifes si estos no están protegidos por medidas ambientales que sean exhaustivas y ejecutadas con eficiencia. Desgraciadamente esto no es la norma. Los números de visitantes por kilómetro cuadrado por año varían desde 110 en las Bahamas a la increíble cifra de 25,000 en Santo Thomas (Islas Vírgenes). Todas las localidades con una media de más de 1500 visitantes por kilómetro cuadrado por año tienen menos de una media de 14% de coral excepto Las Bermudas con 39% y Gran Caimán con 31%. La situación excepcional en Las Bermudas seguramente refleja las regulaciones progresivas del medio ambiente desde 1990 y la infraestructura requerida para hacerlas funcionar. De lo contrario los nocivos efectos del turismo de masa parecen ser inevitables.

FIGURE 5. Sobrepoblación: el turismo de masa en el Caribe (A) Enormes cruceros con miles de pasajeros en Santo Tomas, en las Islas Vírgenes de US (Calyponte, Wikipedia) (B) Grandes complejos hoteleros están alineados a lo largo de la costa en la isla de Cancún, México (Foto Oropia, Foto de Mauro I. Barea G., Wikipedia). (C) Touristas en la playa de Sur en Miami, Florida (Foto de Marc Averette, Wikipedia).

Sobrepesca

La pesca artesanal de subsistencia juega un papel primordial en la mayoría de las economías caribeñas, pero sus consecuencias en los arrecifes coralinos son catastróficas. La sobrepesca ha conllevado a reducciones vertiginosas de peces herbívoros, especialmente los grandes peces loro, los herbívoros más eficaces del Caribe, pero los más vulnerables a todos los tipos de pesca salvo el anzuelo y el sedal.

En cualquier caso las consecuencias de las sobrepesca de los peces loro en relación a la supervivencia del coral fueron muy poco comprendidas hasta el colapso de Diadema antillarum una vez

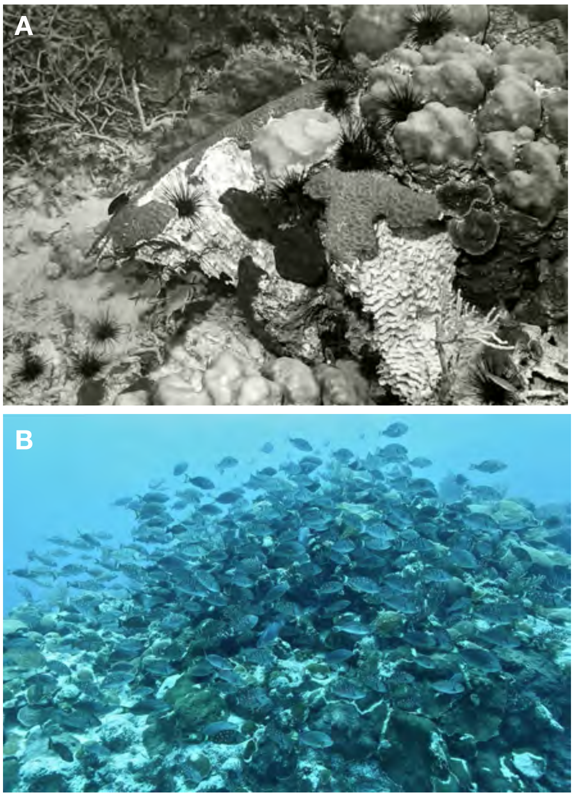

FIGURE 6. La sobrepesca ha diezmado la biomasa y diversidad en el Caribe. (A-C)Declive en la composición y tamaño de los peces trofeo en Florida Keys desde 1950 (adaptada por McClenachan 2008). (D-F) El pez loro era uno de los herbívoros más importantes en los arrecifes del Caribe: (D) Pez loro semáforo atrapado en una red (Sparisoma viride). (E) Un día típico de pesca de fusil en el sudeste de Curasao. (F) Barcos pesqueros en Punto Coco en Barbados (Fotos Ayana Elizabeth Johnson)

FIGURE 7. Los que una vez fueron abundantes grupos de herbívoros en los arrecifes caribeños. (A) Densa agrupación de erizos de mar (*Diadema*) en el arrecife anterior del Oeste de Discovery Bay, Jamaica a 10 metros de profundidad, un año ante de la hecatombe en 1983/84 (Foto Jeremy Jackson). (B) Un numeroso banco de pez loro semáforo en la costa sur de Bermuda, donde la pesca del pez loro está prohibida (Foto Phillipp Rouja). Tales agrupaciones de peces loro son extremadamente raras o inexistentes en la mayoría de los arrecifes caribeños en la actualidad.

abundante, quien se había convertido en el último macro-herbívoro de gran significancia en los arrecifes del Caribe hasta su abrupta casi-desaparición a causa de una epidemia no identificada en 1983-84. Diadema y peces loro son grandes competidores de sus fuentes alimenticias y variaciones de su abundancia son inversamente proporcionales hasta 1983. Esta inversa relación provee una base rigurosa para establecer las consecuencias de la continua sobrepesca del pez loro en relación a la cobertura de coral ya que no poseemos datos cuantitativos de la biomasa del pez loro anteriormente a 1989.

Nuestro análisis se concentró principalmente en 16 de los 21 arrecifes señalados en la Fig. 1 que tenían datos cuantitativos sobre la abundancia de la *Diadema* antes de su mortalidad en masa en 1983-84, además de datos sobre la cobertura de coral durante los tres intervalos en Fig. 3. Nueve de estos arrecifes fueron clasificados como sobreexplotados por peces loros antes de 1983, con densidades de *Diadema* variando entre 6.9-12.4 por kilómetro cuadrado, mientras que los otros arrecifes fueron clasificados como menos sobreexplotados con densidades de *Diadema* de solo 0.5-3.8 por kilómetro cuadrado. Esta clasificación coincide con la literatura cualitativa. Arrecifes donde los peces loro fueron sobreexplotados antes de 1984 sufrieron mayor degradación en la cobertura de coral que aquellos que todavía conservaban poblaciones de peces loro intactas. La relación entre corales y cobertura de macroalga fue independiente de la densidad de *Diadema* antes de 1984 cuando uno de los dos el erizo de mar o el pez loro consumían la macroalga a niveles extremadamente bajos. Todo cambio cuando la *Diadema* sucumbió y consecuentemente la cobertura de coral declinó en proporción directa a la histórica abundancia de la *Diadema*, tendencia que continua hasta nuestros días.

También existen solidas pruebas experimentales y en el terreno de efectos indirectos y persistentes en macroalgas incluido el declive del reclutamiento de larvas y de la supervivencia de corales juveniles, así como el incremento de enfermedades en los corales. El reclutamiento de larvas declinó rápidamente después de 1984 en parte debido al declive de la base de progenitores. Pero también hay evidencia palpable de la activa interferencia de la macroalga.

FIGURE 8. Denso crecimiento de la macroalga: a la derecha superior extremidades de corales *Porites* sobrevivientes son visibles entre el follaje mientras que en la parte inferior a la izquierda ramas de corales *Porites* y *Acropora cervicornis* han sido ya sofocadas (Dry Tortugas, 2000, foto de Mark Chiappone).

La deposición larval sobre paneles experimentales en Curasao fue cinco veces menor entre experimentos idénticos en 1979-1981 y 1998-2004. En los primeros experimentos algas calcáreas incrustantes, uno de los substratos preferidos de las larvas, cubrían completamente la superficie de los paneles mientras que macroalga no estaba presente. En los últimos experimentos, sin embargo, las superficies estaban cubiertas de algas.

Otros experimentos muestran que la larva coralina evita activamente los substratos donde las macroalgas están presentes, y que las larvas sufren mortalidad e inhibición del crecimiento debido a la física interferencia con la macroalga. Pero la mayor evidencia de la interferencia de la macroalga ha sido observada en el reciente gran aumento en reclutamiento de coral y supervivencia juvenil en los arrecifes donde *Diadema* se ha recuperado parcialmente o en áreas protegidas donde los números de peces loro han incrementado.

Experimentos también demuestran que la macroalga induce a una extensa variedad de respuestas patológicas en corales incluso enfermedades virulentas. Las macroalgas también segregan substancias alelo-químicas tóxicas que perturban las comunidades microbioticas asociadas con los corales causando descoloración o fatalidades.

La sobrepesca también afecta indirectamente la capacidad de los arrecifes de recuperarse de los huracanes, algo que habían sido capaces de hacer anteriormente por millones de años o simplemente los corales no existirían. En las últimas décadas, sin embargo, los corales no han sido capaces de reestablecerse en muchos arrecifes después de grandes tormentas. Hemos estudiado este aparente cambio utilizando datos de antes de 1984 en16 arrecifes con corales y *Diadema*. La cobertura de coral era independiente a largo plazo de la probabilidad de que un huracán ocurriese antes de 1984, pero no después. La sobrepesca del pez loro puede haber reducido la habilidad de los corales de recuperarse de los huracanes. Arrecifes protegidos de la sobrepesca en las Bermudas han pasado por cuatro huracanes desde 1984 sin haber perdido la media de cobertura de coral, mientras que los arrecifes sobrepescados recientemente en la Barrera Central de Belice sufrieron un declive del 49% después de tres huracanes.

Contaminación de la zona costera

Una limitada serie de datos comparativos de visibilidad en el agua, basados en observaciones a través del Secchi disk en 4 enclaves CARICOMP (Programa de la UNESCO sobre la productividad costera y marina del Caribe), muestra que la calidad del agua declina rápidamente en áreas donde existe una carencia de regulación de la agricultura y del desarrollo de la zona costera. En particular, la transparencia del agua ha empeorado en gran medida desde hace 20 años en Carrie Bow Cay en Belize debido a un extraordinario auge de la agricultura y del desarrollo en las zonas costeras de Guatemala a Honduras, ilustradas en la figura 9C. Un patrón similar fue observado en la Parguera en la costa oeste de Puerto Rico. Por el contrario la calidad del agua mejoro en las Bermudas.

Las enfermedades del coral han sido vinculadas a un exceso de contaminación orgánica pero los datos son esporádicos y con objetivos limitados. En general hay una necesidad imperativa de establecer un monitoreo sistemático y extensivo de la calidad del agua en la Gran Región del Caribe.

Calentamiento del Océano

Nuestros primeros análisis se basaron en la compilación, a través del banco de datos Reefbase, de acontecimientos de extremo blanqueamiento que no mostraban relación significante entre los números de acontecimientos extremos entre localidades y su cobertura de coral en localidades del Gran Caribe, Golfo de Méjico y las Bermudas. Sin embargo, debido a la subjetividad de estas evaluaciones de blanqueamiento, obtuvimos datos de grados de semanas de calentamiento (en inglés Degree Heating Weeks DHWs) de las 88 localidades con cobertura de coral de NOAA Coral Reef Watch (programa de vigilancia de los arrecifes coralinos).

Seguidamente utilizamos datos para evaluar los efectos de extremo calentamiento de 1998, 2005 y 2010 en la cobertura de coral. Primero calculamos los cambios proporcionales de la cobertura de coral durante los dos años después del fenómeno en relación a los dos años anteriores al fenómeno, posteriormente representamos el cambio proporcional en relación a los números de grados semanales de calentamiento para cada localidad. Hay una débil pero insignificante correlación negativa entre la perdida de cobertura de coral y los valores de grados semanales de calentamiento, independientemente del hecho de que los datos de cada fenómeno sean analizados por separado o conjuntamente; si incluimos todas las localidades o si limitamos los análisis a localidades donde se experimentaron un mínimo de 8 grados semanales de calentamiento. No obstante las mayores pérdidas de cobertura de coral ocurren en arrecifes con menos de 8 grados de calentamiento semanal.

Les advertimos que estos resultados no significan que extremos fenómenos de calentamiento sean factores irrelevantes en la mortalidad coralina causada por blanqueamiento y enfermedades como ocurrió visiblemente en Las Isles Vírgenes de los Estados Unidos, Puerto Rico y Floriday Keys entre otros lugares. Además,

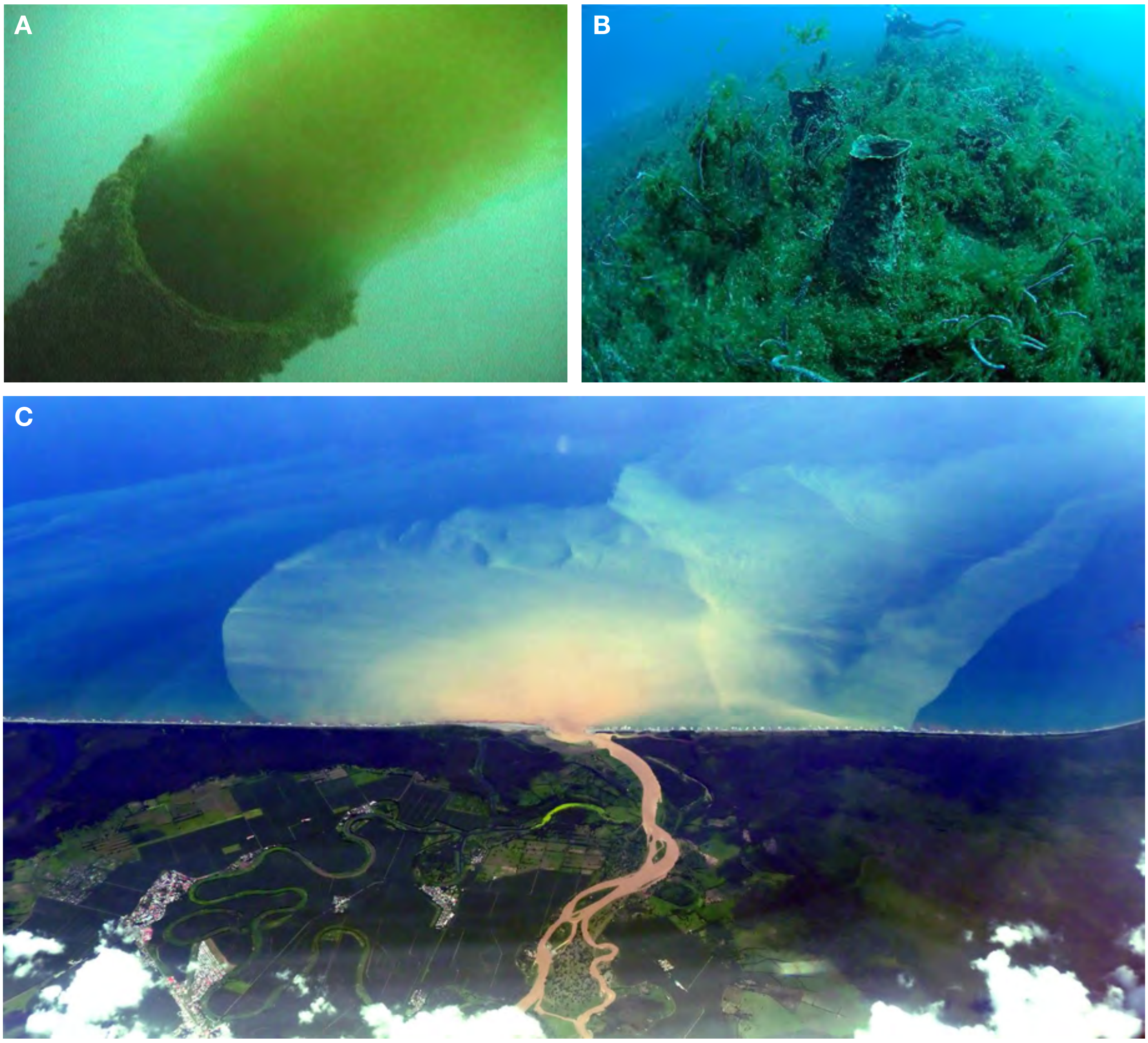

FIGURE 9. Impactos en los arrecifes caribeños producidos por la contaminación costera. (A) Desagüe en Delray Beach, Florida que descarga vertidos que fluyen hacia arrecife de coral 13 millones de galones por día de aguas residuales tratadas. (B) Macroalga cubriendo corales muertos en la proximidad del desagüe (fotos de Steve Spring, Marine Photobank). (C) Masiva descarga fluvial de sedimentos en una desembocadura de la costa mesoamericana en el Mar del Caribe (foto de Malik Naumann, Marine Photobank).

eventos de calentamiento cada vez más severos constituirán un peligro aún mayor para la supervivencia del coral en las próximas décadas. En todo caso nuestros resultados desmienten constantes efectos de fenómenos de extremo calentamiento a nivel regional hasta la fecha e insinúan con firmeza que los estreses locales han sido los motivos principales del declive coralino en el Caribe.

Los efectos potencialmente nocivos de la acidificación del océano no han sido discutidos aquí, debido a la insuficiencia de datos comparativos. Sin embargo, si las tendencias actuales de descenso de pH continúan, la habilidad de los corales y otras especies calcáreas de los arrecifes de formar sus esqueletos se verá cada vez más comprometida.

Especies Invasoras

La expansión del pez león del Pacifico en la Gran Región del Caribe ha devastado las comunidades de pescadores en el Caribe (Fig.12). Y aunque la seriedad de sus consecuencias a largo plazo, son magras comparadas con la introducción del agente patógeno que causó la desaparición de *Diadema antillarum* o los efectos de la enfermedad de la banda blanca on corales acroporidos. La mortalidad en masa de *Diadema* comenzó a tan sólo pocos kilómetros de la entrada del Caribe en el canal de Panamá. Este hecho, acompañado del auge del tránsito de los buques

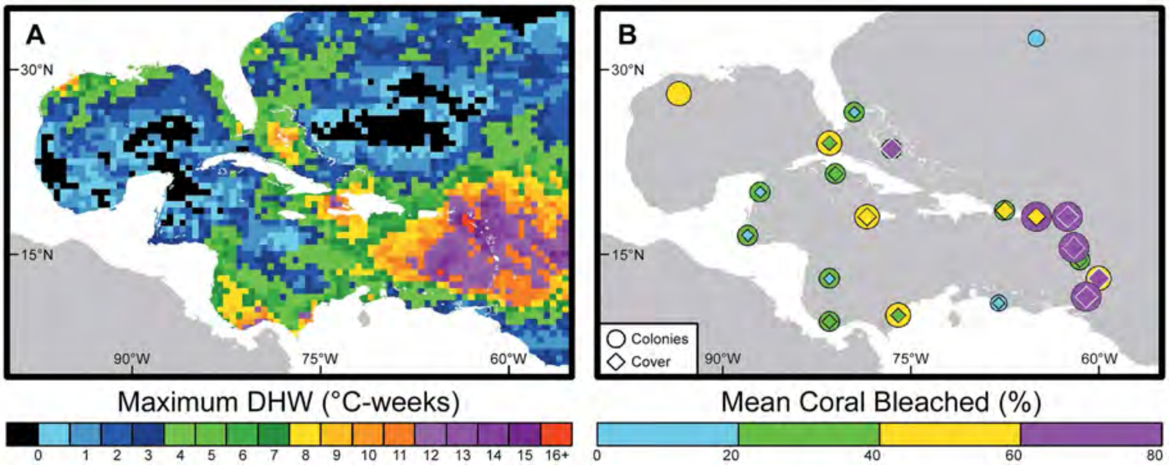

FIGURE 10. Acontecimientos de recalentamiento y asociado blanqueamiento del coral que ocurrieron con mayor severidad en el este del Caribe en 2005. (A) Grados semana de calentamiento tomados del satélite Pathfinder. (B) Informes de la intensidad del blanqueamiento de coral compilados durante observaciones en terreno (transmito por Mark Eakin y colegas).

de cargo en los años 60 y 70 sugiere con firmeza que la enfermedad del erizo Diadema fue introducida por el transporte marítimo. Este también podría ser el motivo de la introducción de las enfermedades coralinas, aunque sus primeras apariciones fueron observadas en toda la Gran región del Caribe.

A causa de su aislamiento durante millones de años y en analogía con el destino de los nativos americanos desde su contacto con los europeos, las especies del Caribe podrían ser excepcionalmente vulnerables al impacto de nuevas enfermedades introducidas en el área. Y así parece ser. No se conoce otro caso similar que se pueda comparar al cataclismo de la *Diadema* y los acroporidos caribeños como la casi desaparición de una especie marina por una enfermedad a lo largo del Océano Pacífico e Indico. Esta interpretación es avalada por el hecho de que no hubo ningún cambio evidente en el medio ambiente en los años 70 que hubiese podido instigar el brote de esta enfermedad. Primordialmente la aparición de estas enfermedades ocurrió muchos años antes de que el primer fenómeno de calentamiento fuera notificado.

Esta hipótesis de la enfermedad de banda blanca podría ser corroborada puesto que el agente patógeno es conocido y disponible para efectuar una secuencia de ADN. Podría ser posible para *Diadema* incluso si el agente patógeno sea desconocido, siempre que se establezcan análisis en especímenes de *Diadema* enteros congelados que murieran a causa de la enfermedad. No se trata solamente de un ejercicio académico: los dos acontecimientos claves en el declive de la mayor parte de los arrecifes del Caribe sigue siendo el mismo misterio que fueron cuando ocurrieron hace 30 años.

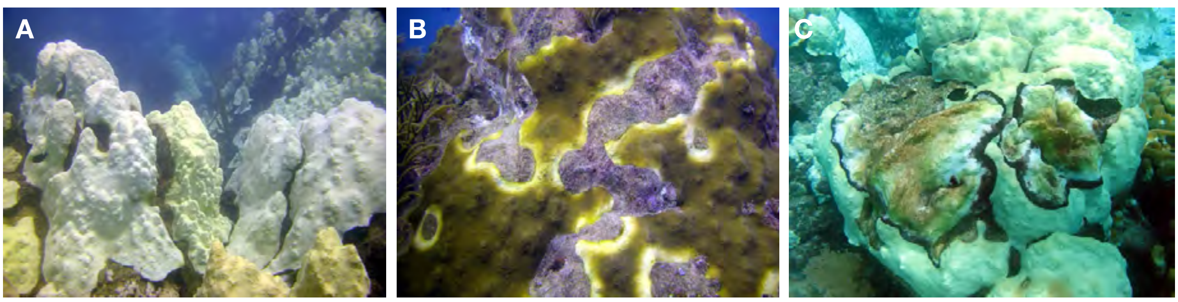

FIGURE 11. Efectos de blanqueamiento y de enfermedades del una vez abundante coral Orbicella faveolata. (A) Corales blanqueados (Turrumote, Puerto Rico, 2005). Mortalidad parcial pero extensiva debido a la infección de (B) la banda amarilla (Turrumote, Puerto Rico, 2005) y (C) banda negra (los Roques Venezuela, 2010). (foto A & B de Ernesto Weil; C de Aldo Cróquer).

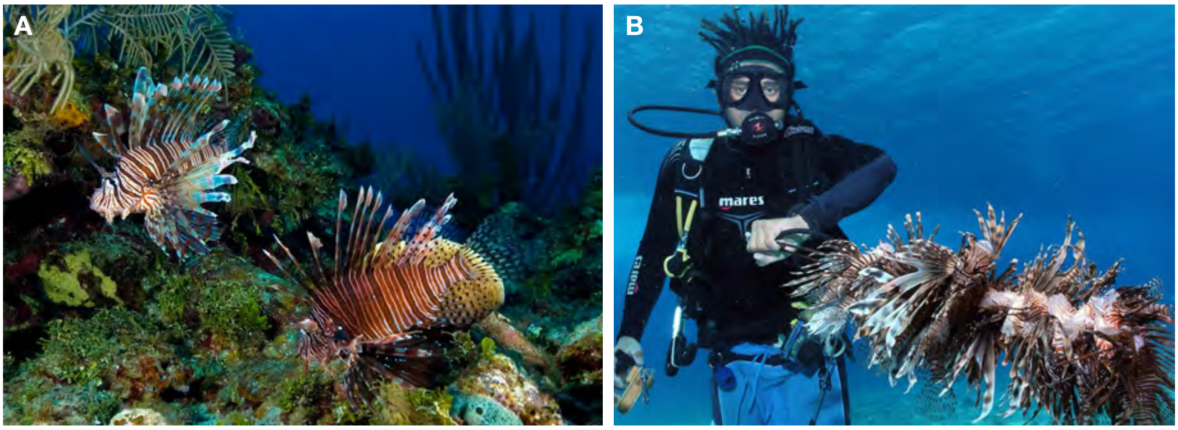

FIGURE 12. Explosión de la población del pez león Pteroi volitans introducido en el Caribe entre 1980 y comienzos del 1990. (A) Los peces león introducidos abundan en los arrecifes de las Islas Cayman (foto cortesía de Niel Van Niekerk con el permiso de IFAS, Universidad de Florida). (B) Peces león arponeados en un intento de controlar la población en Cozumel, México (Foto: Archivos CONANP).

SUMARIO

Tomando como base los datos existentes, las epidemias que decimaron los acroporidos y *Diadema* en los años 70 y 80, el aumento de la población en la forma de un exceso de turistas, y la sobrepesca son los tres mayores causantes del declive de la cobertura del coral en el Caribe desde hace 30 años. La contaminación costera es un factor cada vez más importante pero todavía no hay suficientes datos para deliberar. El calentamiento del océano constituye un peligro inquietante, aunque los efectos de extremo calentamiento hasta ahora han sido sólo localizados y no podían haber sido responsables de las grandes pérdidas de corales en la mayoría del Gran Caribe a mediados de los años 90.

En resumen la degradación de los arrecifes del Caribe se desarrolló en tres etapas distintas.

1. Perdida de acroporidos en masa comienza a mediados de los años 1970 y principios de los 80, causada por una epidemia de banda blanca. Estas pérdidas no están relacionadas con ningún obvio cambio global en el medio ambiente y pudieron ser debidas a la introducción de patógenos asociados con la descarga de aguas de lastre de buques de cargo desde los años 60.
2. Enorme incremento de cobertura de macroalga y declive de la cobertura de coral en la mayoría de las localidades que sufren de sobrepesca tras las mortalidad en masa de *Diadema* en 1983, debido a un patógeno no identificado y probablemente exótico. El cambio de dominación de coral a macroalga culminó a mediados de los 90 y ha continuado por todo el Caribe durante 25 años. Numerosos estudios asocian el incremento de la macroalga con el declive del coral. Macroalga reduce el reclutamiento y crecimiento del coral, produce toxicidad y puede inducir a enfermedades en los corales.
3. El seguimiento del modelo establecido en la segunda etapa ha sido exacerbado a través de una todavía mayor sobrepesca, contaminación costera, expansión del turismo y de los acontecimientos de calentamiento extremo que han sido particularmente graves en el Noreste del Caribe y Florida Keys donde brotes de blanqueamiento extremos, seguidos de brotes de enfermedades han sido los causantes de los mayores declives.

IMPLICACIONES PARA LA GESTIÓN

Nuestros resultados contradicen en gran parte el discurso actual sobre la importancia del calentamiento del océano, de las enfermedades y de los huracanes en los arrecifes coralinos y destaca la importancia de la perspectiva histórica en los medios de gestión y conservación de los arrecifes. Las amenazas del cambio climático y la acidificación de los océanos acechan sombríamente en el horizonte pero los estreses locales tales como la explosión del turismo, sobrepesca teniendo como resultado aumento de macroalga son las causas principales del catastrófico declive de los corales del Caribe.

Esto significa que astutas decisiones y acciones tomadas a nivel local puede ser vitales en el aumento de la capacidad de recuperación y bienestar de los corales y de las comunidades e industrias que dependen de ellos. De este modo cuatro recomendaciones principales se deducen de este estudio:

1. **Adoptar firmes estrategias de gestión de pesca y conservación** que lleven a la restauración del pez Loro incluyendo su inscripción en los anexos pertinentes del protocolo SPAW. La Iniciativa Internacional para los Arrecifes Coralinos (ICRI) en su vigésimo octava Asamblea General en Belize adoptó una Recomendación a este efecto.
2. **Simplificar y estandarizar el monitoreo** de los arrecifes caribeños y publicar los resultados anualmente para facilitar una gestión flexible.
3. **Promover la comunicación y el intercambio de información** entre las autoridades locales para que puedan compartir sus experiencias.
4. **Adoptar e implementar normas y leyes de una manera adaptiva,** siguiendo establecidos indicadores de la salud de los arrecifes, que permitan actuar efectiva y sistemáticamente contra las amenazas que perjudican los arrecifes, especialmente las originadas por la pesca, el turismo y el desarrollo costero.

Somos conscientes de que la acción derivada de estas recomendaciones será sujeta a un debate socioeconómico y político a nivel local y nacional. En cualquier caso las implicaciones de nuestros resultados científicos son irrefutables. Los arrecifes del Caribe y sus recursos están condenados a desaparecer en las próximas décadas si estas medidas no se adoptan y ejecutan con prontitud.

Recomendación adoptada por unanimidad en la 28 ª Asamblea General de ICRI en Belize City, Belice, 17 de octubre de 2013.

Recomendación sobre el declive de la salud de los arrecifes coralinos en el Gran Caribe: la captura de los Peces Loro y otros herbívoros

Recomendación adoptada el 17 de Octubre de 2013 durante la 28 reunión general de la ICRI

Contexto

El informe más reciente de la Red Mundial de Monitoreo de los Arrecifes de Coral (Global Coral Reef Monitoring Network - GCRMN) titulado: *Estado y Tendencia de los Arrecifes de Coral en el Caribe: 1970-2012* es el primer informe que documenta de una manera cuantitativa la tendencia de la salud de los arrecifes de coral tomando como base datos recolectados en el Gran Caribe durante los últimos 43 años.

Los resultados de este estudio muestran claramente que:

- La salud de los arrecifes coralinos depende de un equilibrio ecológico entre los corales y algas en el que la herbivoría juega un papel clave;
- La población de peces loro es un componente crítico de esta herbivor ía particularmente desde el declive del erizo de mar *Diadema* en los años 80;
- Las principales causas de mortalidad de los peces loro es la pesca utilizando fusiles y en particular el uso de trampas.

El estudio además identifica que la sobrepesca de especies herbívoras, el pez loro en particular, ha sido uno de los mayores factores determinantes del declive de los arrecifes en el Caribe, concluyendo que un control efectivo de la sobrepesca a nivel local y nacional puede tener un efecto positivo en la salud de los arrecifes de manera inmediata así como en el futuro.

En ciertas áreas de la región del Caribe (por ejemplo en las Bermudas, en el Parque natural Exuma Cays en las Bahamas y más recientemente en Belize y Bonaire), la activa gestión incluyendo la prohibición de trampas para peces ha contribuido al incremento del número de Peces Loro y consecuentemente al mejoro de la salud del arrecife y su capacidad de recuperación frente al deterioro producido por los huracanes.
Este hecho contrasta con otras áreas del Caribe, donde arrecifes que sufren por la sobrepesca son incapaces de recuperarse frente a los deterioros ocasionados por las tormentas.

Arrecifes sanos han demostrado tener impactos positivos en las economías locales, proporcionando entre otros beneficios la posibilidad de vivir del turismo en lugar de la pesca gracias al incremento de los ingresos del turismo, del número de peces así como de la restauración de servicios ecosistémicos como por ejemplo la protección costera.

Aunque se reconoce que en el Caribe hay varios niveles de dependencia de la pesca y en particular de la captura del pez loro, debido a la evidencia ahora nuestro alcance y en consonancia con la sección sobre la 'gestión integral' del ICRI Marco de Acción (incluyendo la gestión de pesca), la Iniciativa Internacional sobre los Arrecifes Coralinos desea señalar los beneficios de una gestión robusta para proteger los arrecifes de la sobrepesca, y urge a una acción inmediata para proteger el pez loro y otros herbívoros similares de una manera eficaz.

En consecuencia, la Iniciativa Internacional sobre los Arrecifes Coralinos urge a las naciones y a los grupos multilaterales de la región del Caribe a:

1. ***Adoptar*** estrategias de conservación y gestión pesquera que lleven a la restauración del Pez Loro y al equilibrio entre alga y coral característico de los arrecifes de coral sanos;
2. ***Maximizar*** el efecto de estas estrategias de gestión al incorporar las medidas necesarias para sensibilizar, vigilar, sancionar e investigar medios de vida alternativos para aquellos afectados por las restricciones de la captura del pez loro;
3. ***Considerar*** la inscripción del pez loro en los anexos del Protocolo SPAW (Anexo II o III) además de reivindicar en los relevantes foros pesqueros el problema de la hervivoría en el arrecife;
4. ***Educar*** a los grupos indígenas, comunidades locales y otros grupos de interés acerca de los beneficios que estas estrategias producirán en los ecosistemas de los arrecifes coralinos, en el incremento de la pesca y en la economía de la comunidad.

RÉSUMÉ EXECUTIF

JEREMY JACKSON

“L’aspect sans doute le plus frappant de la vie végétale sur un récif corallien en est son absence. Il semble anormal, même à l’observateur non-initié, que les récifs tropicaux, remarquables par leur profusion éblouissante de vie animale, soient ostensiblement dépourvus de plantes.”

Sylvia Earle, 1972

INTRODUCTION

Les remarques initiales de Sylvia Earle sur les récifs coralliens des Caraïbes témoignent d’un monde maintenant oublié. Les récifs coralliens des Caraïbes ont vu leur nombre décliner massivement depuis le début des années 1980 en raison d’un large éventail d’impacts d’origine anthropique tels que l’explosion de la croissance des populations côtières, la surpêche, la pollution côtière, le réchauffement climatique et les espèces envahissantes. Les conséquences de ces impacts incluent l’effondrement généralisé des populations de coraux au profit de macroalgues qui prospèrent, l’essor d’évènements de blanchissement ou de maladies des coraux, et l’incapacité des coraux à se remettre de perturbations naturelles telles que les cyclones.

La sonnette d’alarme a été tirée en 2003, lorsqu’un article paru dans la revue *Science* annonça que le recouvrement de coraux vivants avait décliné de d’une moyenne de 50% dans les années 1970 à seulement 10% de nos jours. Ce déclin spectaculaire fut suivi de près par des évènements de blanchissement sévères et généralisés en 2005, suivis à leur tour par une mortalité en masse de coraux due à des maladies affectant de nombreux récifs dans la région. Des coraux en bonne santé sont un tableau de plus en plus rare dans les récifs intensément étudiés des Keys de Floride, des Iles Vierges et de la Jamaïque. En outre, deux des espèces coralliennes autrefois abondantes, le corail corne d’élan *Acropora palmata* et le corail corne de cerf *Acorpora cervicornis* ont été ajoutées a la Liste des Espèces Menacées des Etats-Unis. Les préoccupations ont été telles que de nombreuses ONG ont décidé d’abandonner leurs efforts pour préserver les récifs des Caraïbes et de les déplacer ailleurs.

C’est dans ce contexte peu encourageant que cette étude a été conduite, dans le but d’évaluer de façon rigoureuse l’étendue des dommages subis par les écosystèmes récifaux dans la Grande Région Caraïbe et d’en déterminer les facteurs responsables. Divers rapports ont suggéré que les récifs de la partie sud des Caraïbes sont en meilleure condition écologique qu’ailleurs, tant par leur couverture corallienne que par le nombre de poissons qui y résident. Si cela est vrai, comprendre pourquoi certains récifs sont en meilleure condition que d’autres pourrait être le premier pas décisif d’une gestion plus efficace, permettant d’améliorer la condition des récifs coralliens dans toute la région Caraïbe.

STRATEGIE ET PORTÉE DU PRÉSENT RAPPORT

Les évaluations caribéennes précédentes avaient analysé les différentes données conjointement, indépendamment de critères les différenciant tels que leur emplacement géographique, l’environnement récifal, la profondeur, les conditions océanographiques etc. Les données provenant de stations lagonaires peu profondes avaient été combinées avec des données provenant de stations de pente externe et d'atolls profonds. La couverture géographique de ces données était inégale, représentant avant tout les sites les plus étudiés, aux données facilement accessibles. Seul le recouvrement corallien total avait été pris en compte, sans tenter de discerner les différences entre espèces de coraux. Aucune tentative n'avait été faite non plus d'intégrer les données existantes sur les macroalgues, les oursins de mer et les poissons, dont l'importance des interactions écologiques avec les coraux est pourtant bien connue.

Nous avons tenté de remédier à ces problèmes de méthode en analysant de façon détaillée l'état et les tendances des communautés récifales sur des sites distincts de toute la Grande Région Caraïbe. Nous avons également compilé des métadonnées essentielles sur la nature de l'environnement récifal, sa profondeur, ainsi que sur l'histoire de la croissance de la population humaine, la pêcherie, les cyclones, les évènements de blanchissement, et les maladies coralliennes pour chaque site. La qualité des informations biologiques varie en fonction des sites, mais des données ont été obtenues dans la mesure du possible sur le recouvrement corallien et de macroalgues, ainsi que sur l'abondance des oursins *Diadema antillarum* (dont le rôle de régulation des communautés algales est déterminant) et la biomasse de poissons, dont le plus important est le poisson-perroquet herbivore.

La majeure partie des données quantitatives des récifs caribéens est inédite ou enfouie dans la littérature grise et les rapports gouvernementaux. Afin d'obtenir ces données, nous avons contacté des centaines de personnes dans tous les pays des Caraïbes par le biais de milliers de courriels, de demandes de données publiées sur des sites internet et de présentations et d'entretiens lors de conférences internationales. Nous avons également correspondu avec les gestionnaires de grands programmes de surveillance de la région. Au final, nous avons réussi à obtenir des données sur les coraux, les macroalgues, les oursins et les poissons provenant de plus de 35000 études quantitatives de 1969 à 2012. Ceci constitue la plus grande quantité de données quantitatives coralliennes jamais compilée, surpassant plusieurs fois celles utilisées auparavant pour ce genre d'étude.

Les données sont reparties sur 90 sites récifaux dans 34 pays (Fig. 1). La majeure partie de ces données provient d'environnements de pente externe et de pâtés coralliens, à des profondeurs allant de 1 à 20 mètres : ils font donc l'objet principal de cette étude. Les données historiques sont rares jusqu'à la mortalité massive de l'oursin autrefois ubiquiste, *Diadama antillarum,* en 1983-84, qui marqua le début de plusieurs programmes de suivi. Les données coralliennes sont prépondérantes, allant de 1970 à aujourd'hui. Les données *Diadema* sont plus limitées car très peu ont été collectées avant que leur mortalité en masse ne réduise leur abondance à néant et que les scientifiques ne réalisent qu'ils étaient perdus. Les données pour les macroalgues sont les plus problématiques du fait du côté aléatoire de leur suivi et taxonomie, et de ce fait, le plus gros des données a dû être retiré de notre analyse. Les données quantitatives pour la taille et l'abondance des poissons coralliens, nécessaires à l'estimation de la biomasse de poissons, n'ont pas pu être obtenues avant 1989 ; mais elles sont riches par la suite.

Les plus longues séries temporelles pour un même récif proviennent de larges quadrats photographiques de 1973 à nos jours pour des stations fixes à Curaçao et Bonaire, avec des séries temporelles plus récentes sur les mêmes îles à partir des années 1990. Des séries temporelles comparables, remontant du début des années 1970 au début des années 1980, sont disponibles pour les Keys de Floride du nord, la Jamaïque, St John et St Croix dans les îles vierges américaines, et Panama. Toutefois, ces données ont été relevées par différents individus à des périodes différentes, et ne sont ainsi pas aussi cohérentes ou parachevées que les données des Caraïbes néerlandaises.

L'intensité de l'échantillonnage varie fortement en termes de temps et d'espace. Nous avons donc segmenté les données en trois intervalles temporels de 12-14 ans, chacun reflétant les évènements écologiques majeurs qu'a vécus la région Caraïbe dans cet espace de temps. Ils sont les suivants:

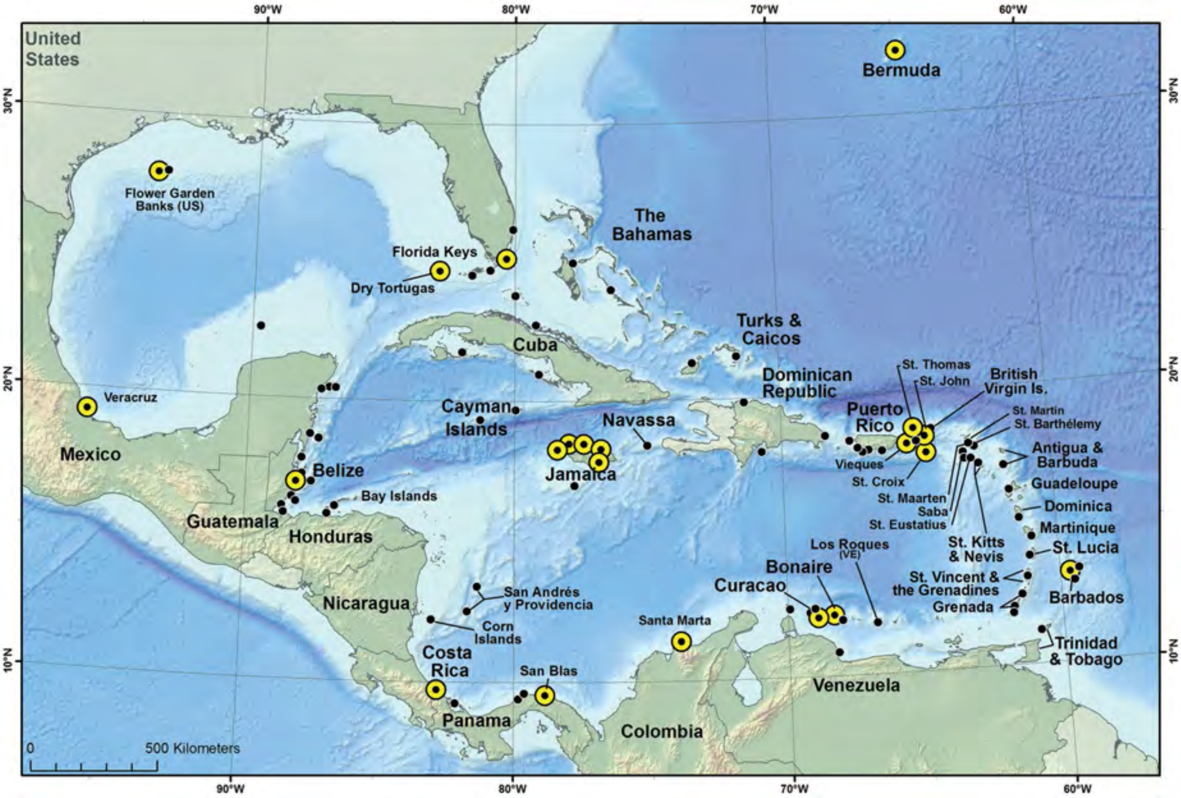

FIGURE 1. Répartition des 90 sites récifaux analysés pour cette étude. Les cercles jaunes indiquent les 21 sites ayant les plus longues séries temporelles pour l'analyse des tendances à long terme de la couverture corallienne.

1. 1970-1983: Intervalle allant des données les plus anciennes jusqu'à la mortalité en masse des oursins autrefois abondants *Diadema antillarum* en 1983 (inclue), comprenant les premiers signalements de maladie de la bande blanche du milieu des années 1970 au début des années 1980.
2. 1984-1998: Depuis juste après l'expiration des *Diadema* jusqu'aux évènements de réchauffement extrêmes de 1998.
3. 1999-2011: L'ère moderne de récifs coralliens sévèrement dégradés.

GRANDES LIGNES DE CHANGEMENT DE 1970 À 2012

La couverture corallienne moyenne pour la Grande Région Caraïbe, si on se fonde sur les données les plus récentes des sites surveillés, est de 16.8% (allant de 2.8 à 53.1%). Prendre en compte la grande variation entre les sites et les fichiers de données réduit cependant cette estimation à 14.3% (+2.0, -1.8). Même cette moyenne établie de façon plus rigoureuse est 43% plus élevée que l'estimation régionale de 2003 de 10% de recouvrement. Néanmoins, la couverture corallienne a décliné dans les trois quarts des sites, avec les plus grandes pertes enregistrées sur les sites surveillés depuis le plus longtemps.

La couverture corallienne moyenne pour les 88 sites pour lesquels nous avions des données a décliné de 34.8%, à 19.1%, à 16.3% lors des trois intervalles, avec une grande disparité entre les sites. A l'opposé, la couverture de macroalgues a augmenté de 7% à 23.6% entre 1984 et 1998, restant stable avec une disparité encore plus grande entre les sites depuis 1984. Les évolutions ont été similaires pour les 21 sites pour lesquels nous avions des données pour les trois intervalles de temps, encerclés dans la Fig. 1. Ces tendances opposées de couvertures coralliennes et macroalgales constituent une permutation importante et persistante de communautés dominées par les coraux à des communautés dominées par des macroalgues, et ce depuis maintenant 25 ans (Figs.2 et 3). Cette propension est réitérée par les analyses d'ordination de composition de communautés benthiques.

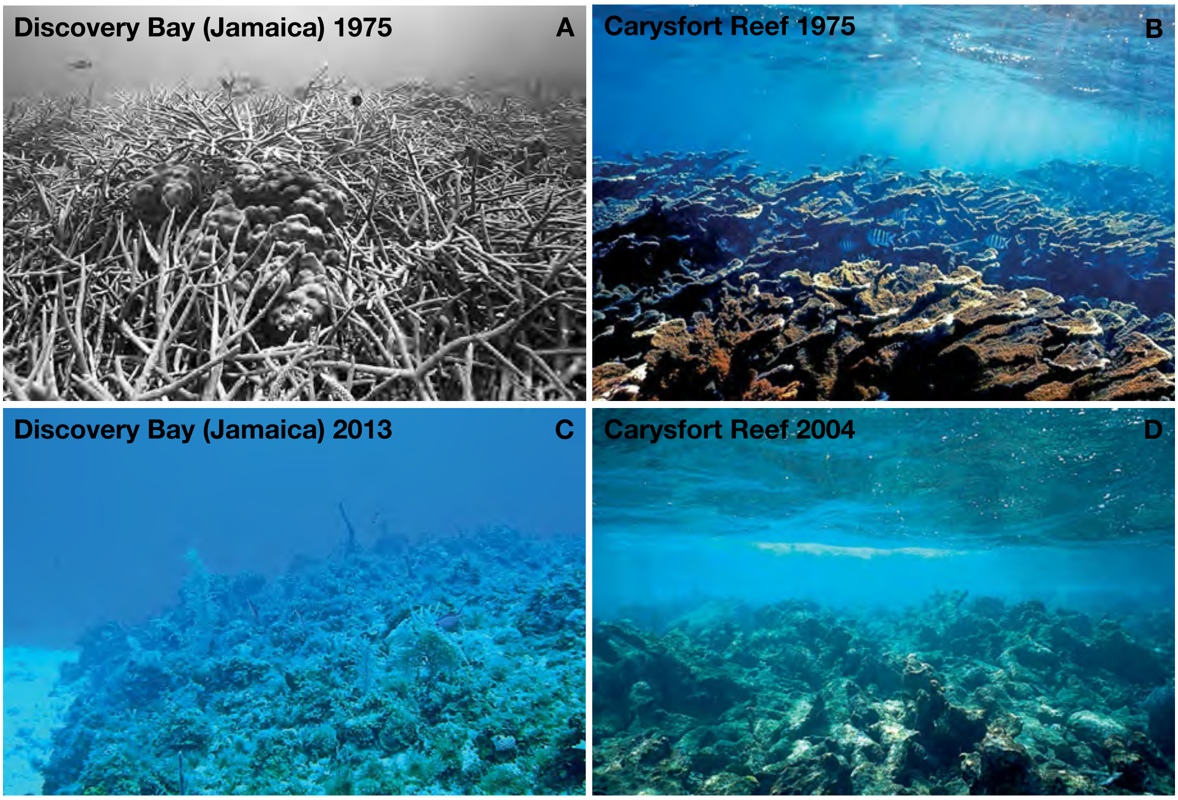

FIGURE 2. Permutation d'une dominance corallienne à une dominance macroalgale sur les récifs de pente externe peu profonds des Keys de Floride du nord et de la côte nord de la Jamaïque. (A) Discovery Bay, Jamaïque en 1975 et (C) au même endroit en 2013. (B) Récif de Carysfort, au sein du Sanctuaire National Marin des Keys de Floride en 1975 et (D) en 2004 (Photos : Phillip Dustan).

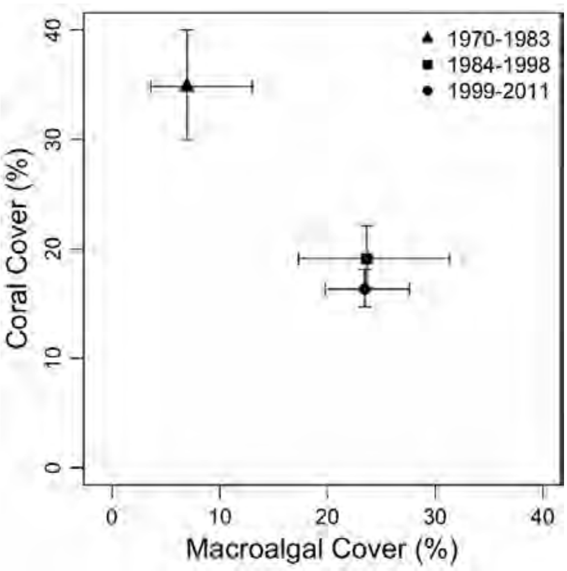

FIGURE 3. Permutations à grande échelle de communautés dominées par les coraux à des communautés dominées par des macroalgues depuis le début des années 1970. Les symboles et les intervalles de confiance représentent les moyennes et écarts-types des trois intervalles de temps, selon un cadre de modélisation mixte prenant en compte la variabilité inhérente aux différents lieux et fichiers de données obtenus.

Les plus grandes mutations dans la couverture corallienne et macroalgale se sont produites entre 1984 et 1998, après quoi il y a eu peu de changement sur la grande majorité des sites, à l'exception de localités particulièrement affectées par les évènements de réchauffement extrêmes de 2005 et 2010. Ceci s'applique également aux *Acropora* corne d'élan et corne de cerf, autrefois abondants, qui ont commencé à décliner dans les années 1960 ; à la mortalité en masse des oursins *Diadema antillarum* en 1983-84 ; et à la surpêche de larges poissons-perroquets dans la plupart des sites du début à la moitié du XX^{ème} siècle. Ainsi, les changements les plus importants et préjudiciables des récifs caribéens se sont-ils produits bien avant que la plupart des experts scientifiques et gestionnaires aient commencé à travailler sur les récifs : un exemple classique du syndrome du changement de référence (*shifting baseline syndrome* en anglais) et un implacable rappel que les problèmes d'aujourd'hui sont seulement le chapitre le plus récent d'une histoire de déclin qui remonte à bien plus longtemps.

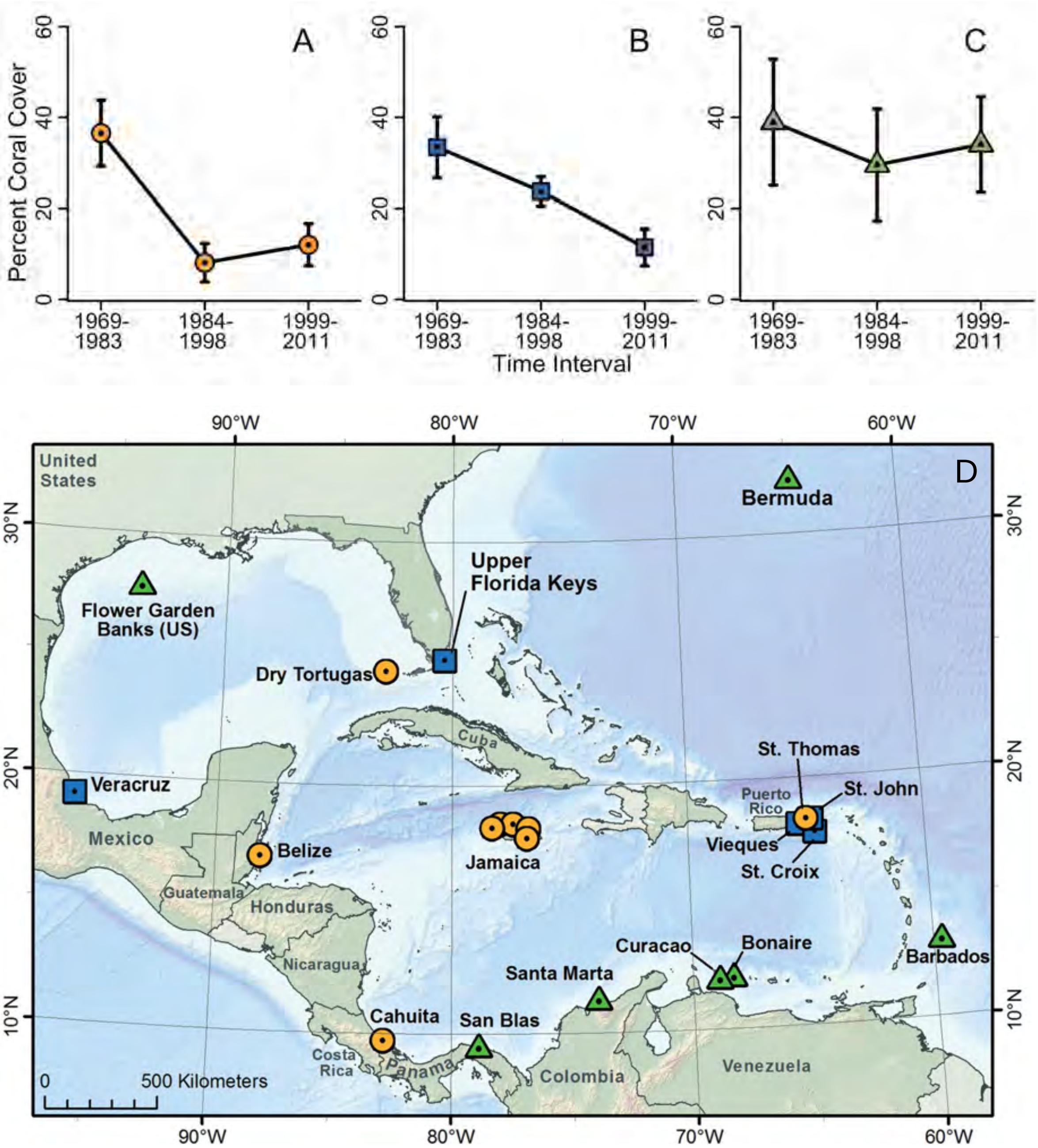

FIGURE 4.Trajets de changement de la couverture corallienne sur les 21 sites cartographiés, regroupés en fonction de la quantité totale de changement sur les trois intervalles de temps et du rythme de changement. (A) Trajectoire en 'batte de hockey' avec un déclin abrupt entre les deux premiers intervalles, suivi de peu ou d'aucun changement. (B) Déclin plus ou moins en continu au cours des trois intervalles. (C) Relative stabilité, avec des changements nets de couverture corallienne beaucoup plus limités.

Au-delà de ce tableau d'ensemble, cependant, les tendances à long terme dans les 21 sites mis en avant dans la Fig. 1 présentent trois trajets étonnamment divergents de changement de la couverture corallienne (Fig. 4). Les trajectoires pour neuf de ces sites se présentent en 'batte de hockey' avec des déclins abrupts de 58-95% entre l'intervalle 1 et l'intervalle 2, pour ensuite rester au même niveau (Fig. 4A). A l'opposé, cinq autres sites présentent un déclin global comparable mais réparti de façon à peu près égale entre intervalles 1 et 2 et intervalles 2 et 3 (Fig. 4B). Le troisième groupe de sept sites présente une beaucoup plus grande stabilité avec des changements globaux (augmentation ou diminution) de seulement 4-35% (Fig. 4C).

FACTEURS DE CHANGEMENT

Les facteurs de la dégradation écologique des récifs caribéens doivent être compris dans le contexte de la situation tout à fait unique des Caraïbes par rapport aux autres mers tropicales. La Mer des Caraïbes pourrait être perçue comme une mer Méditerranée tropicale la plus isolée au monde, géographiquement et océanographiquement parlant. Cette isolement remonte à des dizaines de millions d'années, lors de l'éclatement progressif du paléo-océan Téthys, de l'élargissement de l'Océan Atlantique, suivie de l'isolement de celui-ci du Pacifique Est avec la fermeture de l'isthme de Panama il y a 5.4 à 3.5 millions d'années de cela.

Par conséquent, les biotes récifaux des Caraïbes sont tout à fait singuliers. Les études sur la génétique moléculaire ont ainsi démontré qu'un certain nombre de genres de coraux, une fois combinés avec des taxons du Pacifique, appartiennent à des lignées évolutives exclusivement Atlantiques. En outre, les acroporidés qui représentent plus d'un tiers de la diversité corallienne Indopacifique ne sont représentés que par deux espèces dans les Caraïbes. La diversité taxonomique et la redondance écologique sont minces, et le potentiel de renouvellement par le biais d'autres régions est quasi nul. Les espèces caribéennes n'ont pas non plus d'expérience évolutive de résistance aux espèces envahissantes et aux maladies apparues avec les hommes.

Nous nous sommes concentrés sur les potentiels facteurs de déclin anthropiques pour lesquels les données disponibles nous permettaient d'opérer des comparaisons sérieuses. Chaque facteur a été traité séparément afin de faciliter l'analyse et la discussion, mais ces facteurs sont en réalité inextricablement liés. La maladie corallienne en particulier est un symptôme complexe et peu compris de plusieurs formes de perturbations humaines, et non un facteur direct de changement. C'est pourquoi nous avons traité la maladie en la mettant en relation directe avec différents facteurs, dont l'introduction d'espèces envahissantes, le réchauffement des océans, la pollution côtière, et la surpêche notamment. Globalement, les résultats les plus significatifs (car ayant le plus de données) ont été obtenus pour les effets de l'augmentation de la population humaine, la surpêche, et le réchauffement des océans ; et dans une moindre mesure pour la pollution côtière et les espèces envahissantes.

Une population trop élevée

Le tourisme est le pain quotidien de beaucoup de pays des Caraïbes (Fig.5). Cependant, notre étude démontre que des densités extrêmement élevées de résidents et de touristes sont préjudiciables aux récifs, à moins que les mesures de protection environnementales soient exhaustives et respectées. Malheureusement, c'est rarement le cas. Le nombre de visiteurs par km^2 par an s'échelonne de 110 dans les Bahamas à un ahurissant 25,000 à St Thomas (îles vierges). Tous les sites accueillant plus de la valeur médiane de 1,500 visiteurs par km^2 par an ont moins de la valeur médiane de 14% de couverture corallienne, exception faites des Bermudes avec 39% de couverture et l'île de Grand Cayman avec 31% de couverture. La situation exceptionnelle des Bermudes est sans doute le fruit des régulations environnementales progressives qui ont été mises en place depuis le début des années 1990, et à la

FIGURE 5. Surpopulation: le tourisme de masse dans les Caraïbes. (A) De larges navires de croisière arrivent chaque jour à St Thomas dans les îles vierges (source : Calyponte, Wikipedia). (b) De grands hôtels s'érigent le long de la cote à Cancún au Mexique (source : Foto Propia, Photo de Mauro I. Barea G., Wikipedia). (C) Touristes à South Beach à Miami, Floride (Source: Photo de Marc Averette, Wikipedia).

présence d'infrastructures nécessaires à leur bon fonctionnement. Hormis cette exception, les coûts environnementaux néfastes d'un tourisme galopant semblent inévitables.

Surpêche

La pêche artisanale de subsistance joue un rôle capital dans la plupart des économies caribéennes, mais les conséquences de celle-ci sur les récifs coralliens ont été catastrophiques. La surpêche a mené à un déclin précipité du nombre de poissons herbivores, particulièrement les grands poissons-perroquets, qui sont les brouteurs les plus efficaces des récifs des Caraïbes mais vulnérables à tous les types de techniques de pêche hormis la pêche à la ligne et à l'hameçon.

FIGURE 6. La surpêche a décimé la biomasse et la diversité des poissons dans les Caraïbes. (A – C) Déclin de la composition et de la taille des poissons -trophées dans les Keys de Floride depuis les années 1950 (adapté de McClenachan 2008). (D – F) Les poissons-perroquets furent les brouteurs les plus importants des récifs coralliens des Caraïbes : (D) Poisson perroquet de feu tricolore (Sparisoma viride) pris dans un filet maillant. (E) Récolte d'une journée typique de chasse sous-marine au large du sud-est de Curaçao. (F) Bateaux de pêche à Coco Point, Barbade (Photos : Ayana Elizabeth Johnson).

Néanmoins, les conséquences de la surpêche des poissons-perroquets pour la survie des coraux ont été mal comprises avant l'effondrement de l'oursin *Diadema antillarum,* devenu alors le macro-herbivore le plus important et ubiquiste des récifs caribéens, qui fut décimé par une maladie non-identifiée en 1983-84. Le *Diadema* et la poisson-perroquet sont en forte concurrence alimentaire, et les variations dans leur abondance ont été inversement proportionnelles jusqu'en 1983. Cette relation antinomique nous procure une méthode de comparaison rigoureuse nous permettant d'évaluer les conséquences de la surpêche historique du poisson-perroquet sur la couverture corallienne, en l'absence de données quantitatives sur la biomasse du poisson-perroquet avant 1989.

FIGURE 7. Herbivores autrefois abondants dans les récifs des Caraïbes. (A) Agrégation d'oursins *Diadema antillarum* sur les stations de pente externe de Discovery Bay, Jamaïque a environ 10 mètres de profondeur, un an avant la mortalité de masse de 1983/84 (Photo : Jeremy Jackson). (B) Large banc de poissons-perroquets feu Sparisoma viride au large de la rive sud des Bermudes, où la pêche de ces poissons est interdite (Photo : Philipp Rouja). De telles scènes sont rares, voire absentes dans la grande majorité des récifs des Caraïbes de nos jours.

Notre analyse sur la surpêche provient majoritairement de 16 des 21 récifs encerclés en jaune sur la Fig. 1, pour lesquels nous avons pu obtenir des données quantitatives sur l'abondance des *Diadema* avant leur mortalité en masse en 1983/84, en plus de données sur la couverture corallienne pour les trois intervalles de temps de la Fig. 3. Neuf de ces récifs ont été classifiés comme surexploités (pour les poissons-perroquets) avant 1983, avec des densités de *Diadema* s'échelonnant de 6.9 à 12.4 par m^2, tandis que les sept autres récifs ont été classifiés comme moins exploités avec des densités de seulement 0.5 à 3.8 par m^2. Ce classement rejoint la littérature qualitative sur le sujet. Les récifs dont les poissons-perroquets étaient surexploités avant 1984 ont souffert par la suite d'un déclin de couverture corallienne et d'une augmentation de macroalgues plus élevés que les récifs dont la population de poissons-perroquets était à peu près intacte. Les couvertures coralliennes et macroalgales étaient indépendantes des densités de *Diadema* avant 1984, car le nombre de macroalgues était maintenu à un niveau très bas que ce soit par l'oursin ou par le poisson-perroquet. Tout cela a changé après la disparition du *Diadema* : la couverture corallienne a alors décliné de façon proportionnelle à l'abondance historique de *Diadema*, une tendance qui s'est poursuivie jusqu'à aujourd'hui.

Il existe également de solides preuves expérimentales et de terrain d'effets indirects néfastes de l'augmentation de macroalgues sur la bonne santé des coraux, affectant notamment le recrutement de larves, la survie des coraux juvéniles, et l'incidence de maladies coralliennes. Le recrutement larvaire a décliné rapidement après 1984, du fait au moins partiellement d'une baisse du stock de géniteurs – mais de preuves solides pointent également du doigt une interférence active des macroalgues.

Le recrutement larvaire sur des panneaux expérimentaux à Curaçao a en effet décliné d'un facteur cinq lors d'expériences identiques conduites en 1979-1981 et 1998-2004. Lors des expériences antérieures, les algues calcaires encroûtantes (substrat préféré des larves) couvraient l'ensemble des surfaces supérieures des panneaux tandis que les macroalgues en étaient absentes. Les expériences plus récentes ont vu au contraire les panneaux se recouvrir entièrement de macroalgues.

D'autres expériences montrent que les larves coralliennes évitent activement les substrats où des macroalgues sont présentes, et que les recrues larvaires souffrent d'une mortalité accrue et d'inhibitions de croissance du fait d'interférences physiques avec les macroalgues. Mais la preuve la plus saisissante d'interférences macroalgale vient de l'observation récente d'une augmentation des recrues coralliennes et de survie juvénile dans des récifs où les *Diadema* ont partiellement regagné du terrain, ou des récifs où le nombre poissons-perroquets a augmenté grâce à la mise en place de zones de non-prise.

Les expériences montrent enfin que les macroalgues provoquent une large variété de réponses pathologiques chez les coraux, y compris des maladies virulentes. L'émission de composés allélochimiques toxiques par les macroalgues perturbe également les communautés microbiennes associées aux coraux, causant parfois leur blanchissement ou même leur mort.

La surpêche pourrait avoir également indirectement affecté la capacité des récifs à se remettre des dommages causés par les cyclones alors qu'ils y sont très bien parvenus pendant des millions d'années, sans quoi il n'y aurait pas de récifs aujourd'hui. Au cours des dernières décennies cependant, les coraux de nombreux récifs ont eu de plus eu plus de difficulté à se rétablir à la suite de tempêtes majeures. Nous avons étudié ce changement apparent en se penchant sur les 16 récifs pour lesquels nous disposions de données sur le corail et les *Diadema* avant 1984. La couverture corallienne était indépendante de la probabilité à long-terme de l'occurrence d'un cyclone avant 1984 mais pas après cette date. La surpêche des poissons-perroquets a pu affecter la capacité des coraux à rebondir à la suite de cyclones. Les récifs des Bermudes protégés de la surpêche ont par exemple subi quatre cyclones depuis 1984 sans que cela ait affecté leur couverture corallienne moyenne, tandis que les récifs récemment surpêchés de la Barrière Centrale du Belize ont décliné de 49% après trois cyclones.

Pollution côtière

Des données comparatives limitées sur la transparence de l'eau, basées sur des observations de disque Secchi dans 4 sites CARICOMP (Programme de l'UNESCO sur la productivité côtière et marine des Caraïbes - Caribbean Coastal Marine Productivity Program en anglais), montrent que la qualité de l'eau décline rapidement dans les zones de développements agricoles et côtiers non règlementés. La transparence de l'eau a par exemple décliné de façon drastique sur 20 ans sur le banc de sable Carrie Bow au Belize, du fait de larges développements agricoles et côtiers du Guatemala au Honduras comme l'illustre la Fig. 9C. Des tendances similaires ont été observées à La Parguera sur la côte ouest de Puerto Rico. En revanche, la qualité de l'eau s'est dans le même temps améliorée dans les Bermudes.

FIGURE 8. Dense multiplication de macroalgues. On peut observer les extrémités de branches des coraux Porites survivants à travers la canopée algale dans le coin supérieur droit de la photo ; et des branches de Porites et de Acropora cervicornis précédemment envahis et maintenant morts en bas à gauche de la photo (Dry Tortugas, 2000, Photo : Mark Chiappone).

Les maladies coralliennes ont été rattachées à une pollution organique excessive, mais les données sur le sujet sont sporadiques et limitées dans leur portée. De façon générale, il y a un besoin urgent de mettre en place un suivi plus systématique et intégral de la qualité de l'eau dans la Grande Caraïbe.

Réchauffement des océans

Nos premières analyses se fondaient sur la compilation Reefbase d'évènements de blanchissement extrêmes, qui ne démontraient pas de de lien significatif entre le nombre d'évènement extrêmes par localité et sa couverture corallienne dans des sites de la Grande Caraïbe, de Golfe de Mexico et des Bermudes. Cependant, du fait de la subjectivité inhérente aux évaluations de

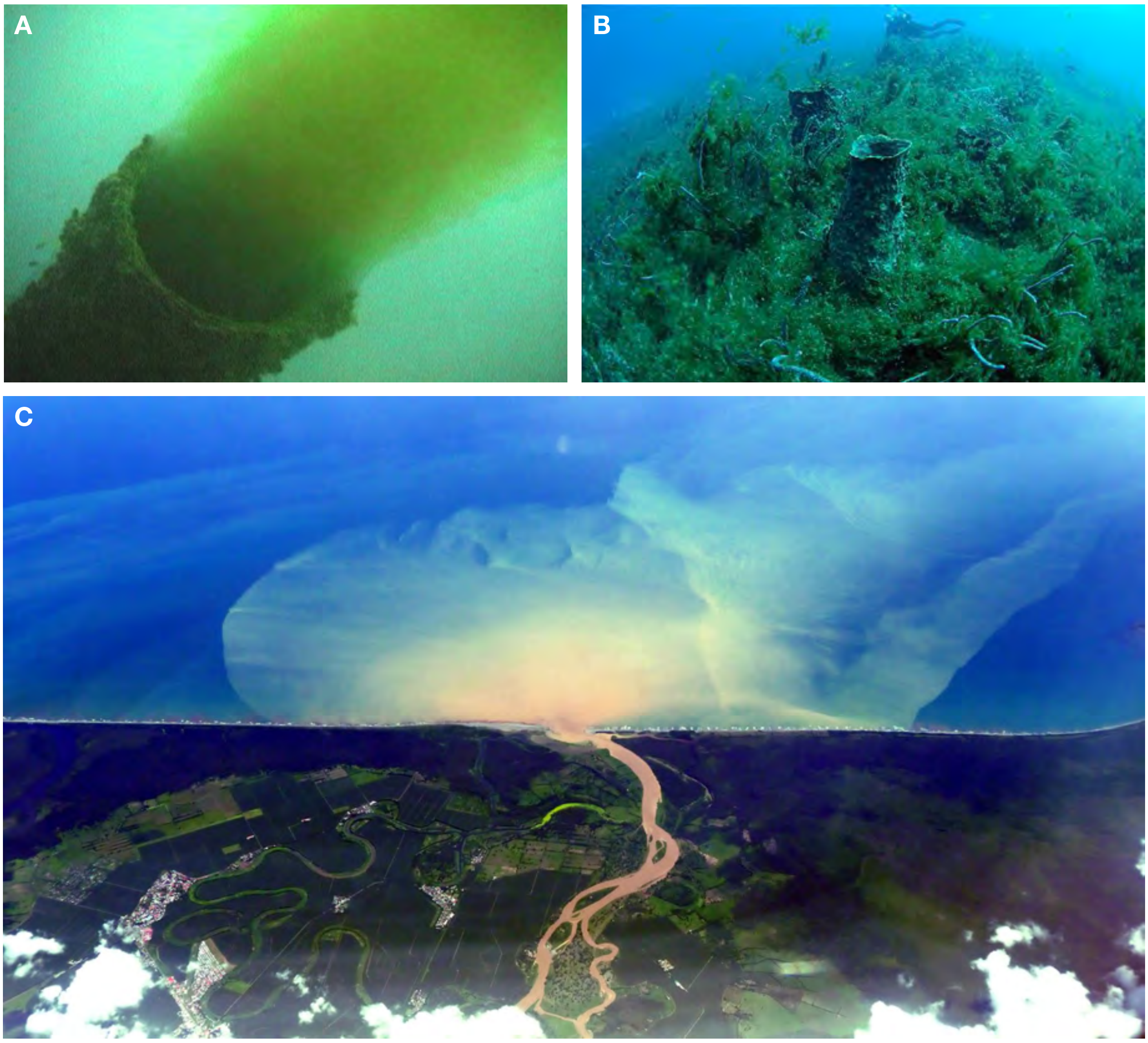

FIGURE 9. Impacts de la pollution côtière sur les récifs des Caraïbes. (A) Déversement d'eaux usées à Delray Beach en Floride, déchargeant 13 millions de gallons par jour d'eaux usées traitées en amont d'un récif corallien. (B) Macroalgues tapissant des coraux morts près d'une embouchure d'égout (Photos de Steve Spring, Marine Photobank). (C) Apport massif de charge sédimentaire à l'embouchure d'une rivière se déversant dans la Mer des Caraïbes, au large de la côte mésoaméricaine (Photo: Malik Naumann, Marine Photobank).

blanchissement, nous avons par la suite obtenu du programme Surveillance Récifs Coralliens (Coral Reef Watch) de NOAA les données degrés-semaines de réchauffement (appelés DHW pour 'Degree Heating Weeks') pour les 88 sites ayant de la couverture corallienne.

Nous avons ensuite utilisé ces données pour évaluer les effets des évènements de réchauffement extrêmes de 1998, 2005 et 2010 sur la couverture corallienne en calculant son changement proportionnel pendant les deux années suivant chaque évènement par rapport aux deux années précèdent l'évènement. Nous avons représenté ce changement proportionnel en fonction du nombre de degrés-semaines de réchauffement pour chaque station. Il existe une corrélation négative faible, mais statistiquement insignifiante, entre les pertes en couverture corallienne et le nombre de DHW, indépendamment du fait que les données soient analysées de façon séparée pour chaque évènement ou qu'elles soient combinées ; ou selon que nous incluions toutes les stations ou que nous limitions l'analyse aux seuls stations ayant subi au moins 8 DHW. De plus, les plus grandes pertes en couverture corallienne sont survenues dans les stations ayant moins de 8 DHW.

Nous tenons ici à avertir le lecteur que nos résultats ne signifient nullement que les évènements de réchauffement extrêmes sont des facteurs

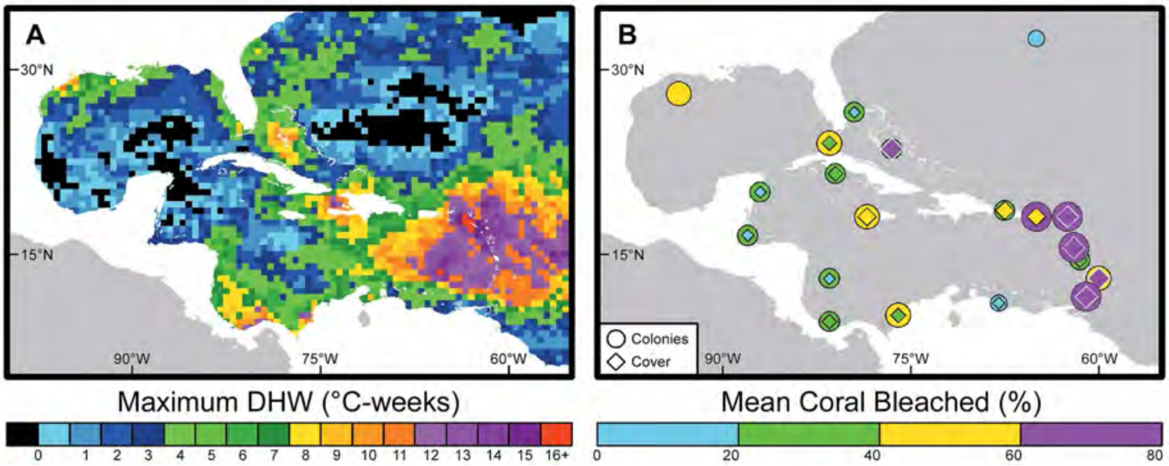

FIGURE 10. Evènements de réchauffement extrêmes et blanchissement des coraux associés ayant touché l'est des Caraïbes en 2005. (A) Degrés-semaines de réchauffement à partir d'observations satellite Pathfinder. (B) Intensité des évènements de blanchissement de coraux à partir d'observations observés de terrain (transmis gracieusement par Mark Eakin et collègues.)

insignifiants de la moralité corallienne du fait de blanchissement et de maladies ; ils l'ont en effet clairement été dans les îles vierges américaines, Puerto Rico, les Keys de Floride, et ailleurs. De plus, ces évènements de réchauffement de plus en plus sévères constituent une menace indéniable pour la survie des coraux dans les décennies à venir. Mais nos résultats contredisent tout effet régionalement cohérent des évènements de réchauffement extrêmes, et impliquent de manière forte que ce sont tout d'abord les facteurs de stress locaux qui ont été jusqu'à présent majoritairement responsables du déclin des coraux dans les Caraïbes.

Les effets potentiellement délétères de l'acidification des océans n'ont pas été traités ici du fait d'un manque de données comparatives. Cependant, si les tendances actuelles de la diminution du pH continuent, la capacité des coraux et des autres espèces récifales calcifiantes à construire des squelettes va être de plus en plus compromise.

Espèces envahissantes

L'explosion du poisson-lion exotique, provenant du Pacifique dans toute la région Caraïbe (Fig. 12) a fait des ravages dans les communautés de pêcheurs. Mais, aussi graves que puissent en être les conséquences à long terme, elles restent falotes si on les compare à l'introduction de l'agent pathogène non identifié responsable de la décimation du *Diadema antillarum,* ou aux effets de la maladie de la bande blanche sur les acroporidés. La mortalité en masse des *Diadema* s'est manifesté a seulement quelques kilomètres de l'entrée du canal de Panama. Ce fait, associé à l'ampleur de l'augmentation du trafic de vraquiers dans les années 1960 et 1970, suggère fortement que la maladie du *Diadema* a été introduite par le transport maritime. Cela pourrait également être vrai des maladies coralliennes, quoique leur premières occurrences aient été signalées dans toute la région Caraïbe.

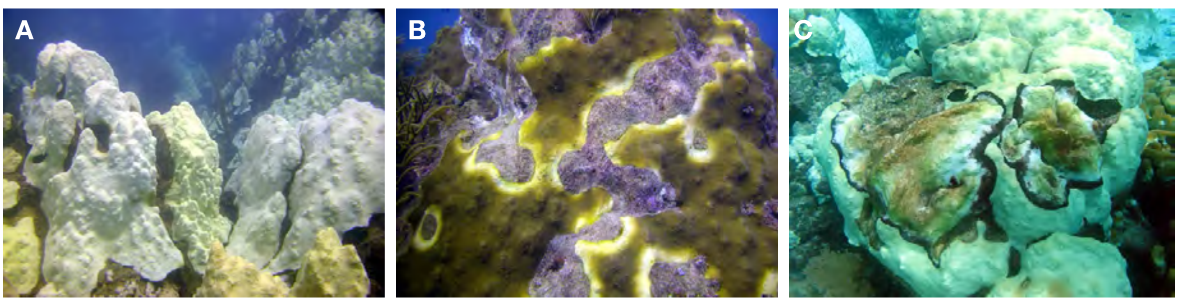

FIGURE 11.Effets du blanchissement et des maladies coralliennes sur l'espèce Orbicella faveolata autrefois abondante. (A) Coraux blanchis (Turrumote, Puerto Rico, 2005). Mortalité partiale extensive d'une colonie corallienne causée par la maladie de la bande jaune (Turrumote, Puerto Rico, 2005) et (C) la maladie de la bande noire (Los Roques Venezuela, 2010). (Photo A & B de Ernesto Weil; C de Aldo Cróquer).

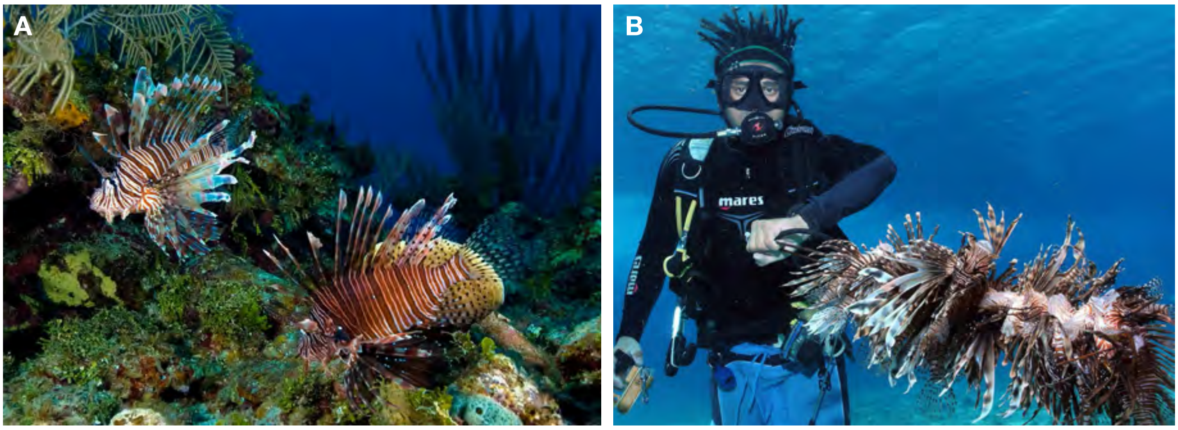

FIGURE 12. Explosion de la population du poisson-lion *Pterois volitans* introduit dans les Caraïbes entre les années 1980 et le début des années 1990. (A) Les poissons-lion envahissants abondent dans les récifs des îles Caïman (Photo : Niel Van Niekerk, avec la permission de IFAS, Université de Floride). (B) 'Brochette' de poissons-lions dans le cadre d'efforts de contrôle des populations dans le parc national de Cozumel, Mexique (Photo : Archives CONANP).

En raison de leur isolement depuis des millions d'années, et en analogie avec le destin des peuples indigènes américains à la suite de leur premier contact avec les européens, les espèces caribeennes devraient être particulièrement vulnérables à l'introduction de nouvelles maladies. Et cela semble être effectivement le cas. Nous ne connaissons pas d'autres exemples de quasi-élimination d'une espèce marine en raison de maladie dans toute l'étendue des océans Indien et Pacifique, qui puisse être comparé à la détérioration des *Diadema* et des acroporidés caribéens. Cette interprétation est également conciliable avec l'absence marquée de changement environnemental majeur dans les années 1970 qui aurait pu expliquer l'apparition de maladies. Enfin et surtout, ces maladies ont émergé bien des années avant que les premiers évènements de réchauffement extrêmes ne soient signalés.

Il serait possible de tester cette hypothèse d'espèce introduite pour la maladie de la bande blanche puisque le pathogène est connu et que le séquençage d'ADN est disponible. Il pourrait même être possible de faire la même chose pour le *Diadema,* même si le pathogène n'est pas connu, en conduisant des analyses génétiques de spécimens entiers de *Diadema* morts de la maladie. Ce n'est pas un exercice entièrement académique : ces deux évènements charnière dans le déclin de la plupart des récifs caribéens restent tout aussi mystérieux de nos jours qu'ils l'étaient lorsqu'ils sont apparus il y plus de 30 ans.

RÉSUMÉ

Les épidémies affectants les acroporidés et les oursins *Diadema* dans les années 1970 et 1980, la surpopulation (sous la forme d'excès de touristes), et la surpêche sont les trois meilleurs prédicteurs du déclin de la couverture corallienne des Caraïbes au cours des 30 dernières années (voire plus), si l'on en croit les données disponibles. La pollution côtière est incontestablement un facteur de plus en plus important, mais les données sont encore trop limitées pour établir cela avec certitude. Le réchauffement des océans constitue une menace fort inquiétante, mais jusqu'à présent les évènements de réchauffement extrêmes n'ont eu des effets que localisés et n'ont pas pu être responsables des pertes les plus grandes des coraux caribéens, survenues presque partout dans la grande région Caraïbe au début et milieu des années 1990.

En résumé, la dégradation des récifs caribéens s'est déroulée en trois phases distinctes :

1. Des pertes massives d'acroporidés commençant au milieu des années 1970 et durant jusqu'au début des années 1980, du fait de la maladie de la bande blanche. Ces pertes ne peuvent être reliées à aucun changement environnemental global évident, et pourraient être le fait de l'augmentation de pathogènes associée à l'énorme augmentation de décharge d'eaux de ballast par les vraquiers dans les années 1960.

2. Des fortes augmentations de couverture macroalgales et de fortes réductions de couverture coralliennes dans la plupart des sites surexploités à la suite de la mortalité en masse des *Diadema* en 1983, causée par un pathogène non-identifié probablement introduit. Ce processus de mutation d'une dominance corallienne à macroalgale a atteint son sommet dans les plupart des sites au milieu des années 1990, et a persisté depuis dans toute la région et ce depuis 25 ans. De nombreuses expériences ont démontré un lien entre l'augmentation des macroalgues et le déclin des coraux. Les macroalgues ont un effet néfaste sur le recrutement et la croissance des coraux, et sont souvent toxiques pour eux, pouvant prodiguer un terrain favorable à la prolifération de maladies coralliennes.
3. La continuation des tendances établies pendant la phase 2, exacerbées par l'intensification de la surpêche, la pollution côtière, l'explosion du tourisme, et les évènements de réchauffement extrêmes qui, combinés, ont été particulièrement destructeurs dans la partie nord-est des Caraïbes et les keys de Floride, où des évènements de blanchissement extrême, suivi d'une prolifération de maladies coralliennes, a causé les déclins les plus graves.

IMPLICATIONS DE GESTION

Nos résultats contredisent en grande partie le discours actuel sur l'importance du réchauffement des océans, de la maladie, et des cyclones sur les récifs coralliens, et soulignent l'importance de la mise en perspective historique pour la gestion et la conservation des récifs coralliens. Les menaces posées par le changement climatique et l'acidification des océans se profilent à l'horizon de manière de plus en plus inquiétante ; mais les facteurs de stress locaux, notamment l'explosion du tourisme, la surpêche et l'augmentation de macroalgues en résultant, ont été jusqu'à présent les principaux facteurs responsables de déclin catastrophique des coraux des Caraïbes.

Cela signifie que des décisions et actions locales intelligentes pourraient faire toute la différence pour accroitre la résilience et le bien-être des récifs coralliens des Caraïbes, et de fait, des communautés et industries qui en dépendent. Quatre recommandations majeures se dégagent ainsi de ce rapport :

1. **Adopter des stratégies robustes de conservation et de gestion des pêches**, conduisant à la restauration des populations de poissons-perroquets, y compris l'inscription du poisson-perroquet dans les annexes pertinentes du Protocole SPAW. Une recommandation a été adoptée à cet effet par l'Initiative Internationale pour les Récifs Coralliens (ICRI) lors de sa 28ème Assemblée Générale en Octobre 2013 au Belize (voir encadré).
2. **Simplifier et standardiser le suivi des récifs caribéens**, et en publier les résultats de façon annuelle pour faciliter la gestion adaptative.
3. **Promouvoir la communication et l'échange d'informations** pour que les autorités locales puissent bénéficier de l'expérience des autres.
4. Adopter et mettre en œuvre une législation et des régulations adaptatives permettant de prendre des mesures sur les menaces pesant sur les récifs coralliens de manière systématique, particulièrement celles posées par la pêche, le tourisme et le développement côtier; et fondés sur des indicateurs de santé des récifs.

Nous sommes conscients que la mise en œuvre de ces recommandations fera l'objet d'un débat politique et socioéconomique au niveau national et local. Mais les implications de nos résultats scientifiques sont sans équivoque *: les récifs coralliens des Caraïbes et les ressources qui en dérivent sont vouées à disparaitre dans les décennies à venir si ces mesures ne sont pas adoptées et mises en œuvre sans délai*.

Encadré 1: Recommandation adoptée à l'unanimité lors de la 28ème Assemblée Générale de l'ICRI a Belize City, Belize, le 17 Octobre 2013.

RECOMMANDATION sur le déclin de la santé des récifs coralliens dans la Grande Région Caraïbe: la prise de poissons-perroquets et autres herbivores coralliens

La présente recommandation a été adoptée par les members de l'ICRI
le 17 Octobre 2013, lors de la 28ème Assemblée Générale de l'ICRI (Belize City)

Contexte

Le dernier rapport du Réseau Global sur le Suivi des Récifs Coralliens (Global Coral Reef Monitoring Network - GCRMN), intitulé: *Etat et Tendances des Récifs Coralliens des Caraïbes: 1970-2012* est le premier rapport documentant les tendances de l'état de santé des récifs coralliens de manière quantitative, en se fondant sur des données collectées au cours des 43 années précédentes dans toute la Grande Région Caraïbe.

Les résultats de cette étude démontrent clairement que :

- La bonne santé des récifs coralliens nécessite un équilibre écologique entre coraux et algues, au sein duquel l'herbivorie est un élément clé ;
- Les populations de poissons-perroquets sont une composante critique de cette herbivorie, particulièrement depuis le déclin des oursins *Diadema* au début des années 1980 ;
- Les causes principales de la mortalité des poissons-perroquets sont l'utilisation de techniques de pêches telles que le fusil sous-marin et, en particulier, la pêche au casier ou à la nasse.

Le rapport identifie en outre que la surpêche des espèces herbivores, particulièrement le poisson-perroquet, a été jusqu'à présent l'un des facteurs principaux du déclin des récifs Caribéens, concluant ainsi que des mesures de gestion aux niveaux national et local peuvent avoir un effet positif direct sur leur santé maintenant et pour les années à venir.

Dans certaines zones de la région Caraïbe (par exemple les Bermudes et le Parc Terrestre et Marin des Bancs de sable Exuma dans les Bahamas, et plus récemment au Belize et à Bonaire), des mesures de gestion proactives, telles que l'interdiction des casiers, ont conduit à une augmentation du nombre de poissons-perroquets et a une amélioration conséquente de la santé des récifs et de leur résilience aux perturbations, y compris celles provoqués par les ouragans. Ceci contraste avec d'autres régions des Caraïbes, où certains récifs fortement exploités peinent à se remettre des dégâts occasionnés par ceux-ci.

Des récifs en bonne santé ont démontré avoir des retombées positives sur les économies locales, fournissant entre autres la possibilité de moyens de subsistance alternatifs à la pêche grâce à l'augmentation des recettes du tourisme et du nombre de poissons; et la restauration de services écosystémiques prodigués par les récifs tels que la protection côtière.

Il est reconnu que le degré de dépendance des communautés côtières à la pêche en général, et à la prise de poissons-perroquets en particulier, varie considérablement au sein de la région Caraïbe. Cependant, au vu des données maintenant disponibles, et conformément à la section 'gestion intégrée' du Cadre d'Action de l'ICRI (qui comprend la gestion des pêches), l'Initiative Internationale pour les Récifs Coralliens tient à souligner les bénéfices de mesures de gestion robustes pour protéger les récifs de la surpêche, et exhorte à une prise de mesures immédiate pour protéger les poissons-perroquets et autres herbivores similaires de manière efficace.

En conséquence, l'Initiative Internationale pour les Récifs Coralliens exhorte les nations et les groupes multilatéraux de la région des Caraïbes à:

1. ***Adopter*** des stratégies de conservation et de gestion des pêches qui conduisent à la restauration des populations de poissons-perroquets, rétablissant ainsi l'équilibre entre algues et coraux caractéristique des récifs coralliens en bonne santé ;
2. ***Maximiser*** l'effet de ces stratégies de gestion en y associant les ressources nécessaires à la mise en place de programmes de sensibilisation, de surveillance, et de mise en œuvre, et en examinant des moyens de subsistance alternatifs pour les personnes touchées par les restrictions sur la prise du poisson-perroquet ;
3. ***Envisager*** l'inscription du poisson-perroquet dans les annexes du Protocole SPAW (annexe II ou III), en plus de soulever le problème de l'herbivorie récifale lors des forums des pêches régionaux ;
4. ***Engager*** les communautés autochtones et locales et autres parties prenantes en leur faisant prendre conscience des bénéfices tirés de telles stratégies pour les écosystèmes coralliens, la reconstitution des stocks halieutiques et l'économie locale.

PART I: OVERVIEW AND SYNTHESIS FOR THE WIDER CARIBBEAN REGION

Jeremy BC Jackson, Mary K Donovan, Katie L Cramer, Vivian YY Lam, Rolf PM Bak, Iliana Chollett, Sean R Connolly, Jorge Cortés, Phil Dustan, C. Mark Eakin, Alan M Friedlander, Benjamin J Greenstein, Scott F Heron, Terry Hughes, Jeff Miller, Peter Mumby, John M Pandolfi, Caroline S Rogers, Robert Steneck, Ernesto Weil, Jahson B Alemu I, William S Alevizon, Jesús Ernesto Arias-González, Andrea Atkinson, David L Ballantine, Carolina Bastidas, Claude Bouchon, Yolande Bouchon-Navaro, Steve Box, Angelique Brathwaite, John F Bruno, Chris Caldow, Robert C Carpenter, Bernadette H Charpentier, Billy Causey, Mark Chiappone, Rodolfo Claro, Aldo Cróquer, Adolphe O Debrot, Peter Edmunds, Douglas Fenner, Ana Fonseca, Marcia C Ford, Kirah Forman, Graham E Forrester, Joaquín R Garza-Pérez, Peter MH Gayle, Gabriel D Grimsditch, Hector M Guzmán, Alastair R Harborne, Marah J Hardt, Mark Hixon, Joshua Idjadi, Walter Jaap, Christopher FG Jeffrey, Ayana Elizabeth Johnson, Eric Jordán-Dahlgren, Karen Koltes, Judith C Lang, Yossi Loya, Isaias Majil, Carrie Manfrino, Jean-Philippe Maréchal, Croy MR McCoy, Melanie D McField, Thaddeus Murdoch, Ivan Nagelkerken, Richard Nemeth, Maggy M Nugues, Hazel A Oxenford, Gustavo Paredes, Joanna M Pitt, Nicholas VC Polunin, Pedro Portillo, Héctor Bonilla Reyes, Rosa E Rodríguez-Martínez, Alberto Rodriguez-Ramirez, Benjamin I Ruttenberg, Rob Ruzicka, Stuart Sandin, Myra J Shulman, Struan R Smith, Tyler B Smith, Brigitte Sommer, Chris Stallings, Rubén E Torres, John W Tunnell, Jr., Mark JA Vermeij, Ivor D Williams, Jon D Witman

Caribbean coral reef ecosystems are severely degraded due to human overfishing, pollution, climate change, and the synergies among them. Coral cover has reportedly declined by more than 80% since the 1970s (Fig. 1), virtually all the large fishes, sharks, and turtles are gone (Fig. 2), and the threats of global climate change loom increasingly ominously for the future (Fig. 3) (Hughes 1994; Jackson 1997; Aronson and Precht 2001; Jackson et al. 2001; Gardner et al. 2003; Pandolfi et al. 2003; McClenachan 2008; Eakin et al. 2010). The severity of the situation has raised serious questions about the future of Caribbean reefs and indeed reefs worldwide (Knowlton 2001; Hughes et al. 2003, 2010; Bellwood et al. 2004; Pandolfi et al. 2005; Hoegh-Guldberg et al. 2007; Hughes et al. 2010).

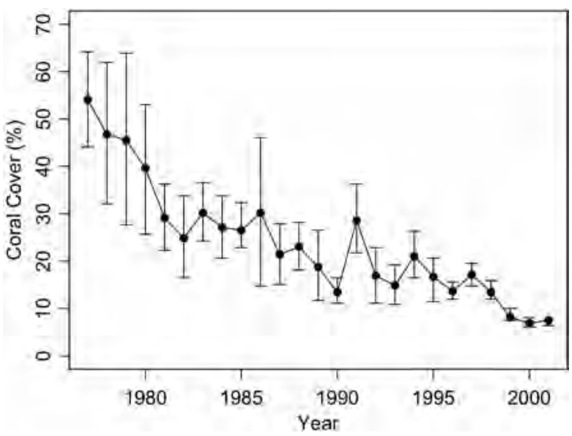

FIGURE 1. Estimates of annual percent coral cover for the entire wider Caribbean region (re-plotted from Gardner et al. 2003).

Nevertheless, there are reasons for hope based upon the remarkable abundance and resilience of corals at some remote Pacific island reefs that are protected from local impacts of overfishing and pollution (Friedlander and DeMartini 2002; Knowlton and Jackson 2008; Sandin et al. 2008a; Pandolfi et al. 2011; Gilmour et al. 2013). Despite increased warming and coral bleaching throughout the Pacific, these reefs have recovered from past episodes of bleaching and still support extraordinarily abundant and resilient populations of fishes and corals.

There are also reports of considerable variability in the condition of Caribbean reefs (Kramer 2003; Newman et al. 2006; Schutte et al. 2010) that is

FIGURE 2. Overfishing significantly reduced fish biomass and diversity in the Caribbean. (A - C) Decline in the composition and size of coral reef trophy fish in the Florida Keys since the 1950s (modified from McClenachan 2008). (D - F) Parrotfish were the most important grazers on Caribbean reefs: (D) Stoplight parrotfish (*Sparisoma viride*) caught in a gill net. (E) A typical day of spearfishing off southeast Curaçao. (F) Fishing boats at Barbuda's Coco Point (Photos by Ayana Elizabeth Johnson).

obscured by plotting a single line for reef condition over time, regardless of location, reef type, depth, environmental conditions, and human impact as in Fig. 1 (Gardner et al. 2003). For example, live coral cover is less than the reported Caribbean average of 10% in the Florida Keys (Dustan 2003; DuPont et al. 2008) and the US Virgin Islands (Edmunds 2002; Rogers and Miller 2006; Miller et al. 2009), but commonly exceeds 30% on reefs in Curaçao and Bonaire (Bak et al. 2005; Sandin et al. 2008b; Steneck et al. 2011; Vermeij 2012), the Flower Gardens Banks (Aronson et al. 2005; Hickerson et al. 2008), and Bermuda (Murdoch et al. 2008; Smith et al. 2013).

The causes of these regional differences are poorly understood despite their obvious significance for conservation and management. Caribbean reefs with the highest coral cover tend to be characterized by little land-based pollution; some degree of fisheries regulations and enforcement; lower frequencies

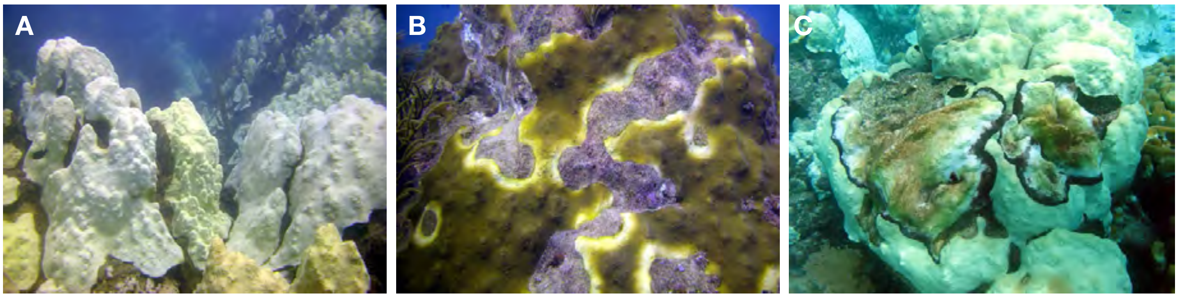

FIGURE 3. Effects of coral bleaching and disease on the formerly abundant coral *Orbicella faveolata*. (A) Bleached corals (Turrumote, Puerto Rico, 2005). Extensive partial colony mortality due to infection by (B) Yellow Band Disease (Turrumote, Puerto Rico, 2005) and (C) Black Band Disease (Los Roques Venezuela, 2010). (Photos A and B by Ernesto Weil; C by Aldo Cróquer).

of hurricanes, coral bleaching, and disease; and moderate economic prosperity. However, these apparent trends have not been rigorously investigated.

There is also a fundamental methodological problem in the common failure to distinguish between the potential anthropogenic drivers of reef degradation such as human overpopulation, overfishing, coastal pollution, introductions of alien species, and ocean warming and acidification due to the burning of fossil fuels, versus their effects such as losses of corals and increases in macroalgae, coral bleaching, and disease (Hughes et al. 2010). This confusion is compounded by scientific provinciality. Most scientists study reefs in a geographically limited area and then project their results to the entire Caribbean. This tendency for over generalization is further compounded by an overall lack of comparative data to address multiple factors in a unified analysis (Hughes et al. 2010).

New insights in science commonly emerge from examining exceptions to general patterns rather than the norms (Knowlton and Jackson 2008). Thus the major goal of this report is to document the variable condition of Caribbean reefs as a means towards better understanding of the factors driving Caribbean reef decline and what actions might be adopted to prevent their demise.

To this end, Part I of the report is divided into five main sections:

1. data, methods, and analysis;
2. description of quantitative changes in the status and trends of major components of Caribbean coral reef ecosystems (corals, macroalgae, sea urchins, and fish) since 1970 throughout the tropical western Atlantic;
3. analysis of the different potential drivers of change to attempt to determine their comparative impact on reefs to the present day and likely impacts in the future;
4. synthesis of results; and
5. recommendations for management.

1. DATABASE, METHODOLOGY, AND ANALYSIS

Most of the quantitative data for Caribbean reefs is unpublished or buried in gray literature and government reports that have not been systematically exploited in previous long-term assessments of changing conditions throughout the region. We contacted hundreds of people in all the countries of the Caribbean via several thousand emails, requests for data posted on relevant websites, and through presentations and interviews at the 64th Gulf and the Fisheries Institute (GCFI) annual conference in Puerto Morelos, Mexico in 2011 and the 12th International Coral Reef Symposium (ICRS) and ICRI meetings in Cairns, Australia in 2012. We also corresponded with managers of large monitoring data sets, including the National Oceanic and Atmospheric Administration (NOAA) Center for Coastal Monitoring and Assessment Biogeography Branch, Caribbean Coastal Marine Productivity Program (CARICOMP), Atlantic and Gulf Regional Reef Assessment (AGRRA), Caribbean Adaptation to Climate Change Mainstreaming Adaptation to Climate Change (CPACC MACC) programs, Coral Reef Evaluation and Monitoring Project (CREMP) carried out by Florida Fish and Wildlife (FWC), and the Inventory and Monitoring Program (I&M) conducted by the National Park Service South Florida Caribbean Network (NPS SFCN).

1a. SCOPE OF THE DATA

We obtained data from 78 principal investigators supplemented by data from 143 published scientific papers and reports. In total, these include data from more than 35,000 surveys of corals, macroalgae, the sea urchin *Diadema antillarum*, and reef fish from 287 data sets, distributed among 90 reef locations in 34 countries, states, or territories (Tables 1 and 2, Fig. 4). This is by far the largest amount of quantitative coral reef survey data ever compiled and exceeds several fold the data employed for previous analyses of Caribbean reefs (Gardner et al. 2003; Schutte et al. 2010).

Sampling units are defined as follows:

Survey: A set of replicate data points collected at a unique reef site, date, depth, or range of depths. Individual surveys are replicates or averaged values for a series of replicates within datasets at a unique site, date and depth.

Data Set: An individual data collection by a single researcher or research team in a particular country, territory, or state.

Site: One or more surveys at the same depth and GPS coordinates on the same reef.

Location: A geographic cluster of exact survey coordinates (sites) revealed by GIS and further defined by prevailing oceanographic conditions (windward or leeward, onshore or offshore, etc.) and political boundaries.

Country, State, or Territory: An independent nation (Cuba, Curaçao, Jamaica, Panama) or political entity attached to or within a single country (Bonaire, Florida, Guadeloupe, Puerto Rico), either of which may be further subdivided to reflect geographic isolation (St. Thomas, St. Croix, and St. John within the US Virgin Islands within the USA).

Compilation of the great majority of the data presented very substantial challenges for organization and management. We obtained two types of ecological data: (1) raw data provided directly by researchers and (2) summarized data extracted from peer-reviewed articles and government or gray literature reports. The datasets were based upon various sampling designs and methodologies, reported widely variable ecological and environmental parameters, utilized differing codes and groupings for reported variables, and were presented in a unique format. Consequently, we had to convert each database into a standardized, uniform format with accompanying crucial meta-data on precise geographic locations for GIS, sampling methodology, reef environmental parameters, and reef management history and status. To accomplish this, we developed a data template (Appendix 1) by soliciting input from study collaborators at the workshop in Panama, the ICRS and ICRI meetings in Cairns Australia, and countless additional emails. Compiling and organizing this information required a coordinated and extremely time-consuming effort to evaluate each dataset individually and to edit, reformat, and check for data consistency and quality before merging datasets into a master database.

The great majority of the data are for reef corals, macroalgae, *Diadema*, and fishes from fore-reef and patch-reef environments in depths between 1-20 m (Fig. 5). Therefore, all of the analyses for this report are restricted to these types of reefs and depths. Data are sparse and geographically limited until the mass mortality of *Diadema antillarum* in 1983. This striking event, combined with growing awareness of the severity of *Acropora* mortality due to White Band Disease (WBD), stimulated a surge of monitoring efforts. Numbers of surveys for corals and *Diadema* are about 12,000, for reef fish about 20,000, but only about 4,000 for macroalgae.

TABLE 1. Summary of numerical extent of data collected for the wider Caribbean, Gulf of Mexico, and Bermuda. For definitions of terms see text.

Number of	Coral	Macroalgae	Urchin	Fish	Overall
Countries/Territories	33	31	32	25	34
Locations	88	73	73	73	90
Datasets	193	129	107	68	287
Principal Investigators	65	55	19	20	78
Individual surveys	12,116	4,109	11,962	20,279	35,577
Datasets from papers	59	30	96	4	143
Start Year	1965	1970	1965	1988	1965
End Year	2012	2012	2012	2011	2012
Years surveyed	42	35	38	18	43

FIGURE 4. Geographic distribution of the 90 reef locations analyzed for this study and listed in Table 2. Large circles indicate 21 reef locations with the most complete time series data for analysis of long-term trends in coral cover.

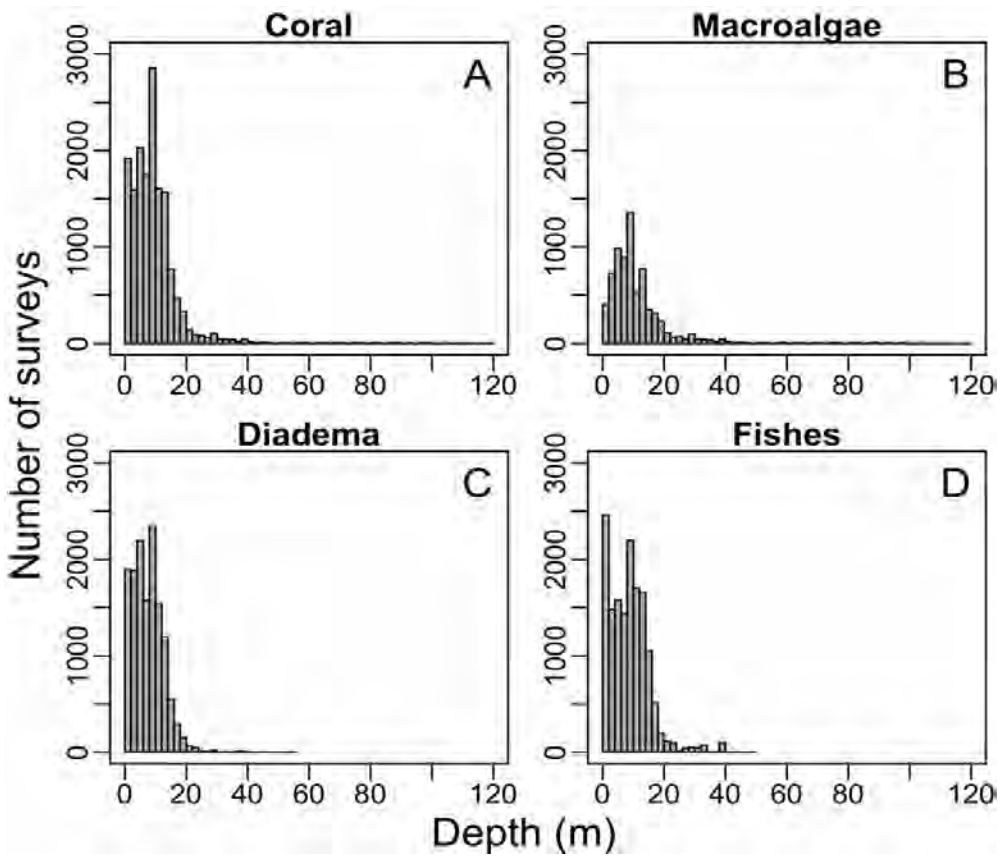

FIGURE 5. Frequency of surveys by depth for (A) corals (= 9.5 ± 8.34), (B) macroalgae (= 11.3 ± 10.7), (C) *Diadema antillarum* (= 7.7 ± 4.9), and (D) reef fishes (= 8.8 ± 6.3).

TABLE 2. List of coral reef locations used for this study with extent of sampling, range of years sampled, depth, changes in coral cover for locations sampled more than once, and recent biomass of parrotfish. Locations without percent coral cover were included for data for macroalgae, sea urchins, or fish.

Label	Country or Territory	Location	# of data sets	# of surveys	Start year	End year	# of years	Year span	Depth range (m)	Oldest coral cover (%)	Most recent cover coral (%)	Change coral cover (%)	Oldest macro-algal cover (%)	Most recent MA cover (%)	Change MA cover (%)	Parrotfish biomass after 1999 (g/m²)
1	Antigua & Barbuda	Antigua & Barbuda	2	227	2005	2008	3	4	2 - 14	16.5	3.8	-12.7	24.8	13.8	-11	19.4
2	Aruba	Aruba	1	13	1986	1986	1	1	5 - 5	24	24					
3	Bahamas	Cay Sal Bank	1	685	2011	2011	1	1	4 - 28	7.1	7.1		68.7	68.7	0	14.8
4		Exuma Land Sea Park	1	138	1993	2007	14	15	0 - 20	7.3	7.8	0.5	11.2	33.7	22.5	9.8
5		Other	4	2237	1994	2011	14	18	0 - 27	9.7	11.7	2	10.5	44.7	34.2	27.7
6	Barbados	Leeward	7	186	1974	2007	24	34	1 - 22	37.4	15	-22.4	0	14.5	14.5	
7		South	1	104	1978	2007	12	30	3 - 25	11.1	17.4	6.3	6.8	1.8	-5	
8		Windward	1	3	2002	2003	2	2	12 - 13							
9	Belize	Atoll Leeward	4	963	1970	2009	9	40	0 - 18	55.5	20.7	-34.8	43.9	45.6	1.7	6
10		Atoll Windward	7	710	1970	2011	12	42	0 - 25	93.2	20.9	-72.3	6.8	51.5	44.7	6.3
11		Central Barrier	6	751	1978	2012	21	35	0 - 49	32.5	15.9	-16.6	4.9	55.5	50.6	7.2
12		Gulf Honduras	1	191	2006	2006	1	1	5 - 18	7.6	7.6					4.5
13		Inner Barrier	2	697	1994	2009	14	16	1 - 15	42.4	16.2	-26.2	4.1	1.6	-2.5	10.7
14		Northern Barrier	6	581	1997	2012	14	16	1 - 26	33.3	16.9	-16.4	20.3	48.8	28.5	8.9
15		Southern Barrier	3	414	1997	2011	9	15	1 - 24	23.5	13.5	-10	12.8	66.8	54	6.4
16	Bermuda	Bermuda	5	365	1977	2012	19	36	0 - 40	19.4	38.6	19.2	8.4	12.1	3.7	21.9
17	British Virgin Islands	British Virgin Islands	2	292	1992	2012	21	21	5 - 13	18	14.3	-3.7	5	10.2	5.2	13.8
18	Cayman Islands	Grand Cayman	5	356	1995	2009	8	15	2 - 20	19	30.7	11.7	13.9	31.4	17.5	12.7
19		Little and Brac	4	700	1988	2011	13	24	2 - 27	36.1	24.6	-11.5				15.7
20	Colombia	Providencia	1	52	1999	2006	6	8	1 - 21	20.5	20.5		41.9	41.9	0	
21		San Andrés	4	85	1992	2006	9	15	2 - 20	28.2	12.6	-15.6	19.8	23.8	4	
22		Santa Marta Region	10	61	1977	2005	21	29	3 - 23	28	31.1	3.1	0	3.3	3.3	
23	Costa Rica	Cahuita	2	90	1977	2011	17	35	2 - 10	40.4	18	-22.4	13.2	12.5	-0.7	39.8
24	Cuba	Jardines de la Reina	3	898	2001	2011	3	11	0 - 17	15.6	30.1	14.5	32	35.7	3.7	20.4
25		North	2	597	1989	2001	7	13	0 - 23	6.8	15.9	9.1	38.5			6.9
26		Southwest	4	1168	1998	2011	6	14	0 - 21	9.8	25.2	15.4	43.2	19.1	-24.1	8.4
27	Curacao	Curacao Northwest	6	202	1983	2011	16	29	2 - 20	18.1	13.3	-4.8	6	8.3	2.3	31.6
28		Curacao Southwest	13	335	1973	2011	27	39	1 - 40	40.7	31.5	-9.2	0	7.8	7.8	15.2
29		Curacao Windward	1	6	2001	2001	1	1	20 - 20							8.6

Label	Country or Territory	Location	# of data sets	# of surveys	Start year	End year	# of years	Year span	Depth range (m)	Oldest coral cover (%)	Most recent cover coral (%)	Change coral cover (%)	Oldest macro-algal cover (%)	Most recent MA cover (%)	Change MA cover (%)	Parrotfish biomass after 1999 (g/m^2)
30	Dominica	Dominica	1	9	2007	2009	2	3	5 - 14	11.4	9	-2.4	10.6	0.1	-10.5	
31	Dominican Republic	North	2	202	2004	2006	2	3	0 - 12	23.4	21.3	-2.1	8.9	8.9	0	3.1
32		Punta Cana	1	235	2003	2003	1	1	1 - 8	8	8					3.9
33		South	2	140	1994	2004	6	11	4 - 33	7.8	28.1	20.3	17.7	9.9	-7.8	9.2
34	French Antilles	Guadeloupe	1	192	1988	2011	20	24	1 - 15	23	18.6	-4.4	52.5	33.8	-18.7	24.4
35		Martinique	1	38	2001	2007	7	7	5 - 10	35.7	17.4	-18.3	33.7	33.8	0.1	20.4
36		St. Barthelemy	1	47	2002	2011	10	10	10 - 10	25.3	10.8	-14.5	14.9	53	38.1	17.1
37	Grenada	Grenada other	1	12	2005	2009	3	5	2 - 20	41.7	27.7	-14				
38		Leeward	1	11	2007	2009	2	3	17 - 30	10.1	12.8	2.7	37.6	65.9	28.3	
39	Guatemala	Guatemala	1	59	2006	2006	1	1	7 - 15	9.9	9.9					3
40	Honduras	Bay Islands	3	981	1987	2010	4	24	0 - 19	20.6	21.6	1	34.5	43	8.5	11.8
41		Near shore	1	328	2006	2006	1	1	2 - 20	12	12					22.5
42	Jamaica	Montego Bay	7	348	1973	2007	18	35	0 - 16	10.6	19.4	8.8	0	53.4	53.4	4.6
43		North central	17	724	1969	2011	38	43	1 - 120	44.6	19.6	-25	1.1	24.2	23.1	6.9
44		Northeast	3	308	1977	2007	9	31	1 - 17	47	11.8	-35.2	26.9	55.9	29	5.4
45		Pedro Bank	1	301	2005	2005	1	1	1 - 21	14.7	14.7					15.4
46		Port Royal Cays	3	20	1977	2011	11	35	4 - 13	24.9	4.7	-20.2	55.7	7.7	-48	
47		West	4	269	1977	2012	12	36	1 - 18	40.3	7.8	-32.5	20.7	55.7	35	8.1
48	Mexico	Alacran	2	7	1985	1985	1	1	2 - 35	11.2	11.2		45.6			
49		Chinchorro Bank	2	486	2000	2008	5	9	0 - 29	17	7.9	-9.1	7.7	7.7	0	1.4
50		Cozumel Leeward	4	678	1984	2011	11	28	0 - 28	25.5	12.1	-13.4	32.9	21.6	-11.3	3.5
51		Cozumel Windward	1	77	2005	2005	1	1	1 - 17	9.2	9.2					0.7
52		North East Yucatan	6	1105	1979	2010	15	18	0 - 28	20.1	7.9	-12.2	30.9	10.6	-20.3	6.2
53		South East Yucatan	2	1028	1985	2009	7	11	0 - 21	29.8	15.9	-13.9	40.4	36.3	-4.1	5.1
54		Veracruz	3	152	1965	1999	4	35	1 - 21	34.1	17.2	-16.9				
55	Navassa	Navassa	1	5	2002	2012	5	11	27 - 28	46.4	10.7	-35.7	41.7	65.7	24	
56	Netherlands	Bonaire Leeward	9	408	1973	2011	23	39	3 - 40	54.8	37.1	-17.7	6.1	17.7	11.6	32.3
57		Bonaire Windward	4	236	1988	2008	4	21	3 - 31	31.9	9.7	-22.2	35	66.3	31.3	19.1
58		Saba	2	219	1993	2003	7	11	3 - 20	19.5	9.4	-10.1	25.1	5.5	-19.6	13.5
59		Saba Bank	1	54	1999	1999	1	1	14 - 21	24.3	24.3					14.5
60		St. Eustatius	1	213	1999	2007	4	9	11 - 19	21.8	21.8					23

Label	Country or Territory	Location	# of data sets	# of surveys	Start year	End year	# of years	Year span	Depth range (m)	Oldest coral cover (%)	Most recent cover coral (%)	Change coral cover (%)	Oldest macro-algal cover (%)	Most recent MA cover (%)	Change MA cover (%)	Parrotfish biomass after 1999 (g/m^2)
61	Nicaragua	Corn Islands	2	269	1993	2003	5	11	2 - 16	28.2	24.4	-3.8	37.4			5.1
62	Panama	Bahia Las Minas	3	215	1985	2011	19	27	2 - 14	23.7	12.3	-11.4	42	3.2	-38.8	
63		Bocas del Toro	4	473	1999	2011	13	13	1 - 17	29.7	13.6	-16.1	12.5	10.4	-2.1	12.3
64		Costa Arriba	3	154	1985	2011	19	27	2 - 17	24.7	13.9	-10.8	56	31.6	-24.4	
65		San Blas	3	1118	1980	2005	23	26	0 - 21	38.8	30.9	-7.9	0.6			13.3
66	Puerto Rico	Guanica	1	6	2005	2006	2	2	2 - 18	27.6	17.3	-10.3				
67		Jobos Bay	1	25	2009	2009	1	1	0 - 12	8.7	8.7		15	15	0	2.1
68		La Paguera	5	1265	1989	2012	20	24	0 - 112	16.4	19.2	2.8	5.4	10.4	5	5.6
69		Mona Islands	1	38	2008	2008	1	1	30 - 103	4.5	4.5		60.7	60.7	0	
70		Turrumote	1	11	2002	2010	6	9	0 - 19	23.8	23.8					9.5
71		Vieques & Culebra	5	358	1978	2008	7	31	2 - 48	42.6	8.1	-34.5	1.9	10.6	8.7	19
72	St. Kitts & Nevis	St. Kitts & Nevis	2	446	2007	2011	3	5	4 - 24	10.3	11.1	0.8	48.5	36.8	-11.7	13
73	St. Lucia	St. Lucia Leeward	2	12	1993	2009	3	17	8 - 21	48.5	10.1	-38.4	41.4	8.1	-33.3	
74	St. Martin	St. Martin	1	52	1999	2007	3	9	8 - 12	12.5	12.5					12.1
75	St. Vincent &	Grenadines	4	304	1976	2007	5	32	2 - 17	30.4	19.5	-10.9				16.7
76	the Grenadines	St. Vincent	2	108	2007	2009	3	3	2 - 11	29.2	24.9	-4.3	2.3	0.4	-1.9	6.8
77	Trinidad & Tobago	Trinidad & Tobago	1	16	1994	2012	16	19	10 - 10	24.1	19.1	-5	0	0.9	0.9	
78	Turks & Caicos	Turks & Caicos Islands	2	565	1999	1999	1	1	2 - 23	17.7	17.7		11.7	11.7	0	7.4
79	U.S.A	Dry Tortugas	9	671	1975	2011	19	37	1 - 28	20.8	8	-12.8	0.6	31.7	31.1	7.5
80		Flower Garden Banks	3	347	1974	2011	6	38	18 - 43	56.7	53.1	-3.6	13.2	25.6	12.4	35.8
81		Lower Florida Keys	5	1094	1972	2011	24	40	1 - 27	31.8	10.3	-21.5	15.3	15.2	-0.1	24.2
82		Middle Florida Keys	4	390	1991	2011	17	21	3 - 24	8.4	8	-0.4	7	22.8	15.8	8.4
83		Southeast Florida	4	256	1989	2011	17	23	2 - 17	12.5	2.8	-9.7	3.4	4.6	1.2	3.6
84		Upper Florida Keys	13	1880	1965	2011	31	47	0 - 27	27.9	6.1	-21.8	0.8	15.3	14.5	20.3
85	U.S. Virgin Islands	St. Croix	10	505	1976	2011	32	36	0 - 40	23.2	4.7	-18.5	3	8.9	5.9	13.1
86		St. Thomas	5	473	1978	2010	19	33	0 - 33	27.4	13.6	-13.8	1.5	47.7	46.2	11.4
87		St. Thomas shelf edge	2	620	2002	2011	10	10	30 - 40	26.1	33.6	7.5	42.9	26.8	-16.1	9.2
88		St. John	11	2991	1978	2011	31	34	0 - 27	34.1	10.1	-24.0	0.6	28.9	28.3	8.3
89	Venezuela	Los Roques	2	209	1999	2008	7	10	1 - 15	69	78	9				60.7
90		Morrocoy	2	165	1996	2011	16	16	5 - 13	55	38.5	-16.5				

There are no quantitative survey data for reef fish biomass prior to 1989. Data for *Diadema* abundance and macroalgal cover are also rare until the sea urchin began to die *en masse*. Most of the coral data are for total coral cover, but there are also considerable data broken down by genus or species since the early 1970s. Many of the fish surveys only recorded certain groups such as parrotfish or groupers, but the identification and recording of these charismatic taxa appears to be generally good. The greatest problems of data quality are with macroalgae, which were not recorded consistently except by a small number of experts in algal ecology and systematics. We defined macroalgae as erect calcareous or fleshy algae greater than 2 cm tall. These include, but are not limited to species of the genera *Cladophora*, *Dictyota*, *Halimeda*, *Liagora*, *Microdictyon*, and *Sargassum*. In many cases macroalgae were recorded as turf and vice versa, and the CARICOMP protocol distinguished macroalgae by such different criteria that we could not use their algal data in our analysis. Considerable energy was invested in vetting the algal data to throw out all of the questionable data sets, which explains why the numbers of surveys for macroalgae are so much smaller than the other groups.

Most of the surveys employed haphazardly placed or fixed transects or quadrats. Examples include the remarkable nearly 40-year data set provided by Rolf Bak for fixed quadrats in Curaçao and Bonaire, larger scale transect surveys for particular reefs by individual scientists, and large monitoring programs such as CARICOMP, CREMP, and FWC. Surveys were varyingly conducted with widely varying frequency and consistency from 1970 to the present, although the numbers of surveys were small and restricted to only a few locations until the 1980s, and coverage did not substantially increase until the 1990s (Fig. 6A).

Two other major survey programs beginning in the 1990s employed entirely different sampling approaches. AGRRA began in 1997 and used widely varying rapid sampling protocols that have changed throughout the history of the project and also differ among regions surveyed (Fig. 6B). In contrast, data are collected from stratified random sites for the NOAA Biogeography Program surveys in Puerto Rico and the US Virgin Islands, and by the Florida Keys Coral Reef Monitoring Assessment team (PIs: S. Miller and M. Chiappone) and Southeast Florida Coral Reef Monitoring and Evaluation Project (PI: Gilliam) in the Florida Reef Tract (Fig. 6C). In this latter case, surveys include sandy and rocky bottoms in addition to actual reef habitat so are not directly comparable to the other data.

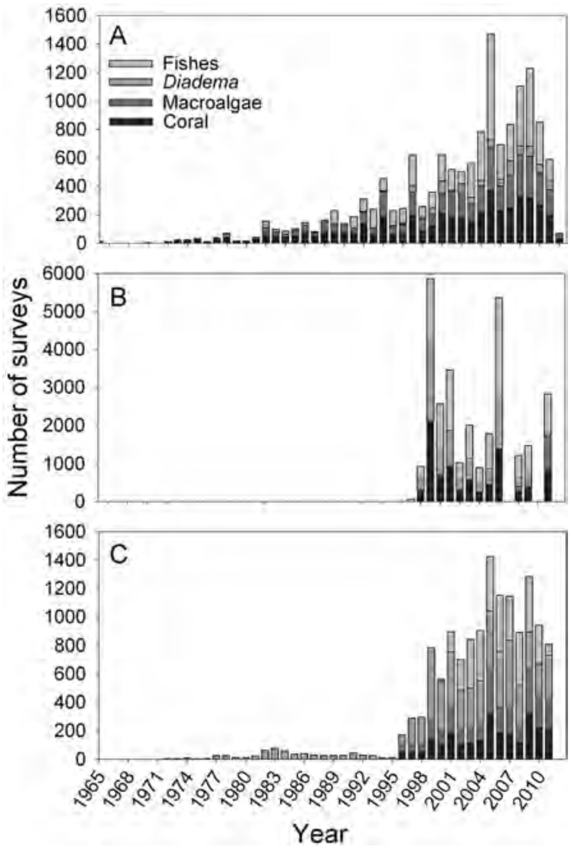

FIGURE 6. Number of surveys by year for coral, macroalgae, *Diadema*, and fishes for (A) all permanent or haphazardly collected data, (B) data from the AGRRA program, and (C) data collected from locations randomly selected for each census so that there are no repeated observations from the same geographic point.

The longest time series with consistent data are Rolf Bak's study beginning in 1973 for large fixed photo quadrats at 4 stations and 4 depths at Curaçao and Bonaire totaling 243 m^2 (Bak et al. 2005). Photographs were taken annually, but data for corals and macroalgae identified from the photographs have been analyzed so far only for 5-year intervals. An additional site in east Curaçao was added starting in 1993. Robert Steneck also began monitoring reefs at Bonaire in 1999 (Steneck and Arnold 2009). Comparably long time series extending back into the early 1970s to early 1980s are available from the northern Florida Keys (Dustan 1977, 1985; Porter and Meier

1992), Jamaica (Hughes and Jackson 1985; Liddell and Ohlhorst 1986, 1992; Hughes 1994; Loya, unpublished data), St. John and St. Croix in the United States Virgin Islands (Rogers et al. 1991, 2008; Edmunds 2002; Rogers and Miller 2006; Miller et al. 2009), and Panama (Guzmán et al. 1991; Shulman and Robertson 1996; Guzmán 2003). However, these records were compiled by different workers at different times and are therefore not as consistent or complete as data from the Dutch Caribbean.

1b. ANALYSIS

Trends in percent cover were assessed for total corals and macroalgae. Trends in density were assessed for *Diadema antillarum* and reef fishes. Analyses were based on a hierarchal structuring of the data and were summarized based on means of surveys within individual datasets for each location. Each survey was assigned to a "location" so each dataset contributed one value to each location unless that dataset covered more than one location. Finally, means were calculated for each location. All statistical analyses were conducted using the software program R version 2.15 (R Development Core Team 2011).
Because the intensity of sampling varies so greatly in time and space, we partitioned the data into three 12 to 14-year time intervals based on major ecological events that extended throughout the wider Caribbean. These are:

1. 1970-1983: Interval from our oldest data until the massive die-off of the sea urchin *Diadema antillarum* in 1983 including the first reports of White-Band Disease (WBD) from the mid 1970s to early 1980s.
2. 1984-1998: From the end of the *Diadema* die-off up to and including the widely reported 1998 extreme heating event.
3. 1999-2011: The modern era of massively degraded coral reefs including the extreme heating events in 2005 and 2010.

We also selected a subset of 21 reef locations for more detailed statistical analyses (large circles in Fig. 4) based upon availability of coral cover data for all three time intervals as well as associated metadata important for the interpretation of the possible drivers of reef degradation.

General and generalized linear mixed effects models (Pinheiro and Bates 2000) were used to test explanatory variables across time and with response variables (R packages lme4: Bates and Maechler 2010, and glmmADMB; Skaug and Fournier 2013). Where the response variable was percent cover we used generalized linear mixed models assuming a beta distribution since the response variable is a percentage. Otherwise, general linear mixed models were used on square-root transformed response variables to reduce the mean-variance relationship and meet the assumptions of linear modeling. We accounted for temporal and spatial autocorrelation by adding random components of year nested within survey and dataset (for definitions see previous section), thus each survey within each dataset was treated as a repeated measure. The model accounted for differences in sampling by location by further nesting within location. For each model 95% confidence intervals were calculated for means that accounted for variation due to dataset and location based on 5000 simulations (R package arm: Gelman et al. 2010). Criteria for comparing model fits were based on minimizing the Akaike Information Criterion (AIC). An estimate of restricted maximum likelihood was used to fit the models. Bonferroni-adjusted pair-wise multiple comparisons were conducted for specific post-hoc hypotheses where appropriate.

Means were modeled for time bins defined above, as well as the values for the oldest (first) year and most recent year a location was studied. In most cases, the oldest or most recent year for a given location was comprised of a single dataset, but in the case of multiple datasets per year the datasets were averaged. Current coral cover was estimated by considering, for 88 locations, the most recent estimate of cover per location as long as the most recent survey was after 1998. Analyses across time bins were conducted for each location with mixed effects models including random effect of dataset. Tukey Honest Significant Differences for post-hoc pairwise comparisons of means were conducted with adjustments for multiple means.

To assess trends in *Acropora* abundance over time, frequency of occurrence and dominance across various time bins were constructed. Because sample sizes are small before 1950, and the locations represented in various time bins are not consistent, care should be taken when interpreting results. Thus we constructed confidence intervals for proportions assuming a binomial distribution with the Pearson-Klopper method. Temporal trends in *Acropora* species percent cover were also examined for data after 1975 in the GCRMN database where means and standard deviations were calculated as for trends discussed above.

Multivariate ordination was conducted to investigate temporal trends in benthic community composition. Locations were included in the ordination if data were available for percent cover of corals at the species (or species group) level as well as for total macroalgae for the same replicate. Coral species were combined into 19 groups by species or genera, and by growth form, to reduce zero occurrences for rare species, especially for species with limited geographic range.

We used two forms of ordination analysis to assess changes in coral and macroalgal assemblage composition. Principal Components Analysis (PCA) uses Euclidean distances to compute a similarity matrix projected on a PCA ordination graph that illustrates the total amounts of the variance "explained" along the first, second, and third PCA axes. PCA has the advantage that results are easily interpretable with taxa represented by arrows that indicate increasing abundance in the direction of the arrow. In contrast, non-metric, multidimensional scaling (MDS) is based on rank order correlation and uses a Bray-Curtis similarity matrix to generate an ordination (Clarke et al. 2005). MDS has the important advantage of not treating zeros as values of occurrence since multiple zero occurrences common in ecological data can play havoc with resemblance based on Euclidean distance. But, the order of the axis does not necessarily imply importance, which renders the results less intuitive and more difficult to interpret.

PCA and MDS were performed on square root transformed mean percent cover data across two time bins to explore the change in benthic assemblage composition over time (R package vegan: Oksanen et al. 2013). Species were scaled proportional to the eigenvalues for graphical purposes, so angles reflect correlations in multidimensional space (Legendre and Legendre 1998).

Relationships between coral and macroalgal cover and anthropogenic drivers were explored using various methods depending on the question and data structure (see text of relevant sections for detailed methodological information). Wherever relevant, we employed generalized linear mixed effects models with a beta distribution as described above to test the relationship between coral cover and drivers.

2. OVERALL CHANGES IN BIOLOGICAL ABUNDANCE

We first discuss status and trends of corals and macroalgae, which are the two major sessile components of Caribbean reef communities. Next we consider the demise of three major taxa that have severely declined over the past 40 years: the branching coral genus *Acropora* that once overwhelmingly dominated most shallow reefs, the sea urchin *Diadema antillarum*, and parrotfishes. The latter two are (or were) the most important macroscopic herbivores on Caribbean reefs.

2a. PATTERNS OF CHANGE FOR CORALS AND MACROALGAE

Mean live coral cover for the tropical western Atlantic based upon the most recent estimates of cover for each of the 88 locations in Table 2 is 16.8% (median 14.5%, range 2.8% for southeast Florida to 53.1% for the Flower Garden Banks). The mean is 68% higher than the mean of 10% cover reported previously for 2001 (Gardner et al. 2003) but almost identical to the mean of 16.0% cover for the years 2001-2005 from a more recent and more rigorous assessment (Schutte et al. 2010).

We further refined the estimate of mean percent coral cover using statistical methods to take into account the great variation among locations and datasets, resulting in a mean of 14.3% (+2.0, -1.8) This lower value reflects the skewed shape of the variation in coral cover across the region, wherein most locations fall well below the mean with several notable exceptions of locations with considerably higher than average coral cover (Fig. 7). This variation is further apparent when the quantiles of current coral cover are considered. The upper quartile is 21.2%, while the 95% quantile is 31.5%. Five locations fall above the 95% quantile including Bermuda, the leeward coast of Bonaire, the southwest coast of Curaçao, the Flower Garden Banks in the northern Gulf of Mexico, and Morrocoy National Park on the mainland coast of Venezuela (However, the high value for Morrocoy resulted from the relocation of the CARICOMP study site to a different reef after all the corals at the original location had died.).

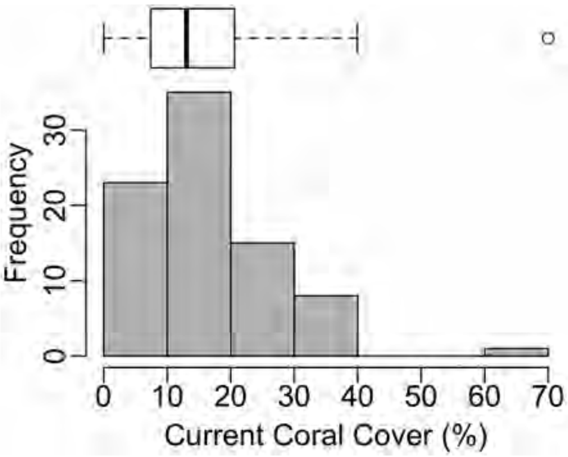

FIGURE 7. Histogram of current estimates of percent coral cover from 88 locations across the Caribbean with box plot reflecting 0, 25%, 50%, 75% quantiles.

Coral cover declined at 52 of 71 (73%) locations in Table 2 for which time series data are available (Fig. 8). The decline was greatest for locations with the oldest estimates of percent cover (Fig. 8A) and the longest periods of observation (Fig. 8B). This is the now classic pattern of "shifting baselines" for fisheries management (Pauly 1995; Jackson and Jacquet 2011; Jackson et al. 2012). Another striking example of the Shifting Baselines Syndrome in the Caribbean concerns the status and trends of green turtle populations on nesting beaches that have been surveyed for varying lengths of time (Jackson 1997; McClenachan et al. 2006). Beaches observed for less than 40 years exhibit a wide mixture of positive and negative trends, whereas all beaches observed for more than 40 years have suffered very large declines of 75-95%.

Long-term changes in corals and macroalgae

Average changes in coral and macroalgal cover over the three time intervals are presented in Table 3 for all locations and the 21 long-term data locations in Fig. 4. Mean coral cover in depths of 0-20 m for all locations declined from 33.0% before 1984, to 18.6% from 1984-1998, and 16.4% from 1999 to today (Fig. 9A, Table 3). The average pattern of decline did not vary greatly with depth. Coral cover before 1984 was 33.2% on reefs from 0-5 m depth versus 32.6% cover in depths of 5.1-20 m (Table 3). After 1999 coral cover declined slightly more on reefs shallower than 5 m. *Acropora palmata* once overwhelmingly dominated reefs in 0-5 m with cover as great as 50 to 85% (Woodley et al. 1981; Gladfelter 1982). Thus, our data suggest that the decline of *Acropora palmata* had begun before the first quantitative surveys at most reef locations. In contrast, a locally variable mix of species including the *Orbicella* (formerly *Montastraea*) *annularis* species complex, other massive and plating corals, and *Acropora cervicornis* formerly dominated reefs from 5 to 20 m (Goreau 1959; Kinzie 1973; Bak 1977; Bak and Luckhurst 1980; Liddell and Ohlhorst 1986, 1988).

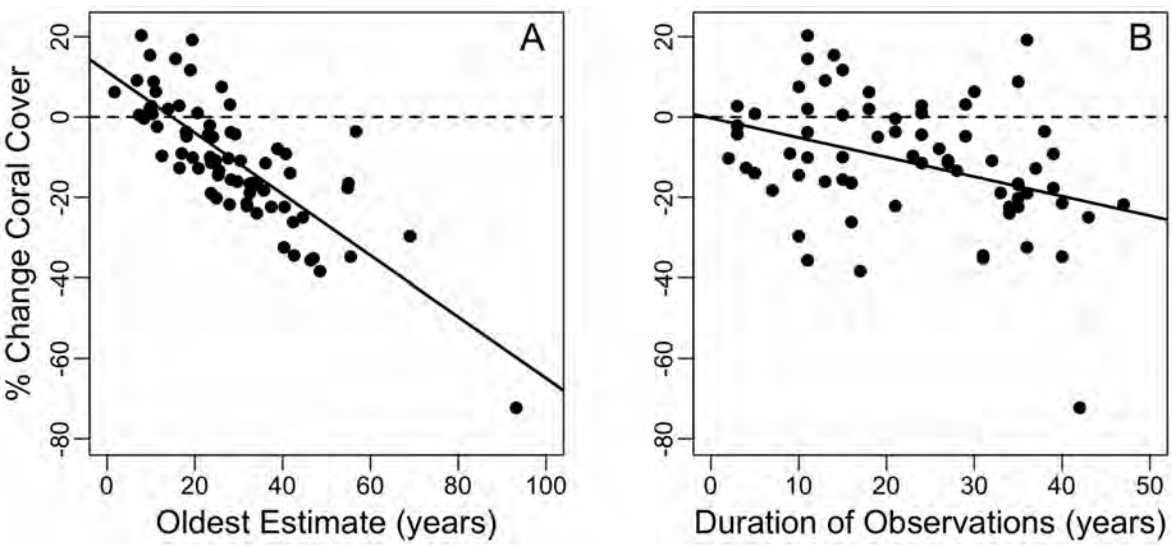

FIGURE 8. Percent change in coral cover at 71 locations in Table 2. Change in percent cover of corals in relation to (A) cover measured in the earliest year of observation ($R^2 = 0.63$, $p < 0.01$) and (B) the duration of the study period for that location ($R^2 = 0.17$, $p < 0.01$).

TABLE 3. Corrected values of percent cover of corals and macroalgae by depth for 3 time periods for all locations and for the subset of 21 circled locations in Fig. 4. Values are means with 95% confidence intervals in parentheses calculated with mixed-effect beta regression that takes into account variability due to location and datasets

Depth (m)	All locations			21 locations		
	1970-1983	1984-1998	1999-2011	1970-1983	1984-1998	1999-2011
Coral cover (%)						
0-20	33.0 (28.7, 37.6)	18.6 (16.2, 21.2)	16.4 (14.8, 18.1)	31.5 (27.7, 35.6)	18 (15.1, 21.3)	15.8 (13.0, 19.0)
0-5	33.2 (22.7, 45.6)	14.1 (11.2, 17.6)	15.4 (13.2, 17.9)	26.6 (20.2, 34.1)	13.4 (9.5, 18.5)	12.2 (8.6, 17.0)
5.1-20	32.6 (28.1, 37.3)	19.4 (16.3, 22.9)	16.5 (14.8, 18.4)	34.6 (30.3, 39.1)	19.6 (16.6, 23.0)	16.7 (12.9, 21.3)
Macroalgal cover (%)						
0-20	7.0 (3.6, 13.0)	23.6 (17.3, 31.4)	23.5 (19.8, 27.6)	5.6 (2.7, 11.0)	21.6 (14.0, 31.8)	23.9 (18.4, 30.5)
0-5	12.1 (5.3, 25.2)	40.1 (24.4, 58.2)	24.0 (17.9, 31.4)	10.2 (3.9, 24.2)	42.4 (29.0, 56.9)	21.1 (16.0, 27.4)
5.1-20	4.0 (1.8, 9.0)	21.5 (15.1, 29.5)	23.2 (19.2, 27.8)	4.0 (1.8, 9.0)	19.3 (11.3, 31.0)	25.8 (18.8, 34.2)

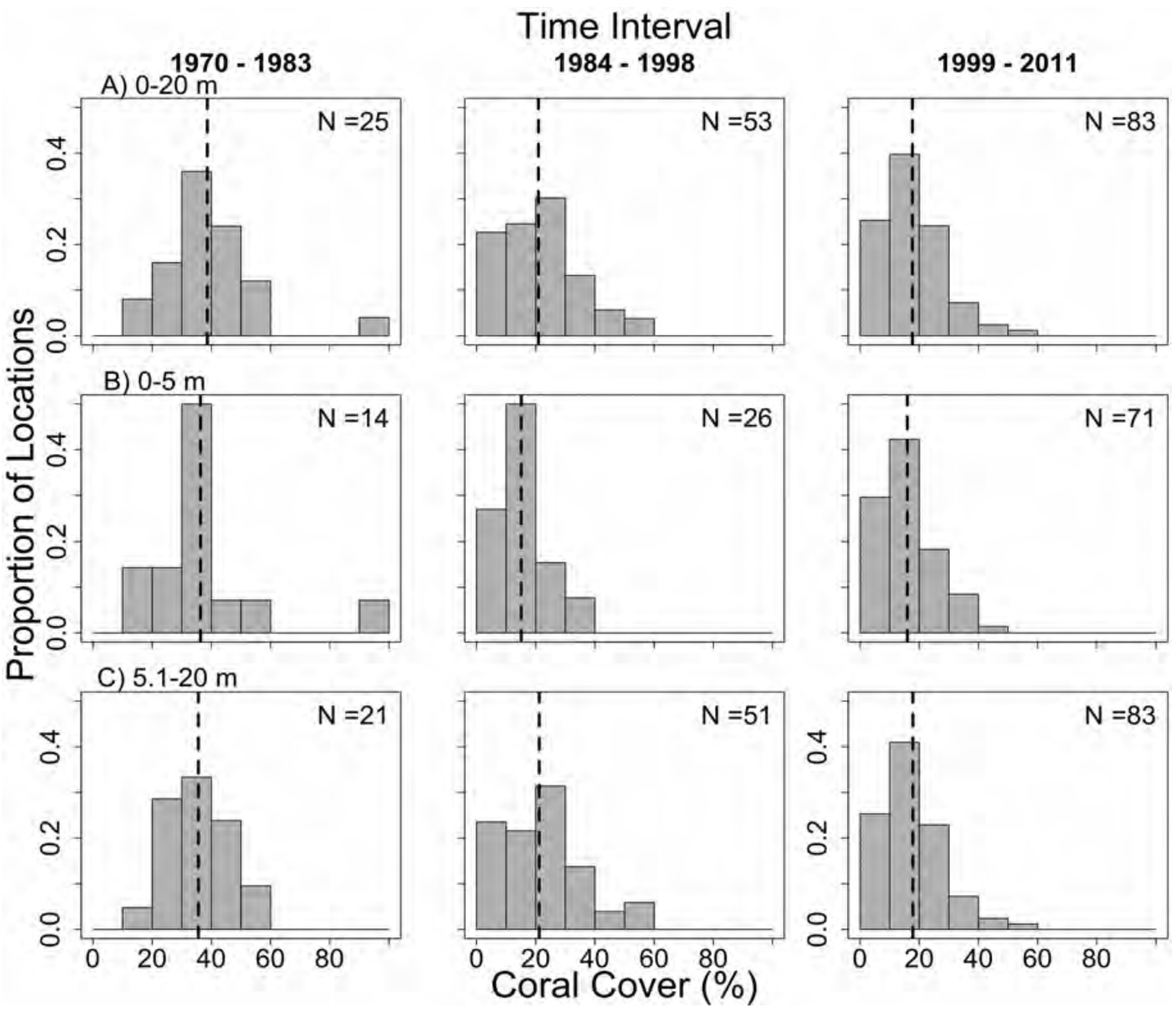

FIGURE 9. Distribution of coral cover among all the locations in Table 2 for all three time intervals and depths of (A) 0-20 m, (B) 0-5 m, and (C) 5.1-20 m. Values represent the means within locations for each time bin. Vertical line indicates uncorrected mean, and N is the number of locations.

Changes in coral cover were similar on the 21 reefs in Fig. 4 except that coral cover was lower in shallow depths before 1984 and the subsequent declines were more abrupt between time intervals 1 and 2 (Table 3, Fig. 10).

Macroalgal cover in 0-20 m was 7.0% prior to the mass mortality of *Diadema antillarum* in 1983 and then tripled to 23.6% afterwards (Fig. 11A, Table 3). However, the patterns vary strongly with depth. Macroalgal cover from 0-5 m depth averaged

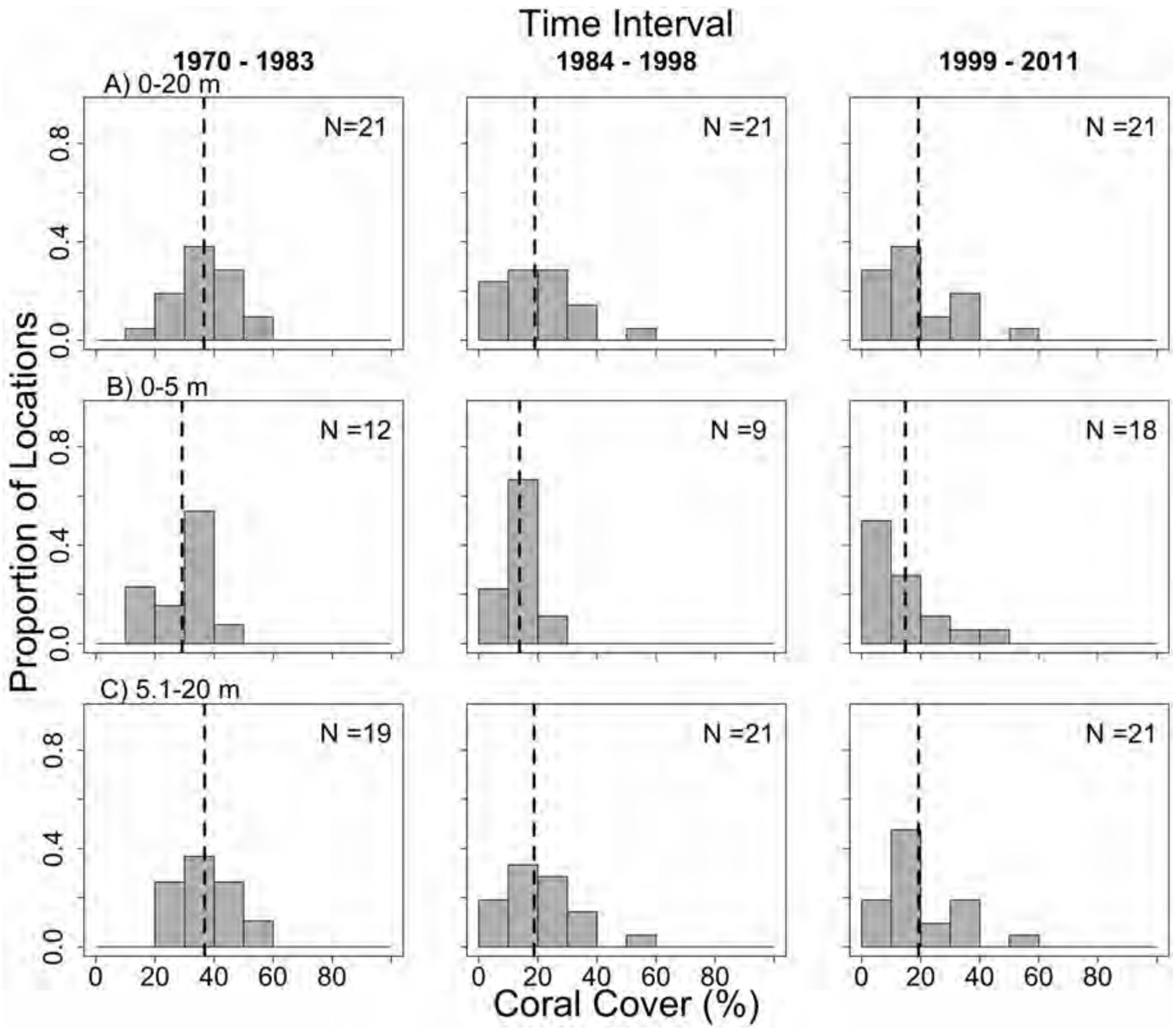

FIGURE 10. Distribution of coral cover for the 21 long-term data locations (large circles in Fig. 4) for all three time intervals at depths of (A) 0-20 m, (B) 0-5 m, and (C) 5.1-20 m. Values represent the means within locations for each time bin. Vertical line indicates uncorrected mean, and N is the number of locations.

12.1% before 1984 and increased afterwards to 40.1% (Table 3). In contrast, macroalgal cover was only 4.0% on reefs in 5.1-20 m before 1984, and then increased 5-fold after the *Diadema* died (Fig. 11C, Table 4). Macroalgal cover since 1999 has averaged about 23.2% but varied enormously among the 67 locations from 1-69%. Changes in macroalgae on the 21 reefs were similar to that for the entire dataset (Table 3, Fig. 12).

The clearly opposite trends in coral and macroalgal cover (Figs. 9 and 10 versus Figs. 11 and 12) demonstrate a highly significant and persistent shift throughout the wider Caribbean from reef communities where corals were the most abundant occupiers of space to reef communities where macroalgae are more abundant than corals (Fig. 13). Such a striking reversal from coral to macroalgal dominance is commonly referred to as a phase shift (sensu Hughes et al. 2010); a pattern first documented in even more extreme form from Jamaica between the 1970s and 1990s (Hughes 1994).

Geographic Variation in Reef Degradation

The preceding histograms demonstrate very large geographic differences in the status and trends for coral cover at different reef locations. To document the nature of this variability in greater detail, we constructed two different kinds of timelines for the status and trends in coral cover for two different subsets of reefs presented below. The focus here is on documenting the patterns of variation among sites. Implications and insights derived from the timelines are discussed in the following section of the report on anthropogenic drivers of change.

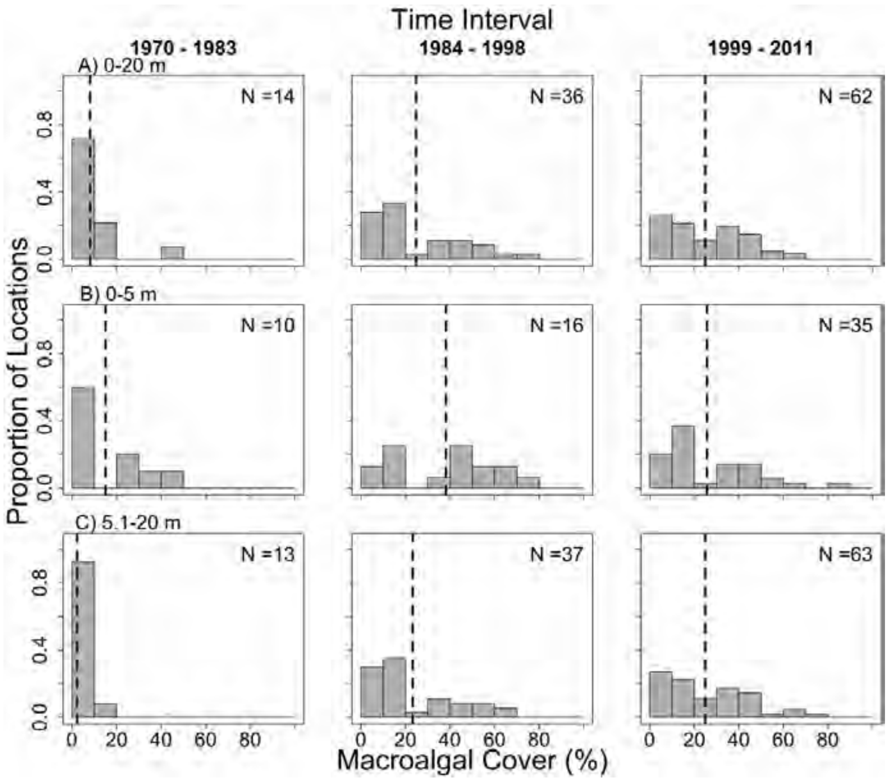

FIGURE 11. Distribution of percent macroalgal cover among all the locations in Table 2 for all three time intervals at depths of (A) 0-20 m, (B) 0-5 m, and (C) 5.1-20 m. Values represent the means within locations for each time bin. Vertical line indicates uncorrected mean, and N is the number of locations.

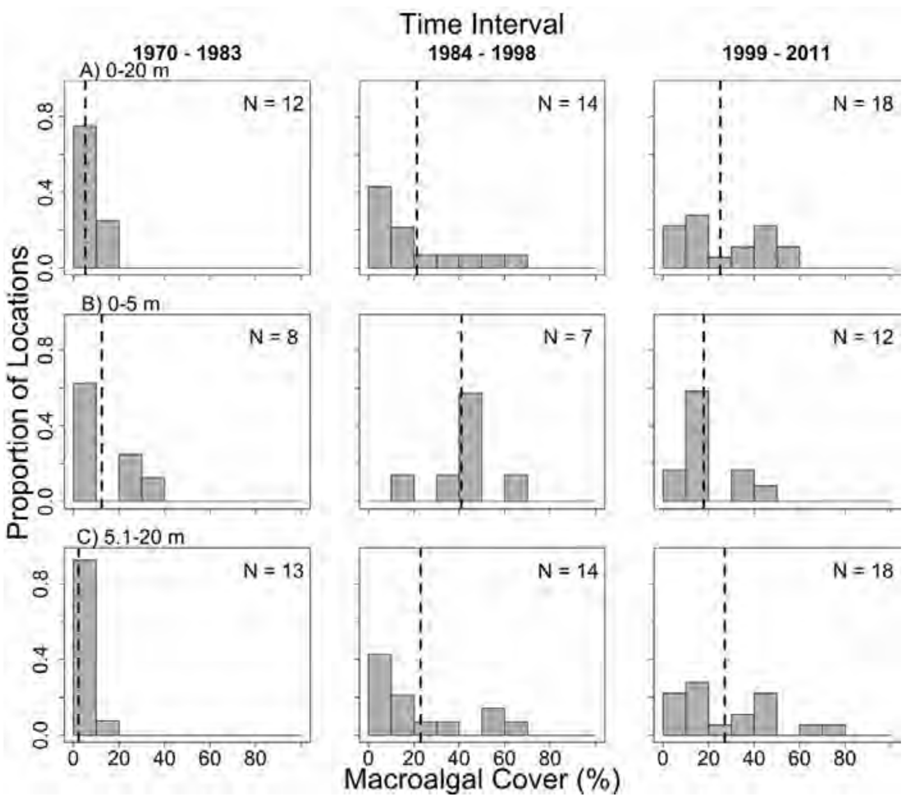

FIGURE 12. Distribution of percent macroalgal cover for the 21 long-term data locations (large circles in Fig. 4) for all three time intervals at depths of (A) 0-20 m, (B) 0-5 m, and (C) 5.1-20 m. Values represent uncorrected means within locations for each time bin. Vertical line indicates mean, and N is the number of locations.

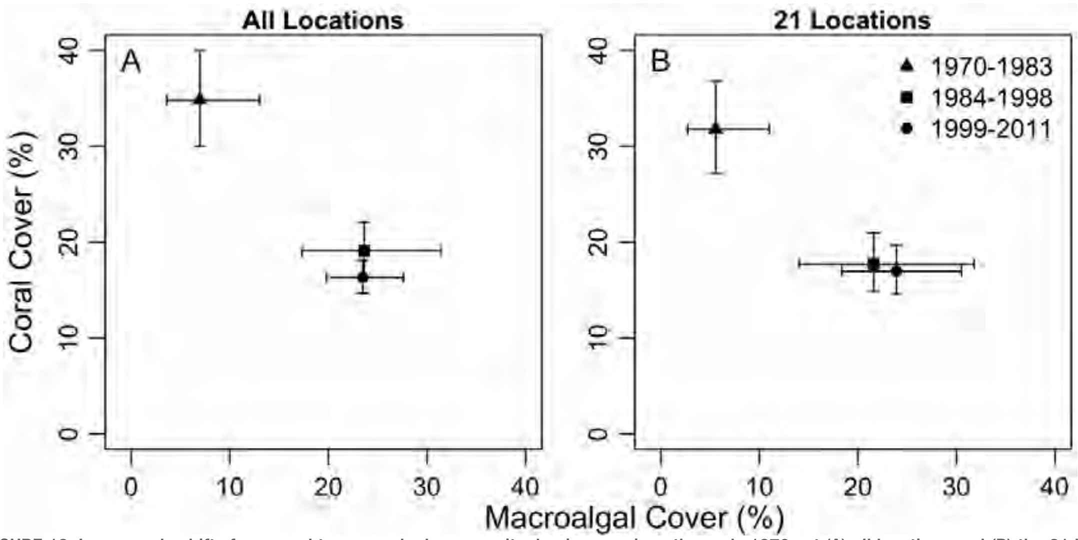

FIGURE 13. Large-scale shifts from coral to macroalgal community dominance since the early 1970s at (A) all locations and (B) the 21 long-term data locations (large circles in Fig. 4). Symbols and confidence intervals represent corrected means and standard deviations for 3 time intervals that take into account variability due to location and datasets using a mixed modeling framework.

Timelines for specific reef sites: These document detailed patterns of variation in coral cover and rates of change on a local scale for 40 particularly well-studied reef sites for which data were available over a span of at least eight years (Table 4, Appendix 2). The earliest of the timelines begins in 1972 but the great majority of sites were not surveyed quantitatively until the 1990s. Average net change in coral cover for the 40 sites is -21% but variation was extreme among sites (range +1 to -64%). Eight sites exhibited remarkable stability with a net change of only +1 to -5% cover. In contrast, four sites declined by ≥ 55% and another six sites by 32% or more.

Whenever possible we chose reefs for plotting timelines for which taxonomic data were available for reef composition at the specific or generic level. Taxa were lumped into eight taxonomic and morphological groups for ease of graphing the data: acroporids (*Acropora palmata* and *A. cervicornis*), other branching corals (principally *Porites* and *Madracis*), agariciids (*Agaricia* and *Helioseris*), *Orbicella* (formerly *Montastraea*) *annularis* species complex, *Montastraea cavernosa*, *Porites astreoides*, and other corals (principally massive species of *Diploria*, *Siderastrea*, and *Colpophyllia*).

Taxonomic data were available for at least some of the surveys from 32 of the 40 sites. The fates of different taxa varied considerably. Species that suffered the greatest proportional losses include most of the former ecologically dominant taxa on Caribbean reefs, including *Acropora palmata* and *A. cervicornis*, branching *Porites* and *Madracis*, the *Orbicella annularis* species complex, and the large plate-like *Agaricia* species. However, most acroporid mortality occurred long before the first surveys at most of the sites. Species that declined the least include species that form massive colonies including the genera *Diploria*, *Siderastrea*, and some *Porites*. Shifts in taxonomic composition are analyzed further in the section on ordination analyses.

Timelines for reef locations with coral cover data for all three time intervals: Twenty-one of the reef locations (clusters of nearby reef sites) enumerated in Table 2 were surveyed at least once before 1984, from 1984 through 1998, and from 1999 to 2011 (Fig. 4,Table 5).

Long-term trends in coral abundance varied greatly among these 21 locations (Table 5) that are grouped into three contrasting patterns of change to highlight their different histories (Fig. 14). Trajectories for nine of the 21 locations (Belize, Costa Rica, Florida Dry Tortugas, Jamaica, and St. Thomas) resemble a hockey stick with steep declines in coral cover between intervals 1 and 2 followed by little change thereafter (Fig. 14A). Proportional losses in coral cover between 1984 and 1998 ranged from 58 to 95% (average 73%). Coral cover at five additional locations (Florida Upper Keys, St. Croix, St. John, Veracruz Mexico, and Vieques Puerto Rico exhibited comparable

TABLE 4. Geographic locations, depths, year span, and net changes in coral cover for detailed timelines for 40 reefs. [1] T. P Hughes original site, now a CARICOMP site.

Place	Reef	Depth (m)	Year span	Start coral cover (%)	End coral cover (%)	Net change (%)
Barbados	Bellairs	3-15	1974-2006	37	15	-23
Belize	Carrie Bow	13	1978-2012	33	11	-22
	Hol Chan	8-11	2005-2012	15	16	+1
Bermuda	Hog Breaker	8	1993-2007	20	20	0
	Twin Breaker	10	1993-2007	25	20	-5
Bonaire	Karpata	10	1974-2008	63	31	-32
	Karpata	20	1974-2008	71	8	-63
BVI – Tortola	Guana Island	8-9	1992-2012	18	14	-4
Cayman Islands	Little Cayman	10-20	1992-2011	28	25	-4
Colombia	Santa Marta	10	1994-2005	33	32	-1
Costa Rica	Cahuita	4-10	1981-2011	40	18	-22
Curaçao	CARMABI Buoy 1	10	1973-2008	36	12	-24
	CARMABI Buoy 1	20	1973-2008	34	10	-24
	CARMABI Buoy 2	10	1973-2008	37	29	-8
	CARMABI Buoy 2	20	1973-2008	35	17	-18
Florida – Upper Keys	Carysfort	0-2	1975-2011	37	3	-34
	Carysfort	14-16	1975-2011	43	4	-39
Florida – Dry Tortugas	Bird Key	13-15	1975-2011	48	10	-38
Jamaica	West	1-18	1977-2012	40	8	-32
	Montego Bay	3-15	1977-2005	47	19	-28
	Rio Bueno	9-18	1978-2010	60	27	-33
	Discovery Bay[1]	9	1977-2011	61	11	-50
	Northeast	1-17	1977-2003	47	12	-35
	Port Royal	5-10	1977-2011	25	5	-20
Mexico	Leeward Cozumel	1-20	1984-2011	25	14	-11
Panama	SE Bastimentos	1	1999-2011	32	24	-8
	SE Bastimentos	9	1999-2011	35	28	-7
	San Blas, Sail Rock	4	1993-1998	18	28	+10
Puerto Rico	La Parguera	10	1994-2012	40	26	-13
Tobago	Bucco Reef	10	1994-2012	24	19	-5
USA – Gulf Mexico	East Flower Garden Bank	20-21	1980-2010	65	55	-10
USVI – St. Croix	Buck Island	7-14	1989-2011	25	6	-19
	Salt River	9-20	1982-2010	25	6	-19
USVI – St. John	Newfound	8	1990-2011	22	6	-16
	Tektite	13	1987-2010	32	28	-4
	Yawzi	13	1987-2011	45	7	-38
USVI – St. Thomas	Black Point	9-14	1979-2010	25	13	-12
	Flat Cay	9-13	1979-2010	65	15	-50
Venezuela – Morrocoy	Cayo Sombrero	5-13	1996-2011	55	39	-16
Venezuela – Los Roques	Dos Mosquises Sur	12	1999-2012	44	25	-19
Summary			**1973-2012**	**38**	**18**	**-20**

proportional decline (50-80%, average 65%) that was spread out more evenly among the three time intervals (Fig. 14B). The third group of seven locations exhibited greater overall stability, although overall mean abundance among these locations differed nearly three fold (Fig. 14C). Coral cover at six of these locations including Barbados, Bonaire, Curaçao, Flower Gardens Bank, San Blas, and Santa Marta declined by just 4-35% over the three time intervals and increased at Bermuda by 35% (Fig. 14C). However, the increase at Bermuda is largely due to more comprehensive sampling of

TABLE 5. Changes in coral and macroalgal cover at the 21 long-term data locations indicated by large yellow circles in Fig. 4. Coral cover data are available for all three time intervals at all 21 locations. Macroalgal data are available for all three of the time intervals for just 9 of the 21 locations. Percent change over the three intervals is expressed as both the absolute change in cover and the proportional change (cover in time interval 3 minus cover in interval 1/cover in interval 1). The pattern of change refers to Fig. 14. P-values are the result of post-hoc comparison of means between the 1st and 2nd time interval and the 2nd and 3rd time interval with significance at the 95% level in bold.

Label	Location	Coral cover (%)					Macroalgal cover (%)							
		1970 - 1983	1984 - 1998	1999 - 2011	Abs change	Prop. loss	1970 - 1983	1984 - 1998	1999 - 2011	Abs. change	Prop. loss	1-2	2-3	Change type
6	Barbados Leeward	26.9	14.7	20.9	-6.0	-0.22	10.5	22.4	6.1	-4.4	-0.4	0.48	0.08	C
11	Belize Central Barrier	33.8	14.1	17.3	-16.5	-0.49	2.8	36.9	43.7	+40.9	+14.6	0.18	0.90	A
16	Bermuda	19.4	21.9	28.8	+9.5	+0.35	NA	8.3	10.6			0.54	0.99	C
56	Bonaire Leeward	54.1	35.2	35.2	-18.9	-0.35	1.0	6.1	15.8	+14.8	+14.8	< 0.01	0.10	A,C
22	Colombia Santa Marta	32.5	30.2	31.8	-0.7	-0.02	19.8	19.8	NA			0.84	0.93	C
23	Costa Rica Cahuita	40.4	11.2	16.1	-24.3	-0.60	NA	NA	NA			< 0.01	0.79	A
28	Curaçao Southwest	43.0	34.6	35.5	-7.5	-0.17	0.1	4.4	5.0	+5.0	+49.0	0.64	0.05	C
84	Florida Upper Keys	32.7	21	6.4	-26.2	-0.80	NA	15.0	15.2			0.06	0.03	B
79	Florida Dry Tortugas	28.9	12.0	10.0	-18.9	-0.65	NA	NA	20.5			< 0.01	0.99	A
42	Jamaica Montego Bay	36.3	8.4	15.3	-21.0	-0.58	NA	56.8	66.4			< 0.01	0.68	A
43	Jamaica North Central	44.6	10.8	14.4	-30.2	-0.67	8.2	57.8	43.6	+35.4	+4.3	< 0.01	0.99	A
44	Jamaica Northeast	47.0	2.5	10	-37.1	-0.78	NA	NA	45.9			< 0.01	0.20	A
46	Jamaica Port Royal Cays	24.9	2.7	2.6	-22.3	-0.90	NA	NA	45.0			< 0.01	0.97	A
47	Jamaica West	40.3	6.5	9.0	-31.3	-0.78	NA	NA	70.3			< 0.01	0.62	A
54	Mexico Veracruz	34.1	28.0	17.2	-16.9	-0.50	NA	NA	NA			0.70	0.18	B
65	Panama San Blas	39.2	19.1	30.9	-8.3	-0.21	NA	69.3	NA			0.06	0.35	C
71	Puerto Rico Vieques	42.6	22.8	12.6	-30.0	-0.70	1.9	4.1	13.5	+11.6	+6.1	0.54	0.80	B
80	USA East Flower Garden Bank	57.5	51.1	55.1	-2.4	-0.04	NA	NA	NA			0.34	0.31	C
85	USVI St. Croix	23.7	20.7	9.1	-14.6	-0.62	1.6	7.2	10.9	+9.3	+5.8	0.28	0.17	B
88	USVI St. John	34.1	26.1	11.8	-22.3	-0.65	0.7	12.7	30.3	+29.6	+42.3	0.19	< 0.01	B
86	USVI St. Thomas	32.5	4.6	13.9	-18.6	-0.57	1.5	NA	35.3	+33.8	+22.5	0.01	0.87	A
	Summary	**36.6**	**19.0**	**19.2**	**-17.4**	**-0.51**	**2.3**	**15.3**	**22.8**	**+19.6**	**+17.7**	**< 0.01**	**0.53**	

a wider diversity of habitats in recent years rather than an increase at earlier monitored sites.

The geographic distribution of the three different patterns of change overlap broadly throughout the wider Caribbean but there are also differences (Fig. 14D). Locations that exhibited the hockey stick pattern of dramatic early decline followed by no change (orange circles) are distributed very widely from Florida in the north to Costa Rica in the south and from Belize in the west to St. Thomas in the east. Locations where coral cover declined progressively through all three intervals (blue squares) are more constrained in latitude and distributed in a band from Veracruz in the west to Vieques, St. Croix, and St. John in the east. Lastly, locations that

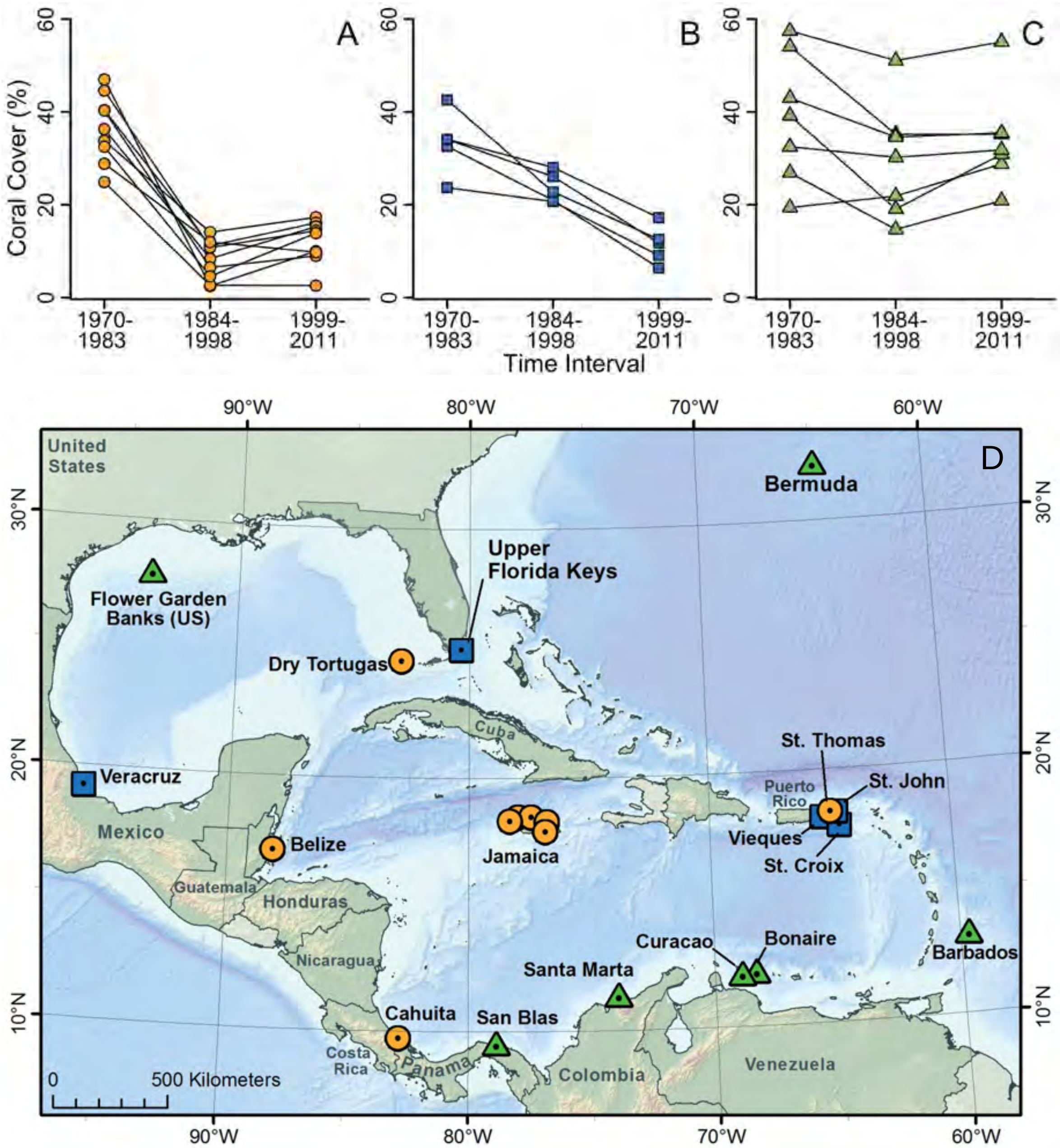

FIGURE 14. Disparate trajectories of coral cover at 21 mapped locations for which data for coral cover are available for all three time intervals (1 = before 1984, 2 = 1984-1998, 3 = 1998-2011). Values are means of percent coral and macroalgal cover averaged over all the data for each location within each time interval. Locations are grouped by eye into thee general categories based on the total amount of change in coral cover over the three time intervals and the tempo of change. (A) hockey stick pattern of 49-90% decline between intervals 1 and 2 followed by little or no change. (B) approximately constant and continuous decline ranging from 50-80% over all three intervals. (C) comparative stability of +35% to -35%. Note that the trajectory for Bonaire is a hybrid of patterns A and C. (D) map showing geographic distribution of the three patterns of change. For further details see text.

TABLE 6. Summary statistics for PCA analyses of coral and macroalgal community composition.

	Percent variation explained			
	PCA 1	PCA 2	PCA 3	sum
Coral + Macroalgae, 21 locations	42.1	17.9	11.8	71.8
Coral + Macroalgae, all locations	41.3	12.7	11.1	65.1
Coral only, 21 locations	24.9	21.8	16.2	62.9
Coral only, all locations	21.8	17.2	13.5	52.5

exhibited the greatest stability in coral cover (green triangles) are concentrated in the extreme south and north of the wider Caribbean plus Bermuda.

The disparate reef histories in Fig. 14 clearly demonstrate the folly of attempting to understand the causes of coral reef decline for the entire Caribbean as a single ecosystem, an approach that ignores the enormous heterogeneity in environments and history of human and natural disturbance among different reef locations. This is even more apparent in timelines of coral cover compiled for individual reef sites (Table 4, Appendix 2). Moreover, locations that suffered the greatest proportional loss in coral cover over the three time intervals (cover interval 1- cover interval 3/cover interval 1) also suffered the greatest absolute loss in cover (cover interval 1-cover interval 3) (Fig. 15). The strong correlation between proportional and absolute decline further strengthens the conclusion that trajectories of change at different locations reflect their unique histories of events rather than some pervasive force throughout the entire wider Caribbean.

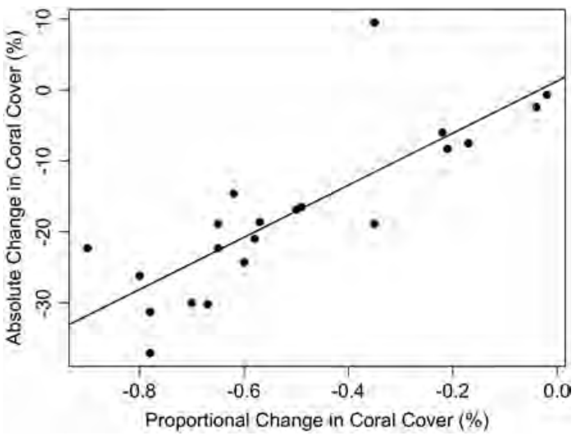

FIGURE 15. Absolute change in percent cover of corals from before 1984 to after 1999 versus the proportional change in coral cover ($R^2 = 0.65$, $p < 0.01$).

Ordination of coral and macroalgal community composition:
We used PCA and MDS to investigate patterns of change in community composition of corals and macroalgae for the entire dataset and the 21 reef locations in Table 5 (Fig. 16-17). Coral taxa were grouped based on an analysis of their average abundance and frequency of occurrence in the overall dataset to minimize zero occurrences in the ordination matrix. The resulting nineteen taxa include: *Acropora cervicornis*, *Acropora palmata*, *Agaricia tenuifolia*, other *Agaricia*, *Colpophyllia*, *Diploria*, *Eusmilia*, *Helioseris*, *Madracis*, *Meandrina*, *Millepora*, *Montastraea cavernosa*, *Mycetophyllia*, *Orbicella* [formerly *Montastraea*] "*annularis*" complex, *Porites astreoides*, other (overwhelmingly branching) *Porites*, *Siderastrea*, *Stephanocoenia*, and other corals. Macroalgae were considered as a single taxon.

Results are presented here for the PCA analyses only. The best results are for the 21 locations with data for coral and macroalgae in the same analysis in which the first three principal components explain more than 70% of the total variance. (Table 6, Fig. 16A-B). The strongest separation along PCA 1 is between macroalgae versus corals. PCA 2 accounts for an additional 20% of the variance reflecting the opposite trends in occurrence of branching *A. palmata* and the *Orbicella annularis* species complex. The same analysis based on all the localities produces a similar pattern but explains less of the total variability (Fig. 16C-D), a difference we attribute to the lack of consistency of locations among time intervals due to very limited sampling at most of the locations in Table 2 and greater variety in reef environments compared to the more restricted analysis.

Ordinations based only upon coral taxa without macroalgae yielded consistently poorer results, underlining the fundamental importance of the phase shift between corals and macroalgae that dominates patterns of change (Fig. 17).

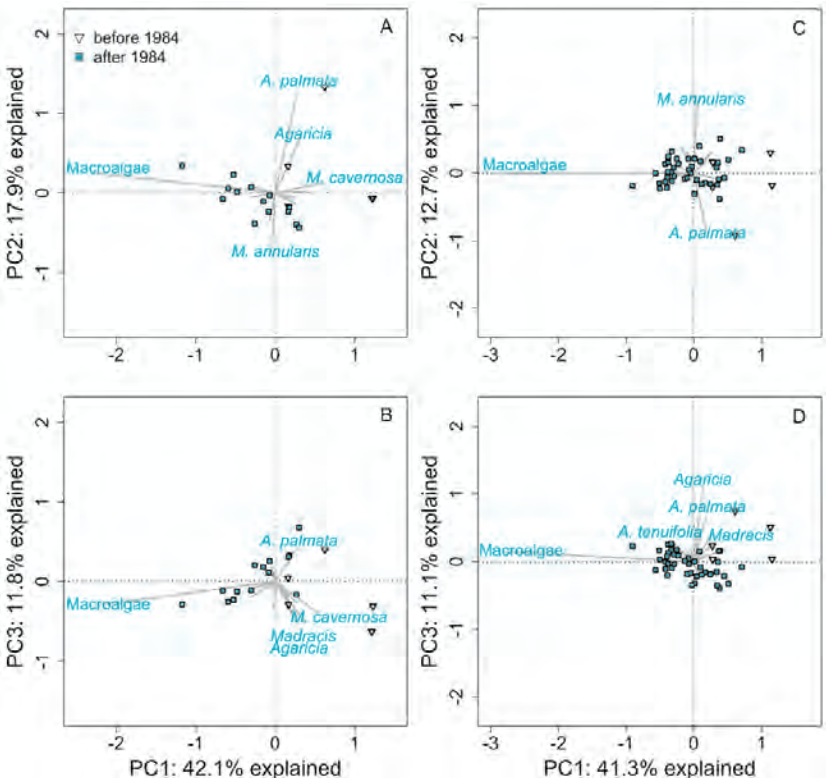

FIGURE 16. Principal components analysis of coral taxa and macroalgae. (A-B) PCA based on all available data for the 16 of 21 locations in Table 5. (C-D) PCA based on all available data for the 44 locations with coral data from more than a single year in Table 2.

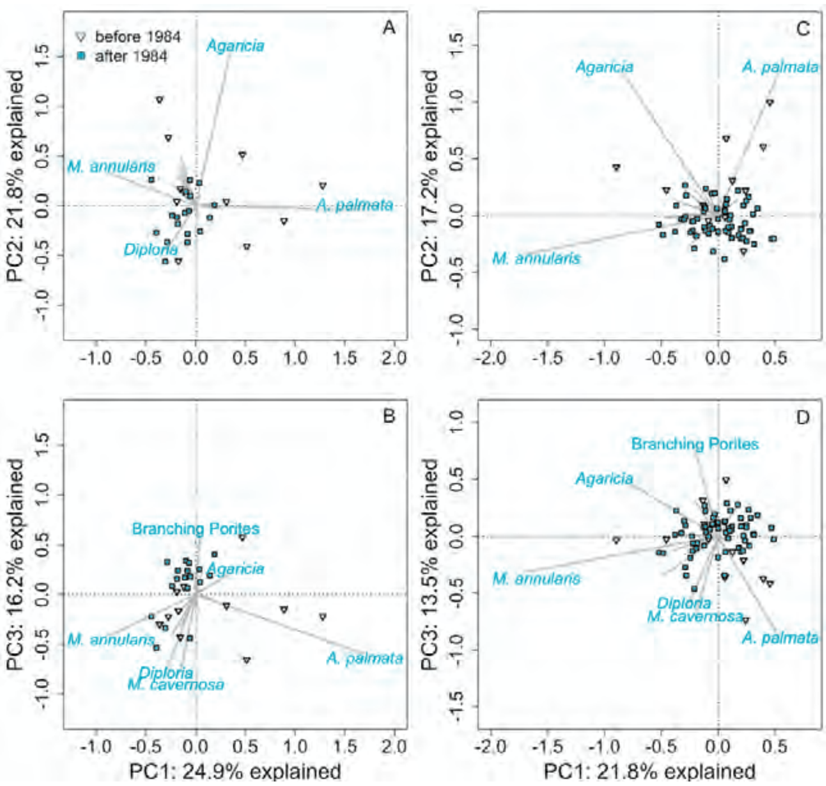

FIGURE 17. Principal components analysis of coral taxa without macroalgae. (A-B) PCA based on all available data for 18 of the 21 locations in Table 5. (C-D) PCA based on all available data for the 64 locations with coral data available at a fine taxonomic level.

2b. EXTREME DECLINE OF FORMERLY DOMINANT SPECIES

Three taxa of formerly great ecological significance on Caribbean reefs suffered massive declines up to several decades before the first quantitative surveys at most of the 90 locations in Table 2. Losses were so great that the species are virtually ecologically extinct; meaning they no longer play any significant ecological role in determining the distribution and abundance of surviving species. Understanding the subsequent decline of Caribbean reefs hinges upon a clear understanding of the magnitude of these early changes that in most places have hardly left a trace.

Decline of *Acropora palmata* and *A. cervicornis*

Acropora palmata and *A. cervicornis* were among the most abundant and ecologically dominant corals on Caribbean reefs in depths down to 20 m for the last one million years until the 1970s and 1980s (Goreau 1959; Geister 1977; Adey 1978; Jackson 1992, 1994; Pandolfi 2002; Pandolfi and Jackson 2001, 2006). Distribution and abundance were highly variable. Nevertheless, the former ecological dominance of *Acropora* is obvious from the composition of Holocene and Pleistocene reef rock, the coral fraction of which is 80-90% *Acropora* by volume in the majority of shallow-water sections (Mesolella 1967; Macintyre and Glynn 1976; Lewis 1984; Liddell et al. 1984; Jackson 1992 and references within; Pandolfi and Jackson 2001, 2006; Aronson and Precht 2001).

Both species experienced intense mortality due to White-Band Disease (WBD) since the mid to late 1970s until today (Gladfelter 1982; Porter and Meier 1992; Aronson and Precht 2001; Porter et al. 2001; Patterson et al. 2002; Weil and Rogers 2011). Hurricanes and outbreaks of predators also devastated acroporids in Jamaica and the USVI in the 1980s (Knowlton et al. 1981, 1990; Woodley et al. 1981; Rogers et al. 1991; Rogers and Miller 2006), and there is strong paleontological evidence for die-offs several decades earlier in Barbados (Lewis 1984), Bocas del Toro, Panama (Cramer et al. 2012), and more broadly throughout the region (Jackson et al. 2001).

Unfortunately, there are remarkably few quantitative data on the abundance of either species until they were already greatly diminished by disease, a spate of hurricanes in close succession, and degrading water quality to be reviewed in the next section. To address this, we compiled a very large qualitative database on the occurrences of both species back into the 19th century to supplement the quantitative data (Appendix 3). The proportion of reef sites with presence and dominance of *Acropora palmata* and *A. cervicornis* was computed for the time period from 1851-2012. Data include qualitative and quantitative information from the primary peer-reviewed scientific literature, government reports, and less commonly historical literature as well as quantitative data received directly from contributors to this study and compiled in the larger GCRMN database. Quantitative data include percent cover for either *Acropora* species, while qualitative data include presence/absence and relative abundance data, as well as descriptions of relative abundance categories (Appendix 3). Data are primarily from underwater field surveys, although a small number are from boat-based observations and high-resolution aerial photographs. Data from the literature were extracted from texts, tables, figures, and maps.

Only data from “reef crest” and “midslope” reef zones were included in the analysis. Generally, reef crest data spanned 0-6 m water depth and midslope data spanned between 6-20 m, as 6 m was the depth at which dominance typically shifted from *A. palmata* to *A. cervicornis* in the quantitative data. However, the distinction between reef crest and midslope was made on a reef site-by-site basis, taking into account additional information on reef zone or reef morphology, if available. For some locations, the cutoff was closer to 10 m, the same value used in Jackson et al. (2001). Data were not included if determination of the reef zone could not be made. Data were recorded at the reef site level and computed by averaging over replicates within the same reef site and reef zone. In total, 1,855 reef sites from 67 locations were compiled for the reef crest zone and 4,543 reef sites from 80 locations for the midslope zone. These included locations that were not represented in the master GCRMN quantitative database (Table 2).

Results are presented in Fig. 18. Sample size is small before 1950, and the locations represented in various time bins are not consistent.

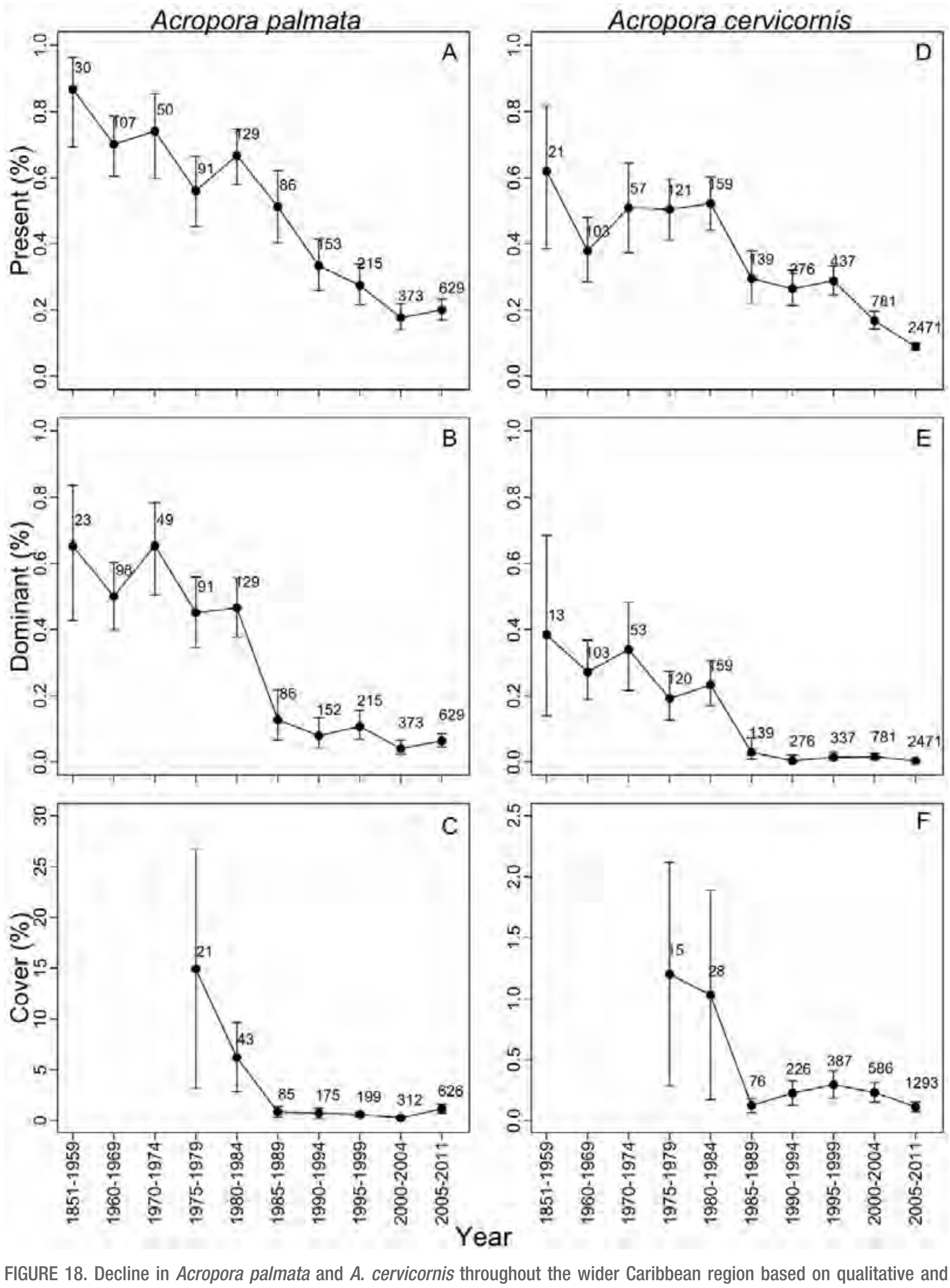

FIGURE 18. Decline in *Acropora palmata* and *A. cervicornis* throughout the wider Caribbean region based on qualitative and quantitative data. (A) frequency of occurrence of *A. palmata*; (B) frequency of reefs at which *A. palmata* was described as the dominant coral; (C) changes in percent cover recorded in the GCRMN quantitative database; (D) frequency of occurrence of *A. cervicornis*; (E) frequency of reefs at which *A. cervicornis* was described as the dominant coral; (F) changes in percent cover recorded in the GCRMN quantitative database. Sample size (numbers of sites) indicated adjacent to points. Confidence intervals are 95% binomial intervals for A, B, D, and E and standard errors for C and F.

Nevertheless, the data confirm the remarkably great abundance of both species before the 1970s. *Acropora palmata* was present at more than 80% of all areas surveyed in depths less than 10 m throughout the wider Caribbean region and was recorded as "dominant" at 60% of these localities Fig. 18 A-C. The data also suggest that the decline in *A. palmata* occurrence and dominance began in the 1960s in accord with the paleontological data (Lewis 1984; Cramer et al. 2012). The patterns for *A. cervicornis* are similar to *A. palmata* (Fig. 18D-F). The species was present at 60% of all localities surveyed before 1959 and dominant at nearly 40% of them. As for *A. palmata*, there is a suggestion that dominance began to decline before the 1970s but there are less data than for *A. palmata*. Quantitative data are extremely sparse showing a maximum average percent cover of just over 1% although some reefs were still blanketed by about 50% cover, emphasizing the dearth of quantitative data before the late 1980s.

Decline of *Diadema antillarum*

Diadema antillarum was variably abundant on Caribbean reefs until 1983 when it rapidly suffered mass mortality from an unidentified pathogen throughout its range in the tropical western Atlantic (Lessios et al. 1983; Lessios 1988). Reported densities before the die-off ranged from a low of about 1/m^2 to a spectacular 90/m^2 in a harbor at Discovery Bay Jamaica. Hughes et al. (2010) compiled all the available data from the literature for trends in *Diadema* abundance since the earliest quantitative surveys to present. We supplemented their analysis with additional data from the GCRMN database with essentially similar results (Fig. 19). Average density throughout the region was about 8-10/m^2, declining to near zero between 1983 and 1984. Average density remained extremely low throughout the second time interval (1984-1998), and then rose almost imperceptibly during the third period. However, some locations have densities today back up to 3-5/m^2 (Edmunds and Carpenter 2001; Carpenter and Edmunds 2006; Idjadi et al. 2010; Vardi 2011).

Parrotfish abundance and biomass

Reef fishes were overfished before the middle of the 20th century throughout large areas of the Caribbean including especially Jamaica and the USVI (Duerden 1901; Thompson 1945; Randall 1961, 1963; Munro 1983; Hughes 1994; Hay 1984; Jackson 1997). This was decades before the first underwater quantitative surveys in the late 1970s and 1980s. Reef fishes were still reported to be abundant at many remote localities such as the Belize Barrier Reef (Lewis and Wainwright 1985; Lewis 1986), but the once large schools of large bodied groupers and parrotfishes had mostly disappeared. Fishing prior to the 1970s was mostly artisanal using small nets and fish traps. Parrotfish were not specifically targeted but their wide bodies made them particularly vulnerable to traps (Johnson 2010).

We examined parrotfish abundance since 1988 in two ways. In the first case, we compiled all the quantitative data on parrotfish biomass in the GCRMN database after the year 2000 to examine the frequency distribution of biomass (Table 2; Fig. 20). Mean parrotfish biomass taking into account differences among locations and datasets was only 14g/m^2, a small fraction of the highest Caribbean value recorded of 71 g/m^2 and an even smaller percentage of their abundance on protected Indo-Pacific reefs (Sandin et al. 2008a).

Time series of parrotfish biomass longer than ten years are available for only three locations at St. John USVI, Guadeloupe, and Bonaire (Fig. 21).

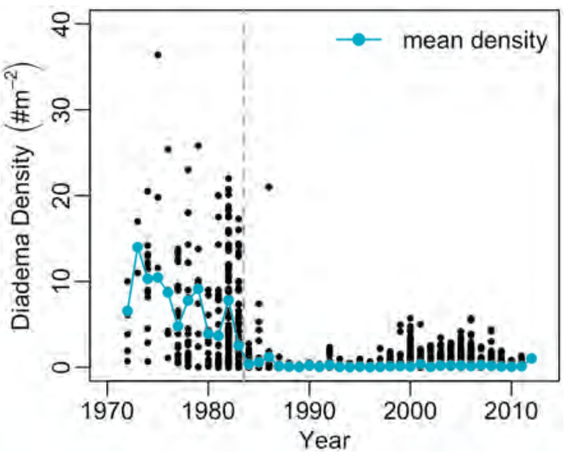

FIGURE 19. Abundance of *Diadema antillarum* throughout the wider Caribbean from 1972 to 2012. Densities of > 25 /m^2 before 1983 are not shown and are not included in average densities. Studies that intentionally surveyed aggregations were not included.

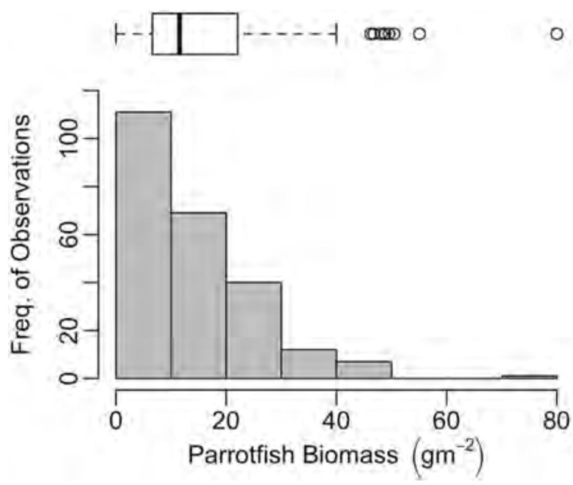

FIGURE 20. Frequency distribution of parrotfish biomass based on all available data after the year 2000 in the GCRMN database (1988-2012) with box plot reflecting the median and the first and third quantiles and dots for outliers.

Values at St. John have been extremely low, hovering around 5-10 g/m^2 since the beginning of the surveys in 1988 in accordance with Randall's (1961) much earlier work in the 1950s when he observed the USVI were already severely overfished (Fig. 21A). In contrast, parrotfish biomass at Guadeloupe also started off at around 10 g/m^2, but has since gradually increased for unknown reasons to between 25-30 g/m^2 (Fig. 21B). Bonaire exhibits a strikingly different pattern (Fig. 21C). Parrotfish biomass in 2003 was similar to the highest recorded in the Caribbean (71 g/m^2) but has since plummeted to less than 30 g/m^2 due to recent targeted fishing on parrotfishes (Fig. 24C; Steneck and Arnold 2009; Steneck et al. 2011).

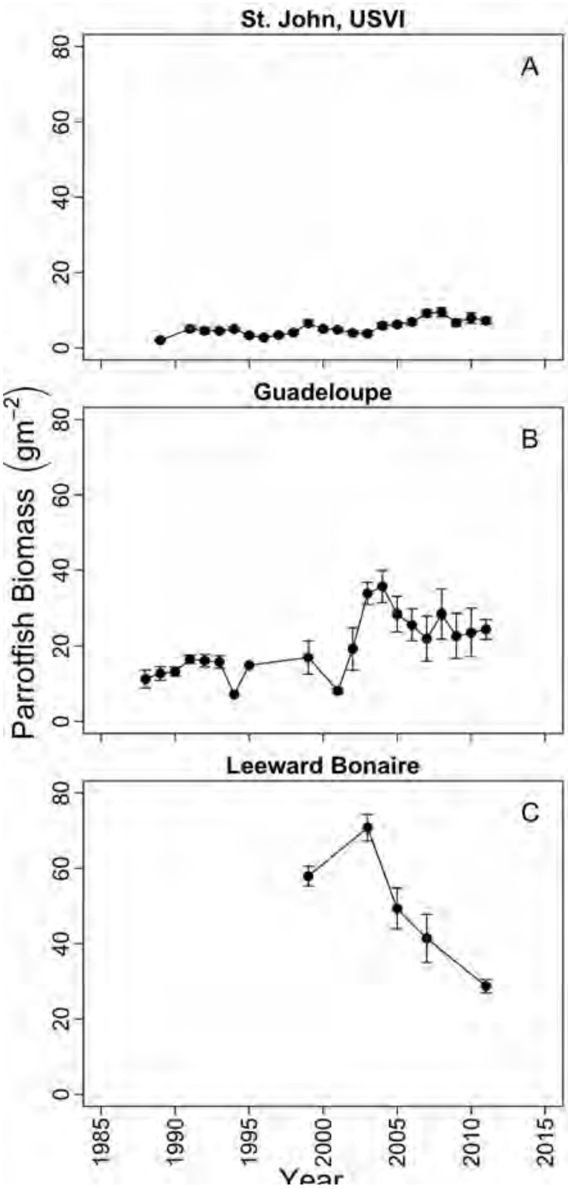

FIGURE 21. Trends in parrotfish biomass since 1988 at (A) St. John, (B) Guadeloupe, and (C) Bonaire. Error bars represent one standard error.

3. ANTHROPOGENIC DRIVERS OF CORAL REEF DEGRADATION

The ultimate driver of environmental degradation is human population growth coupled with inevitable increases in resource consumption, pollution, and habitat degradation as humanity clamors to feed, clothe, and satisfy 7 billion people and counting (Meadows et al. 1972; Vitousek et al. 1997; Wackernagel et al. 2002). But population alone is seldom a good predictor of environmental condition because of enormous disparities in consumption, cultural traditions, and the ways people exploit, pollute, and directly physically impact their natural environment.

Another major difficulty in deciphering cause and effect is the common failure to distinguish between potential drivers of coral decline (overpopulation, overfishing, coastal pollution and development, rising temperatures due to the burning of fossil fuels, introduced species, etc.) and their effects (increases in macroalgae, coral bleaching, coral disease) (Hughes et al. 2010). This problem is especially vexing in the case of coral diseases that have exploded since the first reports of their occurrence in the 1970s (Antonius 1973; 1977; Gladfelter 1982).

Coral diseases have taken a dreadful toll and it is easy to forget that their sudden emergence is almost certainly the result of some combination of anthropogenic stressors (introductions of exotic pathogens, eutrophication, warming, increases in macroalgae, etc.) rather than a natural force. Nevertheless, because of their great and increasing importance, we have treated coral diseases separately in section 3f. A similar confusion exists for the role of hurricanes that are natural phenomena but have been hypothesized to have increased in occurrence and intensity due to climate change. This postulated increase has been blamed for the failure of reefs to recover from the storms. We therefore treat the role of hurricanes separately in Section 3g.

Here we attempt to identify the major drivers of Caribbean coral reef decline by analyzing different anthropogenic stressors one at a time. The results are obviously a work in progress, but we believe they are remarkably clear in identifying the major factors responsible for reef degradation to date and ways in which the nature of stresses on Caribbean reefs are likely to change over the next few decades. Future analyses for publication in the scientific literature will examine all of the drivers together by multivariate analysis.

3a. POPULATION DESITY OF RESIDENTS AND VISITORS (TOURISTS)

We compiled data from the World Wide Web on the average numbers of residents and visitors per year over the past decade at 34 and 29 Caribbean reef locations respectively (Table 7). Considerable effort was invested in locating data for coral reef regions and not for entire countries, as has been the general practice for studies of the impact of people on coral reefs. It is meaningless to compare an ecological metric such as coral cover for an archipelago like Los Roques Venezuela to the entire population of Venezuela, or the condition of the Florida Keys to the population of the entire State of Florida. In general, data for numbers of residents was more forthcoming than for visitors, and in several cases numbers of visitors were combined for two or more different locations such as St. Thomas and St. John because of the lack of an international airport on St. John. Population densities were calculated for land area rather than reef area because the topographic data for land area are more reliable and precise due to satellite mapping. In contrast, published estimates of reef area commonly vary several-fold for the same location because of different definitions of what constitutes a reef and different technologies and methods of observation employed.

TABLE 7. Numbers of visitors, residents, the numbers/km^2 and the most recent coral cover for 34 Caribbean locations plus Bermuda. Data compiled from the World Wide Web based on censuses of residents and tourist board and industry data for numbers of visitors. Most data are for the past five to seven years.

Name	Land area (km^2)	# of residents	Resident density (km^{-2})	# of visitors	Visitor density (km^{-2})	Total population	Total density (km^{-2})	Coral cover (%)
Antigua and Barbuda	443	88000	199	842689	1902	930689	2101	3.8
Bahamas	13940	313312	22	1528000	110	1841312	132	11.7
Barbados	431	284589	660	575000	1334	859589	1994	15
Bermuda	53	67837	1280	306000	5774	373837	7054	38.6
Bocas del Toro	250	13000	52					13.6
Bonaire	294	14006	48	74342	253	88348	301	37.1
British Virgin Islands	153	23552	154	351408	2297	374960	2451	14.3
Cayman Brac	36	1500	42					14.4
Cayman Islands Total	259	56649	219	321650	1242	378299	1461	27
Corn Islands	13	6626	510	50000	3846	56626	4356	24.4
Cozumel	484	79535	164	4000000	8264	4079535	8429	12.1
Curaçao	444	141766	319	419621	945	561387	1264	31.5
Dominica	724	73126	101	354189	489	427315	590	9
Florida Upper Keys	59	19990	339	1185213	20088	1205203	20427	6.1
Grand Cayman	197	56949	289					30.7
Grenada	344	110000	320	360220	1047	470220	1367	12.8
Guadeloupe	1628	452776	278	693000	426	1145776	704	18.6
Kingston Harbor	1645	1184386	720	226164	137	1410550	857	4.7
Little Cayman	26	200	8					24.5
Los Roques	41	1800	44	70000	1707	71800	1751	31
Florida Lower Keys	272	67883	250	2205047	8107	2272930	8356	10.3
Martinique	1128	436131	387	487359	432	923490	819	17.4
Middle Keys	25	10255	410	254585	10183	264840	10594	8
Montego Bay Jamaica	595	184662	310	863214	1451	1047876	1761	19.4
San Andrés, Colombia	57	75000	1316	377619	6625	452619	7941	12.6
San Blas	337	15541	46					30.9
St Ann Jamaica	1213	173232	143	895296	738	1068528	881	19.6
St Bart	21	8902	424	200000	9524	208902	9948	10.8
St Croix	215	50601	235	236000	1098	286601	1333	4.7
St Kitts Nevis	261	50726	194	636924	2440	687650	2635	11.1
St Lucia	617	174000	282	931222	1509	1105222	1791	10.1
St Thomas	81	51634	637	2040900	25196	2092534	25834	13.6
St. Vincent and Grenadines	389	120000	308	199753	514	319753	822	19.5
Tobago	300	60874	203	450000	1500	510874	1703	19.1

Population densities of residents varied 165-fold, from a low of 8 persons/km^2 at Little Cayman to a high of 1,316/km^2 at San Andrés Colombia with 24 of the locations between 100-1000/km^2 (median = 264/km^2). Variations in numbers of visitors per year are even more extreme, ranging 229-fold from 110 persons/km^2 in the Bahamas to an astounding 25,196/km^2 at St. Thomas. Seventeen of the 29 locations have between 1001 to 10,000 visitors/km^2/year (median = 1500/km^2/year). Nine of the locations had less than 1000 visitors/km^2 and 2 have more than 20,000/km^2.

Coral cover is plotted against the density of residents and annual density of visitors in Fig. 22. We used 2 by 2 contingency table analysis explore the relationship between human population density and coral cover. Boundaries of the four quadrants were determined by median values of coral cover and by median densities of residents and tourists. Coral cover is significantly negatively correlated with both the density of residents (N = 34, X^2 = 7.5, df = 1, p = 0.01) and the density of visitors per year (N = 29, X^2 = 5.99, df = 1, p = 0.01). Moreover, all the locations with more than 2,635 visitors/km^2/year have only 6.1 to 13.6% cover.

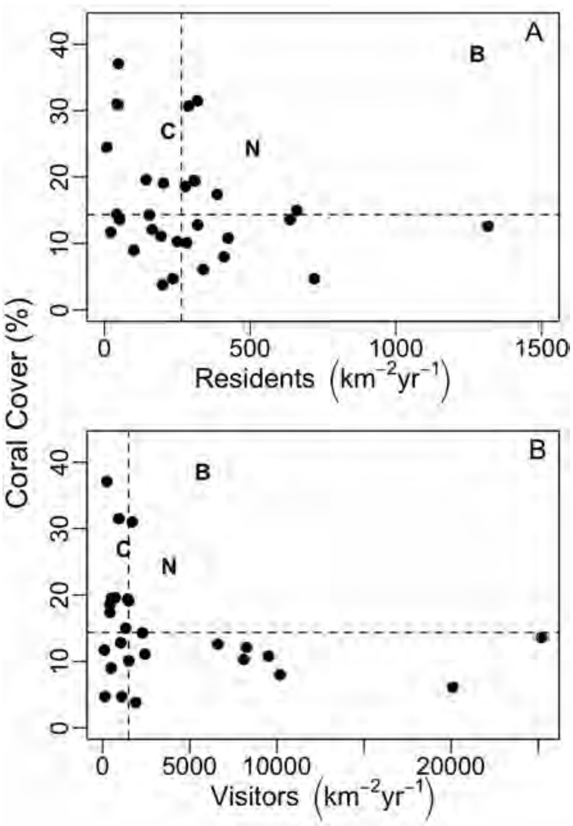

FIGURE 22. Coral cover in relation to human population density. (A) Numbers of residents/km^2, (B) numbers of visitors/km^2/year. B = Bermuda, C = Cayman Islands, N = Corn Islands Nicaragua

Most tourists to Caribbean locations never get beyond the swimming pool to even see a coral reef so their impact on reefs is indirect. The most important indirect effects include runoff of sediments due to unregulated coastal development of roads and hotels, dredging harbors for yachts and gargantuan cruise ships, and nutrient pollution from runoff from golf courses and untreated or minimally treated sewage from hotels, cruise ships, and cesspits (see references in the section on water quality below). In this light, the remarkably high coral cover at Bermuda despite very high densities of visitors and residents is almost certainly a result of strongly enforced environmental and fisheries regulations as discussed below.

3b. FISHING

Artisanal coral reef fisheries are traditionally among the most important sources of protein and livelihood throughout Caribbean coastal communities (Jackson 1997; Hardt 2009). As populations have grown, however, overfishing has resulted in the widespread collapse of reef fish stocks with dire consequences not only for peoples' livelihoods and nutrition but also the ecological condition of coral reefs (Duerden 1901; Thompson 1945; Munro 1983; Hughes 1994; Hawkins and Roberts 2003). The ecological consequences of overfishing are complex and depend on a host of factors including the types of fishing gear employed, the variety of species exploited, trophic cascades, interactions with other kinds of human disturbance, and the unique environmental characteristics of different reefs (Jackson et al. 2001; Mumby et al. 2006, 2007, 2012; Estes et al. 2011). Nevertheless, overfishing is strongly correlated with ecological collapse of reef ecosystems as defined by decrease in coral cover and recruitment and increases in macroalgal abundance and coral disease (Hughes 1994; Sandin et al. 2008a).

The ecological consequences of overfishing in the Caribbean for coral reef communities are most clearly associated with reductions in the abundance and sizes of herbivores, most importantly parrotfishes, surgeonfishes, and sea urchins. Innumerable experiments have shown that exclusion or removal of these grazers results in explosive increases in the abundance of macroalgae (Randal 1961; Lewis 1986; Lirman 2001; Hughes et al. 2007) that potentially compete with corals in numerous ways discussed below. This is perhaps most obvious in the Caribbean where the mass mortality of the sea urchin *Diadema antillarum* coupled with the overfishing of parrotfishes has resulted in the large increases in macroalgal abundance documented in the previous section (Table 3; Figs. 11-13).

Fishing pressure and the state of reef fish populations varied greatly among Caribbean reef locations in the 1970s due to a complex mix of fishing practices, economic well being, and cultural traditions that are beyond the scope of this report. Nevertheless, certain patterns are clear. Densely populated West Indian islands with a long tradition of sugar economies based upon slavery (e.g., Jamaica, Barbados, Hispaniola, the Virgin Islands, and most of the Lesser Antilles) developed labor intensive artisanal fisheries based largely on the use of fish traps and small nets that resulted in extreme overfishing by the early 20th century (Duerden 1901; Thompson 1945; Randall 1963; Munro 1983; Jackson 1997; Hawkins and Roberts 2003; Hardt 2009). In contrast, continental reefs along the coasts of Florida, Mexico, the Mesoamerican Barrier Reef, and northern South America were generally less densely populated and less heavily fished until the 1970s to 1990s (Goode 1887; Jackson 1997; McClenachan 2008).

Diadema abundance before 1984 as a proxy for historical fishing pressure

Most of the historical information on overfishing is anecdotal or qualitative and there were very few hard scientific data to back them up until Mark Hay (1984) conducted a comparative study from 1980-1982 on the intensity of grazing by the sea urchin *Diadema antillarum* versus grazing by parrotfishes and surgeonfishes at several locations across the wider Caribbean. Densities of *Diadema* on eight heavily fished reefs ranged from 5-20/m^2 (median = 10) versus 0 to 8/m^2 (median = 1) on less fished reefs (Hay 1984, his Table 2, $F_{1,12} = 20.7$, $p < 0.01$).

Hay did not count herbivorous fishes but instead used strips of the seagrass *Thalassia testidinum* as standardized “baits” to measure rates of herbivory. Rates of consumption of *Thalassia* bait by fishes and *Diadema* were inversely proportional in relation to the extent of overfishing on the reefs. Fish consumption of bait on lightly fished reefs in Belize, Panama, Honduras, and a protected area in the US Virgin Islands was 5-10 times higher than on heavily overfished reefs in Haiti and the US Virgin Islands. Consumption and abundance of *Diadema* showed the opposite pattern, with very little consumption on less fished reefs and high consumption exceeding that by grazing fishes on overfished reefs.

There is also considerable evidence from ecological surveys and from natural and manipulative experiments for intense competition for food between *Diadema* and grazing fishes, especially parrotfish. Abundance of *Diadema* and grazing fishes were inversely proportional across a depth gradient on reefs near Carrie Bow Cay in Belize (Lewis and Wainwright 1985). *Diadema* were most abundant in the high spur and groove habitat (4.3/m^2) where the high habitat relief likely provided better protection from predators than less complex habitats. In contrast, parrotfish abundance was only 0.07/m^2, the lowest in any of the five reef habitats surveyed. *Diadema* abundance was extremely low (0.1 to 0.7/m^2) in all the other reef zones where parrotfish abundance ranged from 0.09-0.32/m^2.

Hay and Taylor (1985) strengthened the evidence for strong competition between *Diadema* and parrotfish in two *Diadema* removal experiments at St. Thomas and St. Croix that were conducted just before the *Diadema* die-off occurred. Numbers of parrotfish at two control (non-removal) sites at St. Thomas were 0.02 and 0.04/m^2 versus 0.18/m^2 at the removal site (Kruskal-Wallace Test, $p < 0.05$). Similarly at St. Croix, there were 0.08 parrotfish/m^2 at the single control site versus 0.29/m^2 at the removal site (Kruskal-Wallace Test, $p < 0.05$). These patterns were confirmed by surveys before and after the die-off of *Diadema* within four reef zones at Tague Bay, St. Croix (Carpenter 1990b). Numbers of parrotfish increased 3.9-fold from 0.17/m^2 transect before the die-off to an average of 0.66/m^2 afterwards on the backreef and reef crest. Similar comparisons for the three forereef zones surveyed showed a 2.8-fold increase from 0.29 to 0.81 parrotfish/m^2 at 2 m; a 2.3-fold increase from 0.25 to 0.57/m^2 at 5 m; and a 4.1-fold increase from 0.17 to 0.67/m^2 at 10 m. All of these differences were significant by 1-way ANOVA at $p < 0.0001$.

Summarizing the above, Hay’s (1984) study confirmed that overfishing on many Caribbean reefs occurred before the mass mortality of *Diadema* in 1983, a fact consistent with Jack Randall’s (1961,

1963) pioneering investigations in the 1950s and all of the historical data (Duerden 1901; Thompson 1945; Munro 1983; Jackson 1997; McClenachan 2008; Hardt 2009). But this is difficult to document beyond Hay's and the other specific study sites because there are virtually no quantitative survey data on the biomass of Caribbean herbivorous reef fishes prior to 1988 (Tables 1 and 2, Fig. 6).

What we can do, however, is to use the patterns of *Diadema antillarum* abundance prior to 1984 as a proxy for historical fishing pressure based upon (1) the well-documented inverse correlation between *Diadema* abundance and herbivorous fish abundance prior to the mass mortality of *Diadema* in 1983 (Ogden et al. 1973; Hay 1984, Lewis and Wainwright 1985), (2) the increase in herbivorous fish abundance after the die-off of *Diadema* in 1983 (Carpenter 1990a, b; Robertson 1991), and (3) Hay and Taylor's (1985) *Diadema* removal experiments. Besides all of the above, we know of no evidence to suggest that *Diadema* abundance was not inversely proportional to fishing pressure. Thus the proxy relationship is robust.

Contrasting fates of reefs since 1984 in relation to historical fishing pressure

Data on *Diadema* density/m^2 before 1984 were available for 16 of the 21 reef locations in Table 5 and Fig. 14 (Table 8, Appendix 4). *Diadema* densities ranged from a low of 0.5/m^2 at San Blas, Panama to a high of 12.4/m^2 at the Port Royal Cays, Jamaica. There is a clear break in the values between reefs in San Blas, Bermuda, the Upper Florida Keys, Bonaire, Belize, Curaçao, and Cahuita Costa Rica (0.5/m^2 to 3.8/m^2, median = 1.5/m^2, classified here as "less fished" reefs) versus reefs in Barbados, Jamaica, and the US Virgin Islands (6.9/m^2 to 12.4/m^2, median -9.1/m^2, classified here as "overfished" reefs, t-test: t = 9.0, df = 13.6, $p < 0.01$). These values correspond closely with what is known qualitatively about fishing pressure at these locations before 1984 (Appendix 5).

We conducted a linear mixed-effects model analysis to compare median coral cover between "less fished" versus "overfished" reefs based on the density of *Diadema* at the 16 locations before the 1983 die-off (see methods section for model formulation). As expected, there was no significant correlation between coral cover at less fished and overfished

TABLE 8. Data for the analysis of the effects of historical and recent fishing pressure on coral cover for the 16 locations in Table 5 with *Diadema* data from before the die-off.

Label	Location	*Diadema* density (#/m^2)	Parrotfish biomass (g/m^2)	Long –term prob. hurricane	# of hurricanes since 1984	Coral cover since 2005 (%)
6	Barbados Leeward	11.2		0.06	0	15.0
11	Belize Central Barrier	1.7	7.2	0.06	3	15.0
16	Bermuda	0.6	21.9	0.12	4	38.6
56	Bonaire Leeward	1.5	32.3	0.02	0	37.1
23	Costa Rica Cahuita	3.8	39.8	0.00	0	18
28	Curaçao Southwest	3.0	15.2	0.02	0	31.5
84	Florida Upper Keys	1.2	20.3	0.15	2	6.1
42	Jamaica Montego Bay	7.1	4.6	0.08	3	19.4
43	Jamaica North Central	6.9	6.9	0.10	3	19.6
44	Jamaica Northeast	7.9	5.4	0.09	4	
46	Jamaica Port Royal Cays	12.4		0.05	3	4.7
47	Jamaica West	9.2	8.1	0.06	3	7.8
65	Panama San Blas	0.5	13.3	0.00	0	
85	USVI St. Croix	7.0	13.1	0.08	5	4.7
88	USVI St. John	9.1	8.3	0.11	5	10.1
86	USVI St. Thomas	9.8	11.4	0.12	3	13.6
	Mean	**5.8**	**14.8**	**0.07**	**2.4**	**17.2**

locations before 1984 (Fig. 23A; GLMM (generalized linear mixed model) p=0.19) because high *Diadema* abundance compensated for the low abundance of herbivorous fish. But this changed after the 1983 mass mortality of *Diadema* when median values of coral cover significantly diverged between less fished and overfished reefs (Fig. 23B-C; GLMM p<0.01). Similar results were found for coral cover since 2005 (Fig. 23D; GLMM p=0.01). There is also a significant difference between time periods 1 and 3 in the proportional loss in coral cover between "less fished" locations (median = -35%, range +35% to -80%) and "overfished" locations (median = -65%, range -22% to -90%) ($F_{1,14}$ = 4.96, p = 0.04). Comparisons between time periods 1 and 2 and 1 and 4 are not significant but all of the trends are in the same direction.

Data for macroalgae are too incomplete for meaningful statistical comparison of changes in macroalgal abundance between "less fished" and "overfished" reefs although the trends are consistently in the expected direction with two to three times higher macroalgal cover at locations that had been earlier overfished (Table 8, Fig. 23 D-E).

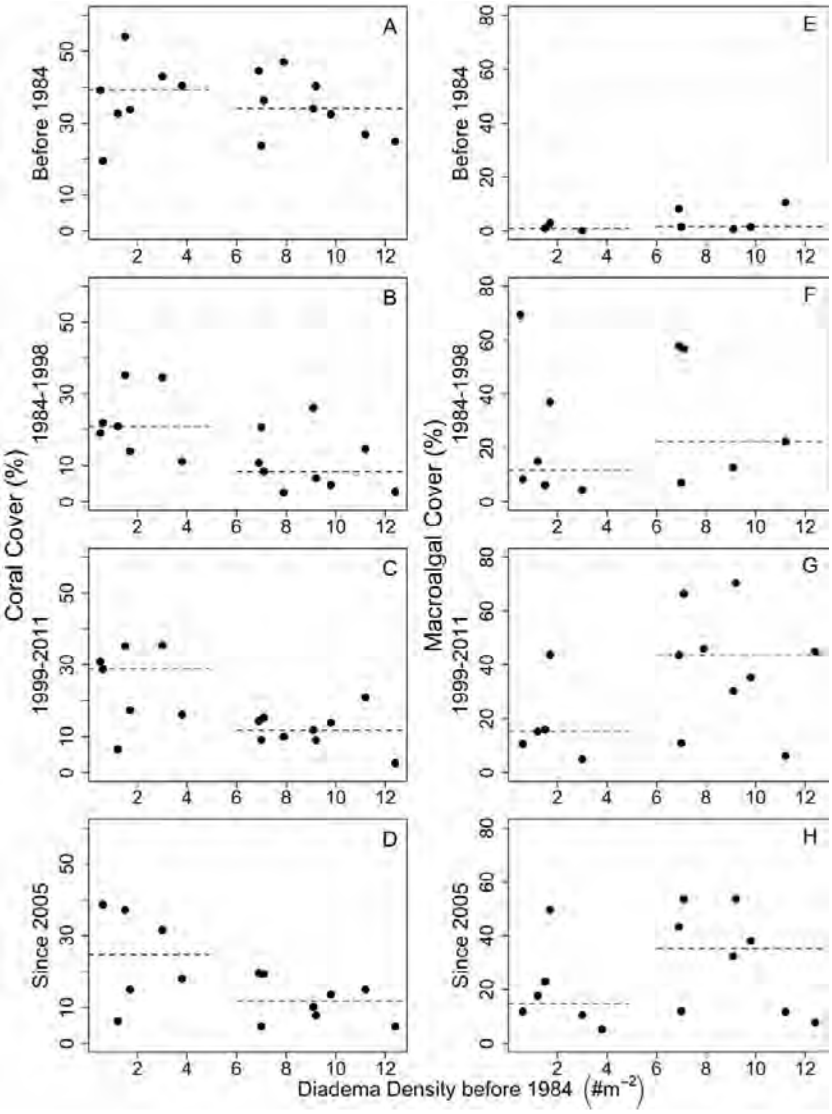

FIGURE 23. Percent coral (A-D) and macroalgal (D-E) cover in relation to the abundance of *Diadema antillarum* before the die-off in 1983 for the 16 reef locations in Table 8. (A) coral cover 1970-1983 (N = 16, median = 39.2/m^2 versus 34.1/m^2, p = 0.19), (B) coral cover 1984-1998 (N = 16, median = 21.0/m^2 versus 8.4/m^2, W = 53, p < 0.01), (C) coral cover 1999-2011 (N = 16, median = 28.8 versus 11.8, p = 0.02), and (D) coral cover since 2005 (N = 15, median coral cover 31.5/m^2 versus 10.1/m^2 p = 0.01). (E) macroalgal cover 1970-1983 (N = 8, median MA cover 1.0 versus 1.6%), (F) from 1984-1998 (N = 11, median MA cover = 11.7 versus 22.4%), (G) from1999-2011 (N = 14, median MA cover = 15.2% versus 43.6%), and (H) since 2005 (N = 14, median MA cover = 14.8% versus 35.1%).

The role of parrotfish today

Further support for the harmful consequences of overfishing herbivorous fishes upon coral cover is apparent from the positive and negative correlations on reefs today between the abundance of grazing parrotfishes and the percent cover of corals and macroalgae for all of the locations in Table 9 with paired data (Fig. 24). The non-parametric correlation is significant for coral cover versus parrotfish (Fig. 24A) but not for macroalgae versus parrotfish due to the smaller sample size for macroalgae and considerably greater scatter in the data (Fig. 24B). Results were opposite for the smaller dataset of 16 reefs (Fig. 24 C-D).

Generally speaking, the few reefs in the upper quartile of parrotfish biomass today (locations with > 15.6g/m^2) have significantly more coral than reefs where parrotfish are less abundant (t = 2.24, df = 60.7, p = 0.03). This is especially evident when the relationship between parrotfish biomass and macroalgal cover is compared with a linear-mixed model between locations that had been overfished before 1984 (high *Diadema* abundance) and those that were not (Fig. 24D). All of the historically overfished localities have low parrotfish biomass and low coral cover (Fig. 24C), and macroalgal cover is significantly greater than at less fished locations (t = -2.12, df = 36.2, p = 0.03), and strongly negatively correlated to parrotfish abundance (Fig. 24D, β = -1.14, SE = 0.40, p < 0.01).

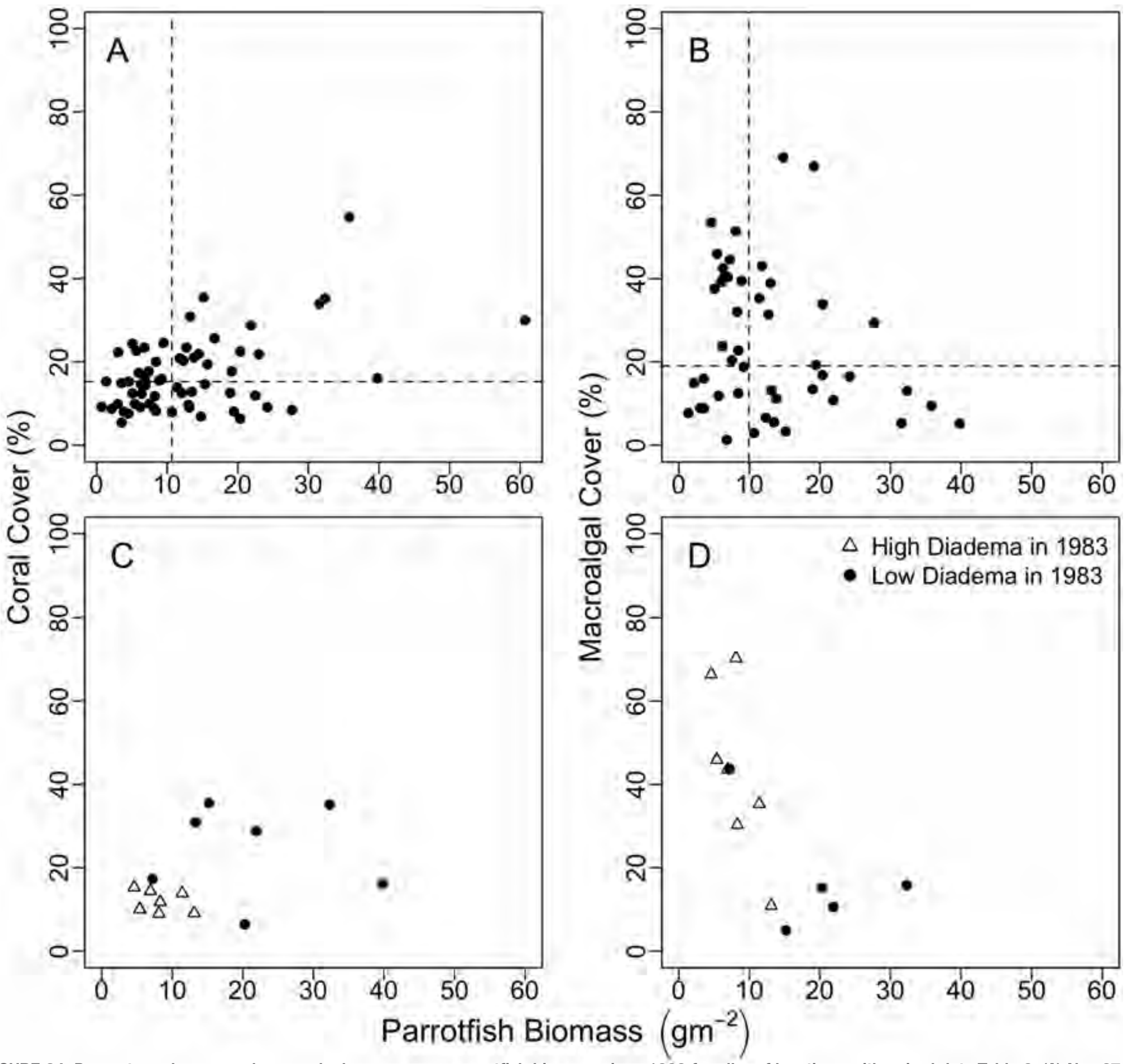

FIGURE 24. Percent coral cover and macroalgal cover versus parrotfish biomass since 1998 for all reef locations with paired data Table 9: (A) N = 67, r_s = 0.31, p = 0.02; (B) N = 46, r_s = -0.19, p = 0.20. The same analysis for only the16 locations in Table 8: (C) r_s = -0.36, p = 0.21); (D) r_s = -0.81, p = 0.01). Dashed lines in (A) and (B) are medians for each axis.

Indirect effects of fishing due to increased macroalgal abundance

Reduction in coral recruitment

Approximately one quarter of Caribbean corals are brooding species that produce comparatively small numbers of large planula larvae that disperse short distances from their parents. Larval mortality is lower than for broadcasting species and populations may be effectively self-seeding (Jackson 1991). In contrast, the remaining broadcasting species spawn gametes that are fertilized in the ocean and larvae disperse farther and suffer higher mortality before settlement than brooding species so that there is little or no correlation between adult abundance and recruitment at different sites (Jackson 1991).

Numbers of coral recruits before the *Diadema* mortality were overwhelmingly dominated by brooding species that comprised > 90% of all recruits (Bak and Engel 1979; Rylaarsdam 1983; Rogers et al. 1984; Hughes and Jackson 1985). Recruits of brooding species are still more abundant than broadcasting species, but rates of coral recruitment have greatly declined. The most extensive long-term data are from Jamaica and Curaçao (Hughes and Tanner 2000; Vermeij 2006). Cover of two brooding species in Jamaica (*Agaricia agaricites* and *Leptoseris cucullata*) declined 83% between 1977 and 1993 while the average number of recruits/year declined 75%. Thus the decline in recruitment was similar to the loss in coral cover. This was not the case in Curaçao, however, where coral cover declined by 50% but coral recruitment in identical settlement panel experiments declined by more than 80% between 1979-1981 and 1998-2004 so that the decline in recruitment cannot be attributed entirely to a simple decline in the abundance of parental colonies.

The differences on the upper surfaces of the settlement panels were even greater and provide a clue to the factors responsible. Upper surfaces in 1979-1981 were almost entirely dominated by crustose coralline algae that favor coral recruitment (Morse et al. 1988; Hughes et al. 2007; Arnold and Steneck 2011) and macroalgae were absent, whereas from 1998-2004 the upper surfaces were covered by macroalgae. Total numbers of recruits after each following year were > 500 in the 1979-1981 experiments compared to about 25 in the 1998-2000 experiments. These included a total of 981 recruits of *Agaricia* and *Porites* in the earlier experiment versus none of these taxa in 1998-2000. Numbers of "juvenile" corals (< 4 cm) on the reefs at Curaçao decreased by only 55% but these small corals can be as much as 13 years old (Vermeij et al. 2011). There was also a dramatic shift in juvenile coral composition: mean density of juveniles of brooding species decreased on average by about 10 recruits/m^2 whereas that of spawning species increased by 1-2 recruits/m^2.

The data from Curaçao strongly suggest that increased macroalgal abundance is a major factor in the reduced recruitment of corals, an observation consistent with earlier observations of Rogers et al. (1984) who concluded: "High rates of coral recruitment tended to be associated with low [non-calcareous] algal biomass and relatively high grazing pressure by urchins and fishes." Recent experiments strongly support this hypothesis. Recruitment of *Porites astreoides* larvae in Florida was inhibited by a variety of the most abundant macroalgae and cyanobacteria on Caribbean reefs (Kuffner et al. 2006). All of the macroalgal and cyanobacterial species tested caused recruitment inhibition or avoidance behavior by larvae and several species also significantly increased mortality of recent recruits. Behavioral avoidance reactions by the coral larvae suggest some form of chemical inhibition. There is also experimental evidence for more direct physical inhibition of recruitment by macroalgae at Roatán, Honduras (Box and Mumby 2007). Shading by *Lobophora* and *Dictyota* caused considerable losses in juvenile coral tissues and increased mortality of recruits and presence of *Dictyota* around the periphery of coral recruits decreased their growth rates by as much as 99%, and decreased cohort survival. Additional settlement experiments in Belize confirm that crustose coralline algae are strongly favorable to coral recruitment whereas macroalgae and turf algae inhibit recruitment (Arnold and Steneck 2011).

The role of macroalgae in inhibiting coral recruitment is even more strongly supported by striking increases in coral recruitment following reductions in macroalgae by recovering populations of *Diadema* and parrotfish. *Diadema* have begun to recover in increasingly large areas across the Caribbean (Edmunds and Carpenter 2001;

Carpenter and Edmunds 2006) reaching densities of 1.7/m^2 to 8.9/m^2 on a spatial scale of several km at sites in Belize, Jamaica, St. Croix in the northern Caribbean and Bonaire, Grenada, and Barbados in the south. Juvenile coral densities ranged from 4.5/m^2 to 32.3/m^2 in areas where *Diadema* have recovered versus 2.5/m^2 to 12.9/m^2 where they have not. A more recent study at Discovery Bay, Jamaica corroborated these results (Idjadi et al. 2010). Macroalgae were reduced from 68% to 6% cover. A combination of crustose corallines, turf algae, and bare space constituted 74% of the reef surface following grazing instead of 16%, and corals more than doubled from 4 to 11% cover in urchin zones compared to areas where urchins were absent.

A similar result emerges from the partial recovery of parrotfish in marine protected areas in the Bahamas (Mumby et al. 2006, 2007; Mumby and Harborne 2010). Increased parrotfish abundance and size in the Exuma Cays Land and Sea Park resulted in a 2- to 3-fold increase in parrotfish grazing intensity compared with unprotected sites. This increase in grazing further resulted in a decrease in macroalgal cover from 20-25% to about 1-5% and a 2- to 3-fold increase in coral recruitment. Moreover, size-adjusted rates of change in cover of five dominant coral species increased in areas within the marine park and decreased outside. The tipping point between positive and negative effects on coral growth occurred at about 10% macroalgal cover. Finally, the benefits of marine protected areas for increasing herbivory on reefs greatly exceed the potentially harmful effects of increased predator abundance on parrotfish (Mumby et al. 2006). This is because large bodied parrotfish can achieve an escape in size from predators.

Thus all of the evidence to date strongly supports the hypothesis that high macroalgal cover strongly reduces the recruitment of juvenile corals into the coral reef community. The negative effects of macroalgae far exceed the effects of decreased parental populations.

Increases in coral disease

There is also increasingly strong experimental evidence that high macroalgal abundance due to overfishing may induce outbreaks of coral disease as will be discussed in Section 3f on coral disease.

3c. COASTAL POLLUTION

It has long been understood that areas of greater sedimentation, seawater turbidity and light attenuation are less favorable to corals than clearer waters (Odum and Odum 1955; Sheppard et al. 2009). Low light affects photosynthesis by microbial symbionts, and sediments and oil interfere with ciliary feeding and may require increased production of mucus for sediment removal (Dodge et al. 1974; Bak and Elgershuizen 1976; Loya 1976; Dodge and Vaisnys 1977; Bak 1978; Rogers 1983b, 1990; Jackson et al. 1989; Guzmán et al. 1991; Burns et al. 1993, 1994; Guzmán and Holst 1993; Wolanski et al. 2003; D'Croz et al. 2005; Cramer et al. 2012). All of these different forms of stress may decrease coral growth rates and survival

Several factors contribute to increased turbidity including unregulated coastal development, dredging, other forms of coastal pollution such as oil spills, re-suspension of bottom sediments by storms, proximity to areas of naturally heavy rainfall and erosion, and excess nutrients from sewage, agriculture, and clearing of land. Nutrient pollution may be especially problematic because of excess production by phytoplankton and benthic algae that further reduce light levels (D'Croz et al. 2005) and may promote macroalgal growth and disease. The resulting positive feedback loop has negative impacts on coral survival including increased growth of macroalgae that may overgrow, abrade, or poison corals as well as inhibit their recruitment (Section 3b) and promote coral disease (Kline et al. 2006; Section 3f).

Most of the evidence regarding nutrient pollution versus grazers for the increased abundance of macroalgae implicates top down control by fishes, sea urchins, and smaller invertebrates (Hughes and Connell 1999; Aronson and Precht 2000; Burkepile and Hay 2006, 2008, 2009). However, nutrient and chlorophyll data are unavailable for most Caribbean reef locations because optical data from satellites cannot yet reliably determine chlorophyl levels in reef waters and there is a dearth of systematically collected data from water samples at different reef locations. Thus it not yet possible to systematically explore whether there is a strong case for the role of bottom-up processes except in the most heavily polluted locations such as parts of the Florida Reef Tract

(Lapointe 1997; Leichter et al. 2003). There is also no consistent monitoring of toxic substances released into Caribbean waters although some of the toxic effects of oil spills suggest that, just as for the Exxon Valdez spill (Peterson et al. 2003), chemical toxins of all sorts may be a greater problem for Caribbean reefs than is generally understood (Jackson et al. 1989; Guzmán et al. 1991; Burns et al. 1993, 1994; Guzmán and Holst 1993; Fernandez et al. 2007; García et al. 2008; Ramos et al. 2009).

Fortunately, simple measurements of water clarity/transparency are an excellent measure of several aspects of water quality including the effects of sediments, nutrients, and organic matter (Fabricius et al. 2012). There are also limited comparative data to examine trends in water transparency based upon secchi disk measurements that record the distance through the water column in meters at which the secchi disk is no longer visible from the surface or along a horizontal plane at depth (CARICOMP). Measurements were made at only seven of the CARICOMP sites and were made consistently for more than ten years at only three: inside the lagoon and on the fore reef at Carrie Bow Cay, Belize and at a single forereef site at La Parguera Puerto Rico (Table 9).

TABLE 9. Average secchi disk depths, degree heating weeks in 1998, 2005, and 2010, changes in coral cover in the two years following the extreme heating events of 1998, 2005, and 2010, and parrotfish abundance and coral cover since 1998 for 88 Caribbean reef locations (numbers same as in Table 2). See the text for the different drivers sections for further details.

Label	Country	Location	Median DHW			Proportional change in coral cover (%)			Secchi disk depth (m)	Parrotfish biomass (g/m²)	Coral cover since 2005 (%)
			1998	2005	2010	1998	2005	2010			
1	Antigua & Barbuda	Antigua & Barbuda	0.64	11.35	11.66					19.4	3.8
2	Aruba	Aruba	0	2.35	8.33						
3	Bahamas	Cay Sal Bank	1.31	2.48	3.33					14.8	7.1
4		Exuma Land Sea Park	8.09	8.98	6.86		0.07			9.8	7.8
5		Other	7.01	3.56	2.25	1.25	-0.01	0.02		27.7	11.7
6	Barbados	Barbados Leeward	0.52	3.3	11.85	0.08	-0.26				
7		South	0.52	3.3	11.85						
9	Belize	Atoll Leeward	1.09	0.55	3.33	0.03	-0.56			6	20.7
10		Atoll Windward	1.09	0.55	3.33	-0.50	-0.30	0.25		6.3	20.9
11		Belize Central Barrier	1.09	0.55	3.33	0.02	-0.27	-0.04	16	7.2	15.9
12		Gulf Honduras	2.93	0	4.02					4.5	7.6
13		Inner Barrier	2.93	0	4.02	-0.91				10.7	16.2
14		Northern Barrier	0.56	2.25	0.53	-0.64	-0.30	-0.18		8.9	16.9
15		Southern Barrier	2.93	0	4.02	-0.67		-0.06		6.4	13.5
16	Bermuda	Bermuda	5.1	1.68	4.55	-0.15	0.24	-0.28	36.8	21.9	38.6
17	British Virgin Islands	British Virgin Islands	1.61	8.92	3.18	-0.24	-0.36	0.08		13.8	14.3
18	Cayman Islands	Grand Cayman	2.27	0.51	1.11	-0.14				12.7	30.7
19		Little and Brac	2.16	2.49	2.19		-0.11	0.49		15.7	24.6
20	Colombia	Providencia	2.54	1.63	4.3						
21		San Andrés	1.9	2.11	5.42						
22		Santa Marta Region	1.51	2.52	5.05	-0.05					
23	Costa Rica	Costa Rica Cahuita	0	1.38	2.66		-0.15	0.00		39.8	18
24	Cuba	Jardines de la Reina	3.34	5.29	5.57					20.4	30.1
25		North	3.19	2.28	1.71					6.9	
26		Southwest	1.77	2.01	1.2					8.4	25.2
27	Curaçao	Curaçao Northwest	0	2.35	8.33	0.00	-0.02	-0.55		31.6	13.3
28		Curaçao Southwest	0	0.51	5.75	0.25	0.47	-0.02		15.2	31.5

Label	Country	Location	Median DHW			Proportional change in coral cover (%)			Seechi disk depth (m)	Parrot-fish biomass (g/m²)	Coral cover since 2005 (%)
			1998	2005	2010	1998	2005	2010			
30	Dominica	Dominica	2.81	8.68	10.54						
31	Dominican Republic	North	1.17	0.63	0		-0.09			3.1	21.3
32		Punta Cana	1.16	2.91	3.93					3.9	
33		South	1.83	2.26	9.96	-0.56				9.2	
34	French Antilles	Guadeloupe	5.19	9.91	14.1		-0.48	0.23			
35		Martinique	2.73	9.49	11.58		-0.38				
36		St. Barthelemy	0.5	6.04	2.57		-0.43	-0.08			
37	Grenada	Grenada other	2.17	9.78	15.08						
38		Leeward	2.17	9.78	15.08						
39	Guatemala	Guatemala	2.93	0	4.02					3	9.9
40	Honduras	Bay Islands	1.77	0	2.9					11.8	21.6
41		Near shore	1.77	0	2.9					22.5	12
42	Jamaica	Jamaica Montego Bay	2.88	2.74	2.66	-0.18				4.6	19.4
43		Jamaica North central	3.78	2.5	3.9	0.25	0.20			6.9	19.6
44		Jamaica Northeast	0.52	1.79	3.79					5.4	
45	Jamaica	Pedro Bank	1.75	0	4					15.4	14.7
46		Port Royal Cays	2.48	1.62	5.89						
47		Jamaica West	5.23	2.83	4.38	0.27				8.1	7.8
48	Mexico	Alacran	2.17	0	0						
49		Chinchorro Bank	1.16	1.67	0					1.4	7.9
50		Cozumel Leeward	1.11	0	0		-0.47	0.09		3.5	12.1
51		Cozumel Windward	1.11	0	0					0.7	9.2
52		North East Yucatan	1.11	0	0	0.22	-0.45			6.2	7.9
53		South East Yucatan	0.56	1.67	0					5.1	15.9
54		Veracruz	0.54	0	0						
56	Netherlands	Bonaire Leeward	0.53	5.76	13.7	0.11	0.13	0.69		32.3	37.1
57		Bonaire Windward	0.53	5.76	13.7					19.1	9.7
58		Saba	3.43	12.6	7.72	0.34				13.5	
59		Saba Bank	3.43	12.6	7.72					14.5	
60		St. Eustatius	4.05	11.61	10.84					23	
61	Nicaragua	Corn Islands	1.21	1.75	4.72					5.1	
62	Panama	Bahia Las Minas	2.18	6.37	2.63	0.01	0.15	0.01			
63		Bocas del Toro	2.89	1.2	6.59		0.00	-0.44		12.3	13.6
64		Costa Arriba	0.52	2.87	4.18	0.17	-0.09	0.02			
65		Panama San Blas	0	0	3.78					13.3	
66	Puerto Rico	Guanica	4.63	8.3	3.5						
67		Jobos Bay	0.74	7.67	9.83					2.1	8.7
68		La Paguera	1.15	8.09	9.18	-0.02	-0.53	-0.03	10.2	5.6	19.2
70		Turrumote	1.15	8.09	9.18					9.5	23.8
71		Vieques & Culebra	4.63	8.3	3.5		-0.84			19	8.1
72	St. Kitts & Nevis	St. Kitts & Nevis	1.03	9.96	3.81			0.41		13	11.1

Label	Country	Location	Median DHW			Proportional change in coral cover (%)			Seechi disk depth (m)	Parrot-fish biomass (g/m²)	Coral cover since 2005 (%)
			1998	2005	2010	1998	2005	2010			
73	St. Lucia	St. Lucia Leeward	2.45	6.87	11.2						
74	St. Martin	St. Martin	3.26	12.84	6.77					12.1	
75	St. Vincent	Grenadines	1.13	4.34	13.25					16.7	19.5
76	& the Grenadines	St. Vincent	1.84	7.39	12.3					6.8	24.9
77	Trinidad & Tobago	Trinidad & Tobago	2.26	4.12	10.05	0.65	-0.31	-0.02			
78	Turks & Caicos Islands	Turks & Caicos Islands	0	3.23	1.06					7.4	
79	U.S.A	Dry Tortugas	4.25	0.54	2.21		-0.19	0.07		7.5	8
80		Flower Garden Banks	2.48	3.88	6.37					35.8	53.1
81		Lower Florida Keys	8.47	4.7	6.54	-0.51	-0.42	0.36		24.2	10.3
82		Middle Florida Keys	0.55	0.5	0	-0.40	-0.41	0.07		8.4	8
83		Southeast Florida	1.93	2.43	2.8		-0.50	-0.38		3.6	2.8
84		Upper Florida Keys	4.15	0.53	1.58	-0.64	-0.24	0.02		20.3	6.1
85	U.S. Virgin Islands	USVI St. Croix	2.47	7.03	5.95	-0.35	-0.53	-0.32		13.1	4.7
86		USVI St. Thomas	4.63	8.3	3.5		-0.43			11.4	13.6
88		USVI St. John	4.63	8.3	3.5	-0.17	-0.47	0.33		8.3	10.1
89	Venezuela	Los Roques	1.74	2.91	13.57		0.01	-0.47		60.7	31
90		Morrocoy	0	1.2	12.48	-0.21	0.04	-0.14	12.9		

Trends in water transparency were assessed by testing the linear relationship of secchi distance to year while assuming an AR-1 autocorrelation (R package *nlme*; Pinheiro and Bates 2013). Water transparency significantly declined at all three consistently monitored sites, while coral cover has declined by approximately two thirds (Fig. 25; Koltes and Opishinski 2009; K. Koltes, personal communication; E. Weil, personal communication). The decline in transparency at Carrie Bow Cay is related to the conversion of lands bordering the Gulf of Honduras to agriculture and urban development. Massive amounts of sediments, primarily from Guatemala and Honduras, were introduced to the Gulf following the 2 m rainfall during Hurricane Mitch (Smith et al. 2002). These sediments became entrained in the gyre of the Gulf and continue to be re-suspended. More recent rapid conversion of the Belize coastline to intensive agriculture and tourism is also a major factor. Coastal development is also responsible for the downward trend at La Parguera (Hertler et al. 2009).

Coral cover declined by approximately two thirds at both Carrie Bow Cay and La Parguera, but the declines were episodic and uncorrelated with the gradual decline in water quality. Most of the decline at Carrie Bow Cay occurred before the transparency data began and coral cover increased by approximately 10% between 1994 and 2003 before declining precipitously by > 25% cover between 2003 and 2007 (Appendix II).

In contrast, water transparency did not change appreciably over 7 years at Morrocoy, Venezuela, but there were very large increases in heavy metals and hydrocarbons in relation to Venezuela's massive oil production (Bastidas et al. 1999; García et al. 2008; Ramos et al. 2009) that may have been a major factor in the dramatic losses in coral cover along the Venezuelan coast. Water transparency increased by about 50% over eight years at Bermuda.

3d. OCEAN WARMING

Reef corals host endosymbiotic photosynthetic dinoflagellates (*Symbiodinium*) that provide sugar to their coral host and are essential for coral growth and survival. Coral bleaching results from the ejection of the symbiotic dinoflagellates from the host coral due to stress. The most common form of bleaching occurs in response to extended increases in sea-surface temperature (SST) that are routinely measured in terms of Degree Heating Weeks (DHWs), defined as numbers of weeks during which SSTs exceed 1°C above the local

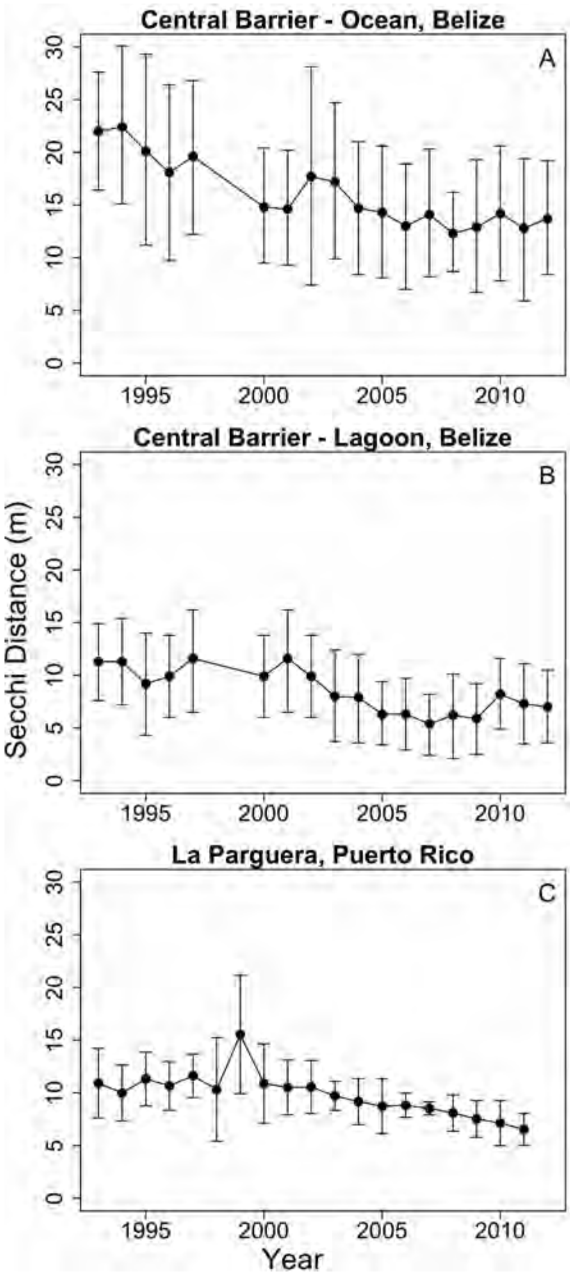

FIGURE 25. Decline in water transparency over time at the CARICOMP at fore-reef and lagoonal sites at Carrie Bow Cay and a fore-reef site at La Parguera, Puerto Rico. All of these trends are highly significant (GLMM, $p < 0.01$). See text for details.

climatological thermal maximum (Brown 1997; Hoegh-Guldberg 1999; Knowlton 2001; Hughes et al. 2003; Baker et al. 2008; Donner et al. 2007). However, different types of *Symbiodinium* are more or less resistant to elevated temperatures so that the bleaching response can be exceedingly varied and complex (Rowan et al. 1997; Knowlton and Rohwer 2003; Baker et al. 2008).

Mass mortality of corals commonly occurs when such high temperatures persist for more than one month. Bleaching events were rare before 1980 (Glynn 1993) but have since increased greatly in intensity and frequency due to rising temperatures that are in turn due to burning of fossil fuels and increases in other greenhouse gas emissions (Hoegh-Guldberg et al. 2007; Donner 2009, 2011). Increasingly severe coral bleaching events occurred in the Caribbean in 1995, 1998, 2005, and 2010 (Wilkinson and Souter 2008; Eakin et al. 2010).

Much progress has been made in prediction of coral bleaching events using long-term records of SST variability and the duration of heating events in association with the ReefBase compilation (http://www.reefbase.org) of coral mass bleaching events (Donner 2011; Chollett et al. 2012a,b). However, the ReefBase dataset has been criticized for three reasons: (1) bias towards reporting the occurrence of mass bleaching events, but less frequently their non-occurrence, (2) uneven spatial distribution of reports, and (3) absence of data from many of the large, well organized monitoring programs (Oliver et al. 2009; Donner 2011).

A major step forward has been provided by NOAA Coral Reef Watch (CRW) that has conducted near real-time global monitoring of thermal stress (http://coralreefwatch.noaa.gov) since 2000 based on satellite SST data at a resolution of 0.5-degree (~50km). These data have the important advantage of measuring the strength of the driver rather than the ecological response and complement observations *in situ*. They also provide an independent and consistent measure of thermal stress over the entire ocean rather than a hodgepodge of scattered measurements using different instruments and methodologies. As such, they provide an invaluable tool for managers and scientists to alert them of likely severe bleaching events before they occur and to facilitate preparations for essential in situ observations of ecosystem response. The DHWs product has been associated with significant coral bleaching (≥4 DHWs), and with widespread bleaching and significant coral mortality (≥8 DHWs; Liu et al. 2003; Eakin et al. 2009). However, extreme bleaching events do not always result in massive coral mortality, as evidenced by very large variations in mortality among locations that were comparably heated following the extreme heating event in 2005 (Eakin et al. 2010). CRW has also produced historical thermal stress products based on retrospective SST data as far back as 1985.

For this study, CRW extended these thermal stress products historically based on retrospective SST satellite data prior to 2000. SST data at 0.5-degree resolution were developed from the Pathfinder version 5.2 dataset (Casey et al. 2010), mimicking the methodology used for the operational CRW near real-time SST product (as described in Eakin et al. 2009). These data were combined with the near real-time SST data to extend the time-series back to 1985. Annual maximum DHW values for 1985-2011 (0.5-degree) were calculated for reef-containing pixels corresponding to each location in Table 9. Data were then combined for each location by taking the median number of DHW per location for 1998, 2005, and 2010. Pathfinder SST data were provided by GHRSST and the US National Oceanographic Data Center, supported in part by a grant from the NOAA Climate Data Record (CDR) Program for satellites. Our use of these data solely reflects the opinions of the authors of this report and do not constitute a statement of policy, decision, or position on behalf of NOAA or the US Government.

We analyzed changes in coral cover for the two years following each of the prolonged and extreme heating events in 1998, 2005, and 2010 relative to the two years preceding the event in relation to the numbers of DHWs experienced (Table 9). There was also a significant heating event across the southern Caribbean in 1995 with numbers of DHWs ranging from > 10 to 19.5 in a broad swath from mainland Colombia and San Andres in the west to Venezuela in the east, and with slightly lower numbers of DHWs in Panama and Barbados (CARICOMP 1997). Unfortunately, coral cover data are too sparse for detailed before and after comparisons around this event. Nevertheless, the timelines for these reefs that go back before 1995 show little decline or even increases in coral cover after 1995, suggesting low coral mortality. Support for this inference comes from the timelines for Morrocoy and Los Roques in Venezuela that begin a few years after 1995 with exceptionally high coral cover of 55% and 44% coral cover respectively (Table 4, Appendix 2). There are also scattered reports of bleaching events at Florida before 1995 (Billy Causey, personal communication) but we lack the quantitative data for comparative analysis.

We explored the relationship between degree heating weeks and proportional changes in coral cover for the 1998, 2005, and 2010 heating events both separately and for all three events combined using non-parametric correlations. Linear statistics are not appropriate in this case due to spatial and temporal autocorrelation causing residual variation to be correlated. Correlations were calculated for proportional changes in coral cover (decreases or increases) as a function of numbers of DHWs using two data sets. In the first case we used all of the data regardless of the numbers of DHWs experienced at any location. The purpose of this broader analysis was to determine the extent to which extreme heating events may have been responsible for changes in the abundance of corals throughout the wider Caribbean in comparison with other drivers of change. In contrast, the second analysis only employed data above the postulated critical threshold of 8 DHWs to examine more closely the impact of extreme heating events on coral cover. We also assessed the extent to which proportional changes in coral cover were related to numbers of DHWs above and below 8 DHWs by constructing contingency tables.

There is a small, non-significant negative correlation between proportional changes in coral cover and numbers of DHWs for the entire data set (Fig. 26A). We also examined the same data using 2 x 2 contingency table analysis for changes in coral cover at locations that experienced < 8 or ≥8 DHWs with marginally significant results (X^2 = 3.11, df = 1, p = 0.07). Remarkably, however, the trend is *opposite to the expected pattern* because the two locations that experienced the highest numbers of DHWs experienced a substantial proportional *increase* in coral cover. Moreover, the proportion of locations that lost coral cover is not different for places that experienced more or less than 8 DHWs (74% and 73% respectively), and six of the eight locations that suffered losses in coral cover > 50% coral were exposed to < 8 DHWs. Finally, and even more remarkably, there is a significant *positive* correlation between proportional changes in coral cover and numbers of DHWs using only the data for locations that experienced > 8 DHWs (r_s = +0.66, p = 0.01).

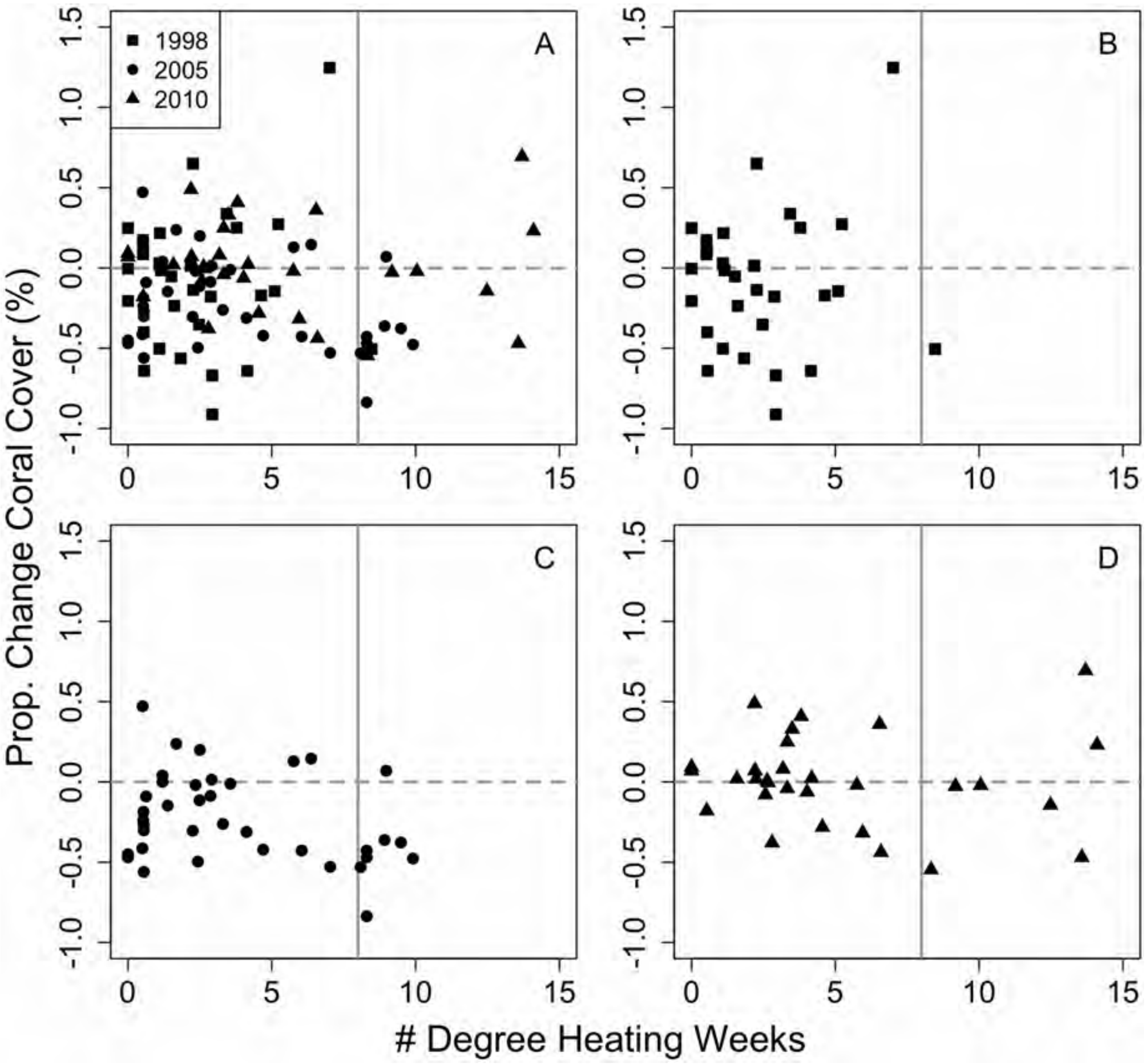

FIGURE 26. Proportional changes in coral cover in the two years following major heating events in relation to the number of degree heating weeks (DHWs) at all the locations for which paired data are available (Table 9) (A) all three events combined, r_s = -0.10, p = 0.34); (B) in 1998, r_s = -0.07, p = 0.71; (C) in 2005, r_s = -0.20, p = 0.25, and (D) in 2010: r_s = -0.20, p = 0.29. See text for details of the analysis.

Graphs of the loss of coral cover against the number of DHWs after 1998, 2005, and 2010 vary in their relationship between coral cover and thermal stress (Fig. 26B-D). For the earliest event in 1998, there are very few Caribbean reef locations that experienced ≥8 DHWs, and overall no relationship was found to proportional loss of coral cover (Fig. 26B). Similarly, no regional correlation between DHW and proportional loss in coral cover was found in 2005 (Fig. 26C) and 2010 (Fig. 26D). Contingency table analyses for 2005 and 2010 were also non-significant (2005: X^2 = 0.78, df = 1, p = 0.38; 2010: X^2 = 1.92, df = 1, p = 0.17). Contingency table analysis could not be done for 1998 because of the lack of data in the upper right quadrant.

Lack of an overall regional correlation between numbers of DHWs and changes in coral cover does not mean that bleaching is an unimportant cause of coral mortality because numerous studies have clearly demonstrated mass mortality following coral bleaching (Hoegh-Guldberg et al. 2007). Moreover, even greater rising temperatures in the future will almost certainly have increasingly severe effects. But the data do belie any strong, regionally consistent effects of coral bleaching upon coral cover *up to the present*. Instead, mortality due to bleaching has been highly localized. At Belize, for example, coral cover dropped precipitously from about 35-45% to zero after the massive bleaching event at two cays on the leeward side of the southern barrier reef (Aronson et al. 2002), but mortality was negligible at Glovers Atoll farther offshore (Mumby 1999). A similar drop occurred at Carrie Bow Cay from the combined effects of coral bleaching and Hurricane Mitch (K. Koltes, personal communication). Mortality was also very extensive after the 2005 massive bleaching event at St. John, with proportional losses in coral cover of up to 60% on some reefs (Miller et al. 2009), and at La Parguera, Puerto Rico (Weil et al. 2009). In both of these cases, however, mortality may have been largely due to a major outbreak of disease that closely followed the thermal stress (see Section 3f).

3e. INVASIVE SPECIES

The Caribbean is effectively a Mediterranean sea and has been the most isolated tropical ocean on the planet ever since the final closure of the Central American Seaway by the rise of the Isthmus of Panama between about 5 to 3 million years ago severed its connection with the eastern tropical Pacific (Coates and Stallard 2013; Jackson and O'Dea 2013). Isolation from the tropical Indian Ocean to the east occurred even earlier due to the movements of the continents of Africa and Asia, the subtropical Mediterranean, and the inhospitable eastern Atlantic. Thus, by analogy to the fates of the myriad island birds and reptiles decimated to the point of extinction by introduced species of snakes, rats, cats, and goats (Fritts and Rodda 1998; Pimentel et al. 2005), Caribbean marine species should be exceptionally prone to the impact of introduced competitors and predators. Moreover, by analogy to the fates of the original Americans after their first contact with European diseases (Crosby 1986; Mann 2005), Caribbean corals should be especially vulnerable to introduced diseases.

Most of the recent focus on introduced marine species has concentrated on highly visible macroorganisms, such as the explosive increase in the abundance of the Pacific lionfish *Pterois volitans* throughout the entire wider Caribbean over the past decade (De Leon et al. 2011; Hackerott et al. 2013) or the uncontrolled spread of the alga *Caulerpa taxifolia* in the northern Mediterranean (Meinesz et al. 1993, 2001). The potential effects of lionfish on Caribbean invertebrates and fishes may be severe, especially in exacerbating the consequences of overfishing by depleting juvenile parrotfishes and surgeonfishes (Albins and Hixon 2018, 2013). However, it is too soon to tell whether native predators might eventually have an impact of lionfish, especially in marine reserves where predators could regain their former abundance (Mumby et al. 2013).

Far too little attention has been paid, however, to the introduction of the myriad marine organisms we cannot see, including virtually all microorganisms and pathogens. The case of the unidentified pathogen that caused the mass mortality of *Diadema antillarum* in 1983-1984 is a case in point. *Diadema* mortality began next door to the Caribbean entrance to the Panama Canal, whence it spread like wildfire on ocean currents eastward to Trinidad and Tobago and northward throughout the western Caribbean, Greater Antilles, and Florida all the way to Bermuda, with mortality in the eastern Caribbean arriving from both the north and the south in 1984 (Lessios et al. 1984; Lessios 1988). Introduction via ballast water from the Pacific is seemingly the most reasonable explanation.

This begs the question of why so many marine diseases first appeared in the 1970s and early 1980s, a pattern for which there is no compelling environmental explanation. Temperatures were not excessively warm in the 1970s and heating in relation to El Nino in 1983 was small compared to the episodes in 1995, 1998, 2005, and 2010. There is also no evidence of a pervasive decline in Caribbean water quality before the 1980s or later.

In contrast, the volume of international shipping exploded in the late 1960s with the advent of bulk carriers and enormous cruise ships that discharged untold volumes of ballast water into coastal waters before stricter regulations may have begun to take effect (Carlton 1996). Greater speed of transport among distant ports may also be a contributing factor. Many introduced species have been transported by ballast water, and this is especially true for microbes that have been calculated to be transported in numbers on the order of 10^{20}/year into the lower Chesapeake Bay alone (Ruiz et al. 2000; Drake et al. 2007). None of this proves that *Diadema* disease or WBD were introduced into the Caribbean from another ocean. But given the numbers of microbes in ballast waters, it is remarkable that all marine diseases have not been introduced throughout the global ocean. Introductions of aquarium species and so-called "live rock" for aquaria are another potentially major avenue for incidental introductions of pathogens.

Once introduced, different environmental factors may retard or promote the growth of introduced species including species that cause disease. But it is important not to confuse the causes of an initial outbreak from factors that may subsequently promote or inhibit its spread and increase.

3f. INCREASING INCIDENCE OF CORAL DISEASE

Corals are complex ecological communities (holobionts) comprising the coral host and an extraordinary diversity of associated eukaryotic and prokaryotic microorganisms (Rohwer et al. 2001, 2002; Knowlton and Rohwer 2003; Rosenberg et al. 2007). These associates include a great diversity of intracellular, endosymbiotic dinoflagellates (*Symbiodinium*) and a bewildering variety of bacteria, archaea, and viruses that confer essential nutritional and immunological benefits to the host coral by photosynthesis, provision of nutrients, nitrogen fixation, and resistance to infection. The ecological balance among all of these mutualistic ecological components of the holobiont community is essential for coral health. Breakdown in that balance due to a change in the environmental or genetic landscape of the holobiont or the invasion or increase in a pathogen compromises the health of the holobiont in the form of myriad forms of coral bleaching or disease. Understanding of the underlying mechanisms of these ecological interactions that compromise coral health is in its infancy, so that scientists are required to describe phenomena in terms of their gross phenotypic expression (e.g., bleaching, White-Band Disease, Black-Band Disease, Yellow-Band Disease, etc.) rather than the precise underlying ecological components of cause and effect (Weil and Rogers 2011).

In recognition of this complexity, disease is commonly defined as “any impairment to health resulting in physiological dysfunction” due to a pathogen (virus or microorganism), environmental perturbation, toxic substance, or genetic changes in the affected organism (Weil and Rogers 2011). Defined so broadly, coral bleaching, mercury poisoning, or smothering by sediments can be treated as a disease – a definition so broad as to be of little use. For this report, therefore, we define coral diseases more narrowly as impairments to coral health caused by a demonstrable or presumptive infectious pathogen that results in varying pathological responses or death (see Martin et al. 1987; Wobeser 1994 for further discussion of these criteria).

Coral diseases so defined occur in a bewildering variety of forms that may affect a few or many coral taxa (Weil and Rogers 2011). In most cases the diseases are identified by the pathological expression exhibited by the affected coral. The actual pathogens have been identified in only a few cases, and similar manifestations of disease in the changing appearance of the affected coral may be caused by different pathogens in different circumstances. Failure to identify pathogens is the major impediment to any real advance in understanding the causes and consequences of coral disease. Little is known about transmission, but there is evidence that various predators of corals including fishes, polychaete worms, and snails may transmit diseases from one coral prey to the next, as well as transport by currents (Williams and Miller 2005; Rosenberg et al. 2007; Weil and Rogers 2011) or in the ballast water of ships (Drake et al. 2007).

The first report of coral disease in the Caribbean was for BBD in Belize, Florida, and Bermuda in the early 1970s and throughout the western Atlantic soon after (Antonius 1973, 1977; Weil and Rogers 2011). BBD appears as a dark microbial mat and infects 19 species of Caribbean corals. BBD was followed closely by a virulent outbreak of WBD that caused mass mortality of *Acropora palmata* in the US Virgin Islands in the late 1970s, and spread throughout the western Atlantic to cause mass mortality of both *A. palmata* and *A. cervicornis* in the early 1980s to the present (Gladfelter 1982; Goreau et al. 1998; Aronson and Precht 2001; Weil and Rogers 2011). In total, about 13 different diseases of corals have been identified whose distribution and prevalence varies greatly among different locations within the wider Caribbean (Weil and Cróquer 2009; Cróquer and Weil 2009; Weil and Rogers 2011).

Despite numerous breakthroughs in documenting the agents of coral disease we do not understand why outbreaks of disease occur. The two most likely explanations are (1) introduction of a pathogen to an area where it was previously absent, as in the case of bubonic plague, and (2) increase in the abundance of a previously rare pathogen due to changes in the physical or biotic environment as with outbreaks of cholera in polluted waters. A third possibility is the evolution of a new pathogen that sweeps through host populations with devastating effects. Such an explanation is extremely unlikely for Caribbean corals because it would require the synchronous evolution of more than a dozen major coral pathogens within one or two decades.

Data are so far inadequate to identify whether invasions or environmental change were the major factor in the emergence of particular Caribbean coral diseases, but there are valuable hints related to the timing of appearance and severity of diseases. This is especially true for the first outbreaks in the 1970s and early 1980s, most notably WBD, BBD, and the pathogen that caused the massive die-off of the sea urchin *Diadema antillarum* in 1983/84 (Lessios et al. 1984; Lessios 1988; Weil and Rogers 2011). In each case, mass mortality approaching 95-100% occurred 15-25 years before the first episodes of extreme heating events due to global warming or any other documented regional environmental change. It is therefore of considerable interest that the effects of WBD and *Diadema* disease have been so much more extreme than in other tropical seas. Nothing like the mass mortality of *Diadema* has affected any echinoderm throughout the entire Indian Ocean or tropical Pacific, nor has any genus of Indo-Pacific acroporid suffered such broad and lasting extirpation as Caribbean *Acropora palmata* and *A. cervicornis*.

More progress has been made in understanding the causes of more recent and seemingly chronic disease outbreaks in relation to rising temperatures and the increased abundance of macroalgae after the demise of *Diadema* (Table 3; Figs 12-14). Evidence for a temperature effect comes from increases in the incidence in disease after extreme heating events and coral bleaching (Weil and Rogers 2011). However, such outbreaks of disease may result either from a general weakening of coral due to the physiological distress caused by bleaching or thermal stress per se. Experiments are needed to help resolve these alternatives.

In contrast, numerous recent experiments have demonstrated that physical contact or even close proximity to various macroalgae may also trigger the outbreak of a wide variety of pathological responses including virulent diseases in corals (Nugues et al. 2004; Kline et al. 2006; Smith et al. 2006; Rosenberg et al. 2007; Knowlton and Jackson 2008; Barott and Rohwer 2012; Morrow et al. 2012; Rasher et al. 2012; but see Vu et al. 2009 for somewhat contrary results). Toxic allelochemicals from macroalgae also disrupt the complex microbial communities present on the surface of coral colonies, and may cause bleaching and death of coral tissues when in direct contact (Rasher and Hay 2010; Rasher et al. 2011).

In summary, increases in macroalgae principally due to overfishing can disrupt the ecological balance of reef coral assemblages in many ways. Macroalgae inhibit coral growth and may cause direct mortality by shading or abrasion. They also inhibit coral recruitment and disrupt symbiotic assemblages resulting in outbreaks of disease and coral death. These are all testable hypotheses in marine protected areas and wherever else that populations of herbivores may recover and graze down macroalgae to previously low levels of abundance. If the macroalgal disease hypothesis is correct, incidence of coral disease should decline in concert with the decline in macroalgae.

Bleaching and disease are increasingly closely associated in their occurrence but the reasons are obscure because coral cover at some reefs *increased* or was stable after experiencing very high numbers of DHWs (Table 9; Fig. 26). For example, the leeward coast of Bonaire experienced < 1, 5.8, and 13.7 DHWs in 1998, 2005, and 2010 with a proportional *increase* in coral cover of 11, 13, and 0.1% respectively in the two years thereafter. The southwest coast of Curaçao also experienced > 10 DHWs during all three events and percent coral cover *increased* proportionately by 25% after 1998, by 47% after 2005, and declined by just 2% after 2010. Northwest Curaçao and nearby Los Roques experienced 2.3 and 2.9 DHWs in 2005 with 2% and 4% change in coral cover. However, these same locations experienced a precipitous proportional decline in coral cover of 55% and 14% respectively after experiencing 8.3 and 12.5 DHWs in 2010. The decline in NW Curaçao was due to a combination of factors including exceptional storms, increased coastal development, and coral bleaching (Mark Vermeij, personal communication), but the decline at Los Roques was due to massive coral bleaching followed by disease (Bastidas et al. 2012).

In contrast, coral cover on reefs in the USVI and at La Parguera and Vieques in Puerto Rico declined proportionately by 47-53% after enduring 7 to 8.3 DHWs in 2005. These much greater losses in coral cover after experiencing less heat stress than in NW Curaçao and Los Roques strongly imply that the consequences of extreme heating stress are somehow mediated by other environmental factors than heat stress alone. Coral mortality in the USVI and Puerto Rico after 2005 was due primarily to outbreaks of coral disease (Rogers and Miller 2006; Muller et al. 2008; Rogers et al. 2009; Miller et al. 2009; Weil et al. 2009). We postulate that these greater losses in the USVI and Puerto Rico may reflect regional differences in macroalgal abundance, which is generally considerably lower in the southern Caribbean. Support for this hypothesis comes from the experiments discussed above and the anomalous increase in total algal cover at Los Roques of 34 to 54% before and after the 2010 extreme heating event when coral cover declined precipitously, versus the minor proportional losses in coral cover in SW Curaçao where macroalgal cover is much lower.

3g. THE ROLE OF HURRICANES

Strong hurricanes have been a natural occurrence on coral reefs for millions of years and are potentially highly destructive to corals (Woodley et al. 1981; Rogers et al. 1982, 1991). Reefs have routinely recovered from hurricane damage in the past or reefs would not exist. The occurrence of hurricanes varies greatly throughout the wider Caribbean region (Chollett et al. 2012a). Hurricanes are frequent and intense in a broad swath from the northern Lesser Antilles across Puerto Rico, eastern Cuba, Jamaica, and the Cayman Islands to eastern Yucatan as well as southern Florida. In contrast, hurricanes are rare all across the southern third of the Caribbean from Barbados to Nicaragua and points south. Despite these differences, however, average coral cover from 1970 through 1983 was remarkably similar among the 16 locations with old *Diadema* data in Tables 5 and 8. Corals differ greatly in their rates of recruitment, growth, and reproduction. These differences in life history characteristics are believed to have been responsible for a natural pattern of succession of reef communities extending for up to several decades after a storm had passed (Woodley et al. 1981; Rogers 1983a).

Nevertheless, the frequency and intensity of hurricane occurrence have been proposed as important drivers of coral decline on Caribbean reefs, especially since the 1980s when corals have failed to recover in many cases due to some combination of human stressors (Gardner et al. 2005). We therefore examined this hypothesis in two ways using the 16 reefs in Table 8. The first analysis addresses the null hypothesis that coral cover at the 16 locations prior to the mass mortality of *Diadema antillarum* in 1983 was independent of the long-term annual probability of hurricane occurrence at each location over the past 160 years. The second analysis addresses the null hypothesis that the changes in coral cover after 1983 were independent of the numbers of hurricanes that actually occurred at each location after 1983. Hurricanes vary in intensity and the details of their tracks through an area that affect their potential impact on reefs (Fabricius et al. 2008), but such detailed data are available for only a small proportion of hurricanes. Nevertheless, the long-term probability of hurricane occurrence, and their actual frequency since 1983, should provide a good first order estimate of the impact of hurricanes on coral cover both in the past and on reefs today.

Hurricane incidence was measured using the Atlantic Hurricane data set (1851-2012), which tracks the location and intensity of the eye of tropical cyclones every six hours (Jarvinen et al. 1984). Hurricane force winds may extend several kilometers from the hurricane track. We captured the spatial influence of hurricanes by using the buffering system described by Keim et al. (2007) and Edwards et al. (2011). Buffers capture the area of influence of each hurricane by taking into account the intensity of the storm, its asymmetry, and the reduction in wind speed away from the track (Keim et al. 2007; Edwards et al. 2011). The hurricane dataset was used previously by Chollett (2012a) but is here updated to include Bermuda and four more years of data from 2009-2012 (Table 8). Hurricane incidence was extracted for each pixel within the polygon drawn for each reef location in Table 5 (Fig. 27). The number of pixels extracted and the average and standard deviation of hurricane incidence were reported for each of four time periods: (1851-2012, 1970-1983, 1984-1998, and 1999-2012).

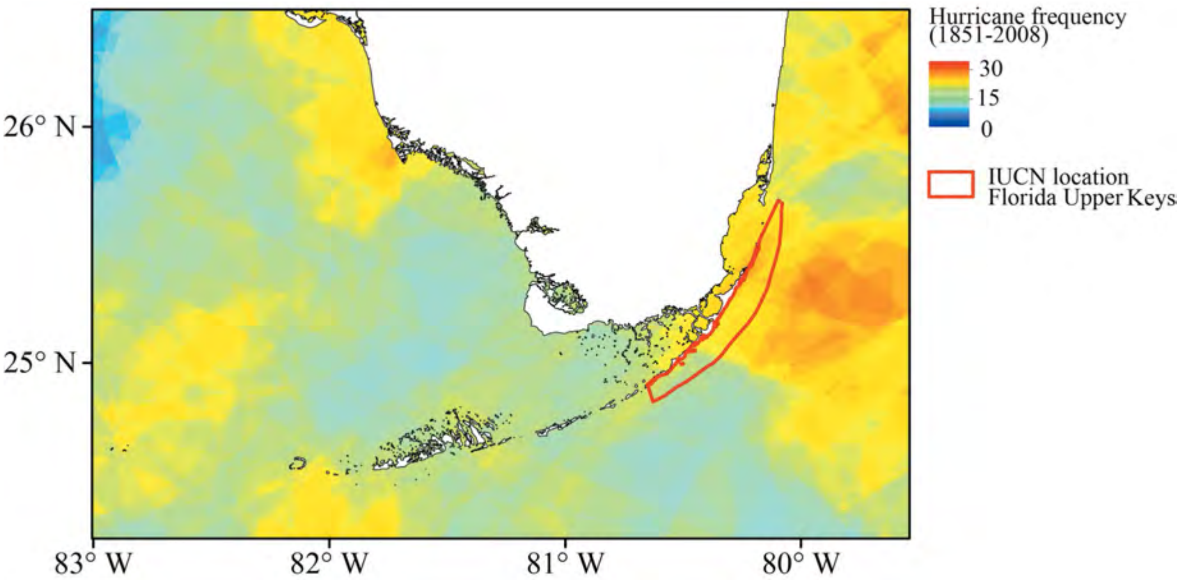

FIGURE 27. Example of the methodology for extraction of the incidence of hurricanes for the Upper Florida Keys.

Coral cover on reefs before 1984 is negatively correlated with the long-term probability of hurricane occurrence but the relationship is not significant (Fig. 28A). This suggests that hurricane frequency was not a major determinant of coral cover on reefs prior to 1984.

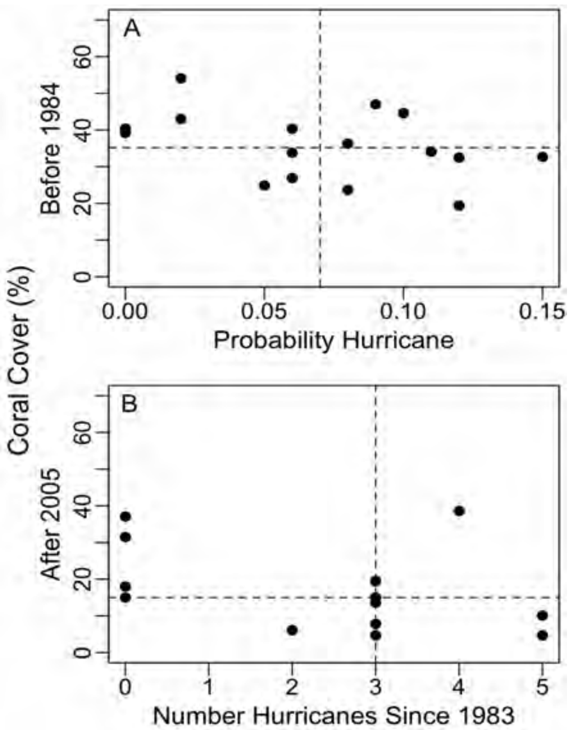

FIGURE 28. Coral cover versus hurricane occurrence for the 16 reef locations in Table 8. (A) There is no relation between the long-term probability of hurricane occurrence at the 16 reefs in Table 5 up to 1983 (r_s = -0.4, p = 0.15). (B) Since 1984, the number of hurricanes is also uncorrelated with coral cover (r_s = -0.25, p = 0.38) except when the protected reefs at Bermuda are removed from the analysis (r_s = -0.57, p = 0.04).

The number of hurricanes that have occurred at the 16 locations since 1984 is highly positively correlated with the long-term probability of hurricane occurrence at the same locations, demonstrating that the incidence of hurricanes over the past 30 years has not departed from the normal pattern (r_s = 0.67, p = 0.01). Average coral cover since 2005 is negatively but insignificantly correlated with the numbers of hurricanes that have occurred since 1984, due to the very high coral cover at Bermuda despite four hurricanes since 1984 (Table 8, Fig. 28B). Removal of Bermuda from the analyses had no effect on the results for the years prior to 1984, but the negative correlation between coral cover and number of hurricanes since 1984 was significant (Fig. 28B). It is important in this context to remember that acroporids have always been absent from Bermuda where reefs are overwhelmingly dominated by massive corals, which are more resistant to hurricanes than branching species (Woodley et al. 1981).

Fish traps were banned in Bermuda in 1990 and parrotfish are still abundant. In contrast, reefs on the Belize Central Barrier have been overfished since the 1990s (Mumby et al. 2012) in addition to having experienced three hurricanes. Coral cover declined proportionately by 49% (Table 5). Coral bleaching was extensive at Carrie Bow Cay ten days after the passage of Hurricane Mitch (K. Koltes, personal communication), but the reason(s) for bleaching are obscure because of a huge influx of sediments and freshwater due to > 1 m of rain within 24 hours (Koltes and Opishinski 2009; K. Koltes, personal communication). Massive influxes of freshwater have been previously shown to have caused nearly 100% coral bleaching in Jamaica following hurricane Flora in 1963 (Goreau 1964). Jamaica was already overfished in the 1960s (Jackson 1997) but *Diadema* were extremely abundant and macroalgae virtually absent. These are only isolated examples but suggest that protection of herbivores and wiser land use as in Bermuda may have conferred greater resilience of reefs to hurricanes.

3h. THE SPECIAL CASE OF THE FLORIDA REEF TRACT (FRT)

The ecological situation of the FRT is unique due to its particular environmental setting and the unprecedented scale of human impacts that include all of the drivers discussed in this report (Ault et al. 2005; Keller and Causey 2005; Causey 2008; Kruczynski and Fletcher 2012).

The FRT is a predominantly continental reef system in south Florida and the Florida Keys that is situated towards the northern geographic occurrence of Atlantic coral reefs. Fluctuations in environmental conditions and the long-term probability of hurricane occurrence are among the highest in region. The FRT is also positioned at the junction of Caribbean waters from the south, Gulf of Mexico waters from the West, and the subtropical western Atlantic. Moreover, the reefs of the Florida Keys sit just offshore of Florida Bay into which the Everglades drain. For all of these reasons, the assemblages of species and habitats of the FRT were considerably different from anywhere else in the wider Caribbean region long before human impacts intensified.

Unprecedented increases in land use, coastal development, and pollution of south Florida occurred over the past half century as human populations exploded. The hydrology of the Everglades and Florida Bay has drastically changed and nutrient and sediment influx greatly increased with direct impacts on coastal estuarine habitats and water quality in the Florida Keys. These local dynamics have combined with regional and global environmental change to impact reefs along most of the FRT.

Today the FRT is adjacent to the major metropolitan area of greater Miami with a rapidly growing human population exceeding 5 million while also serving as a popular tourist destination with unparalleled access to the reefs for recreation and exploitation. Intensity of human use and environmental impacts greatly exceeds that of any other region in the wider Caribbean, if not the world. Numbers of fishers, boaters, and divers increase every year. Nearly a million vessels are registered in Florida with a majority in the southern portion of the state. Damage due to boat groundings, propeller scour, anchoring, and shipwrecks is extensive and wastewater runoff enormous. Overfishing has virtually eliminated formerly abundant Goliath and Nassau Groupers and stocks of other target species such as snappers, lobsters, and conchs are overfished.

Establishment of the nearly 10,000 square kilometer Florida Key's National Marine Sanctuary (FKNMS) in 1997 led to the creation of a modest network of no-take marine reserves with a total protected area of only 6% of the total area of the Keys. This action, combined with increasingly restrictive fishing regulations, has led to small increases in a limited number of stocks, and a general slowing of the decline in fish stocks overall. The FKNMS has also been successful in encouraging collaborative management strategies allowing Florida to successfully document and implement corrective actions to improve wastewater and storm water treatment and disposal.

In conclusion, the FRT epitomizes a kind of worst-case scenario in which unprecedented population growth and inadequate governance and regulations have resulted in the critical endangerment of an entire coral reef ecosystem. Despite the positive and courageous actions of the Sanctuary, coral cover is well under 10% and declining. Much more stringent actions will be required for any hope of coral survival.

4. SYNTHESIS

We first review the major results of the analyses of pattern and then focus on the apparent importance of the different drivers of coral reef decline.

4a. PATTERNS OF CHANGE

The three overarching results of this report are that

1. Most of the degradation of Caribbean reefs occurred between the 1970s to early 1990s well before most ecological surveys began.
2. Phase shifts from greater coral to greater macroalgal abundance happened early and are geographically pervasive.
3. Geographic disparity in the fates of reefs at different locations was and is truly enormous.

Timing and rates of reef degradation

Average coral cover throughout the wider Caribbean, Gulf of Mexico, and Bermuda declined by 49% from an overall average of 33.0% before 1984 to 17.7% since 2005 (Tables 2 and 3, Fig. 7). Refinement of our 2005 estimate to take into account the great variation among locations and datasets lowers the 2005 estimate to 14.3% coral cover with an overall decline of 59%. These estimates of loss are considerably lower than Gardner and colleagues' estimate of an 80% decline from 50% to 10% (Gardner et al. 2003) but in good agreement with the estimate of Schutte et al. (2010) of a 60% decline from about 40% to 16% cover. The earlier estimates were based on considerably less data and were disproportionally dominated by surveys from the Florida Reef Tract, US Virgin Islands, and Jamaica that are among the most severely degraded reefs in the entire region.

Coral cover declined at 73% of locations with time series data (Fig. 8). The declines were greatest for locations that began to be studied earliest and over the longest period of time. Indeed, 88% of the total overall Caribbean decline in coral cover occurred between 1984 and 1998, and this increases to 100% for the 21 reef locations with long-term data extending back before 1984 (Table 3). Likewise, 99% of the overall Caribbean increase in macroalgal cover occurred before 1998, with a somewhat lower value of 81% for the 21 long-term reefs.

The same was true for the dramatic declines of iconic species. *Acropora palmata* and *A. cervicornis* began to decline in the 1960s and were virtually ecologically extinct at most Caribbean locations by the mid 1980s (Fig. 18). *Diadema antillarum* was the most important grazer on overfished Caribbean reefs and common elsewhere until 1983-1984 when more than 95% of all Caribbean *Diadema* died due to an unidentified pathogen (Lessios 1988; Fig. 19). Parrotfish had been extremely reduced at Jamaica, the USVI, and other overfished locations by the 1960s or before (Randall 1961, 1963; Munro 1983; Hay 1984; Lewis and Wainwright 1985), and are rare on most Caribbean reefs today (Fig. 20).

These sobering results of very early decline long before most coral reef ecologists today had ever seen or read about a coral reef are a classic example of the Shifting Baselines Syndrome (Pauly 1995; Jackson and Jacquet 2011; Jackson et al. 2012) and a harsh reminder that what is going on today is the end of a much longer story.

Phase shifts

The dramatic reversal between coral and macroalgal abundance (Fig. 13) occurred over about a decade and is strong evidence for a phase shift in coral reef community structure (Done 1992; Knowlton 1992, 2004; Hughes 1994; Hughes et al. 2010; Schutte et al. 2010). Forty-one percent of the total variation in the PCA ordination of coral and macroalgal community composition at the 21 long-term locations is explained by the shift from coral to macroalgal dominance (Fig. 16).

Some have questioned the generality of phase shifts on coral reefs claiming that the Caribbean example of corals to macroalgae is unrepresentative of the general pattern of overall change (Aronson and Precht 2006; Bruno et al. 2009). However, our results are based on vastly more data and greater geographic coverage than any previous analysis and overwhelmingly support the occurrence of a phase shift at most Caribbean locations from coral to macroalgal dominance. The question is not whether a phase shift occurred, but what might be done about it to return reefs to their thoroughly documented former dominance by abundant corals.

Geographic variation in reef decline

Clues to the possible recovery of Caribbean reefs lie in the enormous variability among Caribbean reef locations today (Table 3 and 5; Figs. 9-12, 14; Appendix 1).

Coral cover for 53 locations since 2005 varies from a low of < 3% off of Port Royal, Jamaica to a high of 53% at the east Flower Garden Banks in the northern Gulf of Mexico (Table 9). Seventeen locations have < 10% coral cover and another 21 between 10 to < 20%. Thus, three quarters of all the Caribbean locations for which we could find data have degraded by at least 50% below the average coral cover before 1984. But 15% of the locations have > 20% cover and another 13% have > 30% cover including Bermuda, Grand Cayman, Jardines de la Reina on the south coast of Cuba, southwest Curaçao, the leeward coast of Bonaire, Flower Garden Banks, and Los Roques Venezuela. This pattern is virtually identical to the distribution of cover in the third time interval of 1999-2011 (Fig. 7). The obvious question is why these reefs with > 30% cover are doing so well compared to all the rest?

4b. DRIVERS OF CORAL REEF DECLINE

Our analyses focused on potential drivers of decline for which there were adequate data for meaningful comparisons. The results are particularly strong for evaluating the effects of overpopulation, overfishing, and global warming, and less so for coastal pollution and invasive species.

Too many people

Tourism is the lifeblood of many Caribbean nations but our evidence strongly suggests that extremely high densities of tourists and residents are harmful to reefs unless environmental regulations to protect reefs are comprehensive, stringent, and effectively enforced. All locations with > 2635 visitors/km^2/year have < 14% coral cover except for Bermuda with 39% (Table 7, Fig. 22). Likewise, islands with substantially > 500 residents/km^2 have < 15% coral cover except for Bermuda. The situation at Bermuda reflects exceptionally effective regulations and the infrastructure to enforce them, as well a greater level of economic well being that obviates the need for subsistence fishing. But without similar protections, the harmful environmental costs of runaway tourism and population growth seem inevitable.

Overfishing

Artisanal fishing for subsistence is crucial to most Caribbean economies but the consequences have been catastrophic for coral reefs. Overfishing caused steep reductions in herbivores, especially parrotfishes, which are vulnerable to all gear types except hook and line. The greatest reductions occurred where fish traps were the favored gear, although low catches today are resulting in increased spearfishing and larger nets.

The severe consequences of the overfishing of parrotfishes for coral survival were generally unappreciated until the abrupt demise of the sea urchin *Diadema* in 1983-1984 that had increasingly become the last important herbivore on Caribbean reefs (Fig 19; Hay 1984; Hughes 1994; Jackson 1997). *Diadema* and parrotfish compete intensely for food (Randall 1961; Lewis and Wainwright 1985; Hay and Taylor 1985; Carpenter 1990b), and their abundance was inversely proportional until 1983. This inverse relationship provides a rigorous proxy for the assessment of the consequences of historical overfishing of parrotfish in the absence of quantitative data for reef fish abundance before 1983 (Table 8).

Most of our analysis of overfishing focused on the fates of 16 reefs for which we have quantitative data on *Diadema* abundance before the die-off, plus coral cover for the three time intervals 1970-1983, 1984-1998, and 1999-2011 (Tables 3, 5, 8). Nine of the 16 reefs were classified as overfished for parrotfishes by 1983, with *Diadema* densities ranging from 6.9-12.4/m^2, whereas the other seven reefs were classified as less fished with *Diadema* densities of 0.5-3.8/m^2. These classifications agreed well with what we could glean from the qualitative literature (Appendix 4).

Reefs where parrotfishes had been overfished before 1984 suffered greater decreases in coral cover (Fig. 23 A-D) and increases in macroalgae (Fig. 23 E-H) than reefs that still had functional populations of parrotfish. Coral cover was independent of *Diadema* densities before 1984 (Fig. 23A) when either *Diadema*, or parrotfish, or both managed to graze down macroalgae to extremely low levels. But all that changed dramatically after the *Diadema* die-off when coral cover became negatively correlated with historical *Diadema* abundance right up to the present day (Fig. 23B, C, D). Conversely, macroalgal cover became positively correlated with historical *Diadema* abundance since there were no longer any abundant herbivores to hold it in check, but the scatter was much greater and correlations weaker and generally not significant (Figs. 23 E-H).

There is also compelling field and experimental evidence for persistent indirect effects of increased macroalgal abundance that strongly impede coral recovery through decreased recruitment and increased disease (Box 1). Coral recruitment has greatly declined since 1984, at least in part due to a decline in the parental brood stock, but there is also strong evidence for active interference by macroalgae.

Macroalgae also induce a wide variety of pathological responses including virulent diseases and may release toxic allelochemicals that disrupt microbial communities associated with corals causing bleaching or death.

Box 1. Harmful effects of increased macroalgal (MA) abundance on larval recruitment and outbreaks of disease of Caribbean reef corals (for further details see text).

Type of study	Observation	Reference
Reduction of coral recruitment and survival of juvenile corals		
Field surveys in St. Croix	Coral recruits most abundant in locations of high grazing pressure and low abundance of non-calcareous (fleshy) algae	Rogers et al. 1984
Fouling panel experiments in Curaçao	20-fold reduction in larval recruitment onto upper surfaces of panels in 1998-2000 compared with 1979-1981 (after versus before mass mortality of Diadema) due to blanketing of the panels by MA	Vermeij 2006
Settlement experiments in Belize	Greater larval recruitment onto substrates covered by crustose coralline algae and low recruitment onto surfaces covered by MA	Arnold and Steneck 2006
Laboratory experiments on Larval behavior	Larval avoidance of substrates with all species of MA or cyanobacteria tested	Kuffner et al. 2000
Field observations in the Bahamas	2 to 3-fold increase in coral recruitment at sites where parrotfishes have increased and MA have decreased in protected areas	Mumby et al. 2006, 2007; Mumby and Harborne 2010
Field observations at numerous sites around the Caribbean where Diadema have recently recovered to densities >1/m2	Reduction of MA to very low percent cover and several-fold increases in juvenile corals and coral cover	Edmunds and Carpenter 2001; Carpenter and Edmunds 2006; Idjadi et al. 2001
Pathological responses of corals to proximity to macroalgae		
Laboratory experiments with corals and macroalgae from numerous Caribbean locations	Close proximity or contact with MA results in coral death[1]	Nugues et al. 2004; Smith et al. 2006; Rosenberg et al. 2007; Barott and Rohwer 2012; Morrow et al. 2012
Laboratory experiments at various Caribbean locations	Toxic allelochemicals from macroalgae disrupt microbial communities on coral surfaces and may cause bleaching or death on contact with corals	Rasher and Hay 2010; Rasher et al. 2011

[1]But see Vu et al. 2009

Finally, overfishing may have also indirectly affected the capacity of reefs to recover from damage by hurricanes; something they have routinely done for millions of years (Woodley et al. 1981; Jackson 1991). Over the past few decades, however, corals have increasingly failed to become reestablished on many reefs after major storms (Gardner et al. 2005). We investigated the causes of this apparent shift using the data for the 16 reefs with data from before 1984 in Tables 5 and 8. Coral cover was independent of the long-term probability of hurricane occurrence before 1984 (Fig. 28A), but not afterwards (Fig. 28B). The reasons are obscure because the locations that have experienced the most hurricanes since 1984 were also among the most extremely overfished (Table 8; median for overfished locations = 3 hurricanes since 1984, median for less fished locations = 0 hurricanes since 1984). But it is unlikely to be just a coincidence that the greater vulnerability to storms began just after the *Diadema* die-off, especially given the extraordinary resilience of coral cover at Bermuda after 4 hurricanes since 1984.

Coastal pollution

Almost everyone agrees that coastal pollution is an increasingly serious problem for coral reefs but there are precious few rigorously and consistently collected data comparable to that for Degree Heating Weeks (Table 9). Thus, it is difficult to do more than compile a list of local situations on coral reefs and attempt to generalize as has been done for sedimentation stress (Rogers 1990; Fabricius 2005) and oil spills (Guzmán et al. 1991; Burns et al. 1993, 1994; Guzmán and Holst 1993), but not yet for nutrients.

Nevertheless, limited comparative data for water transparency at three CARICOMP sites based on simple secchi disk observations suggest that water quality on Caribbean reefs is declining greatly (Table 9; Fig. 25). Water transparency declined significantly over 20 years at Carrie Bow Cay due to steep increases in the clearing of land for agriculture and for coastal development in Belize and continued deforestation of the high coastal mountains along the Gulf of Honduras in Guatemala and Honduras (Burke and Sugg 2006; Fig. 25 A-B). Similar declines were observed at La Parguera, Puerto Rico (Fig. 25C). Secchi disk measurements were a standard part of the CARICOMP protocol and it is unfortunate that the measurements were made at so few CARICOMP sites. The results from strongly suggest a very serious decline in water quality that is being widely ignored.

Global climate change

We began our study expecting to document very large and pervasive consequences of coral bleaching but that was resoundingly not the case. Our first analyses were based on the ReefBase compilation of extreme bleaching events that showed no significant relationship between the numbers of extreme events/locality and coral cover at locations across the wider Caribbean, Gulf of Mexico and Bermuda. We next requested and obtained Pathfinder Sea Surface Temperature data from the National Oceanographic Data Center through the assistance of Mark Eakin and Scott Heron. The result is the comprehensive data for degree heating weeks (DHWs) for all 88 localities with coral cover in Table 9.

Graphs of the proportional loss in coral cover in relation to numbers of DHWs in the two years following the 1998, 2005, and 2010 major heating events are surprisingly flat, essentially mirroring our earlier results (Fig. 26). All the slopes are weakly negative but non-significant in spite of the well-documented cases of extreme coral bleaching followed by disease that has severely affected reefs in the USVI and Puerto Rico after 2005 and elsewhere (Miller et al. 2009; Weil et al. 2009). Repeating the analyses using only the data for locations that suffered $\geq$ 8 DHWs gives even weaker and anomalous results.

The reason for the general lack of correlation is that coral cover at several locations has substantially increased or held steady after extreme heating events (points on or above the lines of zero percent change in Fig. 26. Many of these exceptional locations have either high parrotfish abundance or low macroalgal cover, or both (Fig. 26, Tables 2 and 5). This implies that high grazing pressure and/or low macroalgal abundance may have somehow increased the resilience of corals to the otherwise fatal combination of massive bleaching followed by disease, which has been the generally accepted pattern for the consequences of extreme heating events. Our results do not imply that coral bleaching is unimportant or that it will not become even more dangerous in the future (Hoegh-Guldberg et al. 2007). But they do belie any regionally consistent effects of coral bleaching up to now, and suggest that strong measures to protect parrotfish and other grazers could make an important difference for the survival of corals in an increasingly warmer world.

None of this would necessarily apply to the deleterious effects of ocean acidification which has not been treated here because it is too soon to know what the effects are now much less in the future. If present trends of decreased pH continue, however, the ability of corals and other calcareous reef species to deposit skeletons will be increasingly but perhaps not fatally compromised (Hoegh-Guldberg et al. 2007; Pandolfi et al. 2011).

Invasive species

The explosion of exotic Pacific lionfish throughout the wider Caribbean has wreaked havoc in Caribbean fish communities. But as serious as the potential consequences may be, they pale in comparison to the introduction of the pathogen that caused the die-off of *Diadema antillarum* or the effects of WBD on acroporid corals. The first occurrence of *Diadema* mass mortality at the Caribbean entrance of the Panama Canal (Lessios 1988) coupled with the enormous increases in bulk carrier shipping and the salt water aquarium trade in the 1960s and 1970s (Carlton 1996; Drake et al. 2007) can hardly be a coincidence.

The Caribbean is effectively a Mediterranean sea and has been the oceanographically and geographically most isolated tropical ocean on the planet since the continuous emergence of the Isthmus of Panama 3-5 million years ago (Jackson and O'Dea 2013). This strongly suggests that, by analogy to the fates of the original Americans after their first contact with Europeans (Crosby 1986; Mann 2005), Caribbean species should be exceptionally prone to the impact of introduced diseases. And this appears to be the case. We know of no other examples of the virtual elimination due to disease of any marine species throughout the entire extent of the Indian or Pacific oceans comparable to the demise of Caribbean *Diadema* and acroporids. This interpretation is also consistent with the failure to discern any environmental shift in the 1970s that could have triggered the outbreak of disease.

Concluding remarks

Overpopulation in the form of too many tourists and overfishing appear to be the two best predictors to date of the overall decline in Caribbean coral cover over the past 30 or more years. Coastal pollution is undoubtedly increasingly significant but there are too little data. Increasingly warming seas have caused extensive coral bleaching and mortality and pose an increasingly ominous threat in the future. But so far extreme heating events appear to have been of surprisingly limited and local significance.

5. RECOMMENDATIONS FOR MANAGEMENT

Our results challenge much of the conventional wisdom about the relative importance of global climate change versus more local impacts of overdevelopment, coastal pollution, and overfishing as the primary drivers of coral reef degradation to date and emphasize the critical importance of historical perspective for coral reef management and conservation (Jackson et al. 2001; Pandolfi et al. 2005; Knowlton and Jackson 2008; Hughes et al. 2010). The threats of climate change and ocean acidification loom very large for the future but have not been the major drivers of the decline of Caribbean corals up to now.

Overemphasis on climate change distracts attention from acute local to regional problems about which much could be done to improve conditions on reefs. It also provides an excuse for managers and governments not to make the hard decisions required to stop overfishing, coastal pollution, and unsustainable development and to do the simple, basic monitoring essential for adaptive management.

Smart decisions can make an enormous difference for the wellbeing of coral reefs and the people and enterprises that depend upon them. No place is close to perfect and everywhere is threatened, but the higher coral cover and comparative resilience to extreme heating events or frequent hurricanes on most reefs in Bermuda, Bonaire, Curaçao, the Venezuelan parks, the Flower Garden Banks, and the Jardines de la Reina in Cuba provide clear examples of what could begin to be achieved by strong and effective environmental regulation (albeit that the regulations greatly differ among these different sites).

Four major recommendations emerge from this report:

1. Adopt robust conservation and fisheries management strategies that lead to the restoration of parrotfish populations, including the listing of the parrotfish in relevant annexes of the Protocol concerning Specially Protected Areas and Wildlife (SPAW protocol) of the UNEP Caribbean Environment Programme. A recommendation to this effect was passed unanimously at the October 2013 International Coral Reef Initiative Meeting in Belize (see Box 2 below).

The most important recommendation based on the evidence of this report is the urgent and immediate need to ban fish traps and fishing of any kind for parrotfish and to severely restrict and regulate all other kinds of fishing throughout the wider Caribbean including spearfishing, gill nets, long lines, and all other destructive fishing practices.

The need for strong fisheries regulations has been obvious for decades (Thompson 1945; Randall 1963; Munro 1983; Hay 1984; Hughes 1994; Jackson 1997; Jackson et al. 2001), but only the managers of Bermuda, Los Roques, the Flower Gardens Banks, Bonaire, Jardines de la Reina, and most recently Belize have taken effective action. Given current trends, reef corals can be expected to become ecologically extinct in the Florida Keys, US Virgin Islands, and most of Jamaica within a decade.

With a few local exceptions, reef associated fish stocks are severely overfished and depleted throughout the wider Caribbean. The market value of remaining fisheries is miniscule compared to the damage fishing does to reefs in terms of lost tourist revenues, coastal protection, and the other ecosystem services reefs provide (Pandolfi et al. 2005). Without effective management and welfare, subsistence fishing of ever-depleted stocks will remain vitally important for the very survival of artisanal fishers living on the edge, but the costs of providing alternative dignified livelihoods for these fishers pale in comparison to the enormous losses of coral reef resources and biodiversity caused by continued overfishing.

2. Simplify and standardize monitoring of Caribbean reefs and make results freely available in real time to promote adaptive management.

There is an urgent need to develop *simple, standardized* monitoring protocols to assess in real time the condition of reefs throughout the wider Caribbean. CARICOMP and AGGRA made important progress but protocols were not consistently followed. Highly elaborate and costly programs in the US Virgin Islands and Florida are impractical to achieve elsewhere.

Most of the information for this report came from individual scientists who generously shared their data. But it took nearly two years to begin to use it reliably because of the diversity of metrics, formatting errors, and internal inconsistencies. Much of the data was unusable because we could not verify locations, depths, and missing metadata. The situation is inexcusable and no one should ever have to go through such an exercise again. In contrast, the Center for Tropical Forest Science and partners monitor 48 standardized forest plots in 22 countries containing 4.5 million trees that are routinely surveyed with up-to-date data readily accessible online (Losos and Leigh 2004).

The results of this report further suggest that regular and consistent monitoring of a small number of key variables would be sufficient to establish status and trends for well-informed adaptive management:

1. Percent cover of corals and macroalgae,
2. Abundance and biomass of parrotfish and *Diadema* abundance,
3. Coral recruitment measured as the density of small colonies < 40 mm,
4. Prevalence of coral disease, and
5. Water transparency measured by a secchi disk

Additional information including abundance of other herbivores and outbreaks of bleaching and coral disease are also highly informative. The bottom line, however, is that reefs with abundant coral, little macroalgae, abundant herbivores, strong coral recruitment, and clear water are healthy by any standard, and those that depart from that pattern are not. We should make sure that all Caribbean nations have all of this simple, basic information before embarking on more complex and challenging endeavors of greater interest to scientists than any value to the managers on the ground.

3. Foster communication and exchange of information
Resources are needed to revitalize the Caribbean node of the GCRMN and other mechanisms to foster exchange of information and cooperation. The GCRMN Workshop in Panama was the first time most of the participants had met or interacted with each other. Ignorance of the work of participants from different countries was great and participants expressed frustration about working in isolation of what was going on elsewhere.

4. **Develop and implement adaptive legislation and regulations** to ensure that threats to coral reefs are systematically addressed, particularly threats posed by fisheries, tourism and coastal development as determined by established indicators of reef health.

We understand that action upon these recommendations will be a matter of local and national socioeconomic and political debate. But the implications of our scientific results are unmistakable: *Caribbean coral reefs and their associated resources will virtually disappear within just a few decades unless all of these measures are promptly adopted and enforced.*

RECOMMENDATION

on addressing the decline in coral reef health throughout the wider Caribbean: the taking of parrotfish and similar herbivores

Adopted on 17 October 2013, at the 28th ICRI General Meeting (Belize City)

Background

The latest report of the Global Coral Reef Monitoring Network (GCRMN), entitled: *Status and Trends of Caribbean Coral Reefs: 1970-2012* is the first report to document quantitative trends of coral reef health based on data collected over the past 43 years throughout the wider Caribbean region.

The results of the study clearly show:

- Coral reef health requires an ecological balance of corals and algae in which herbivory is a key element;
- Populations of parrotfish are a critical component of that herbivory, particularly since the decline of *Diadema* sea urchins in the early 1980s;
- The main causes of mortality of parrotfish are the use of fishing techniques such as spearfishing and, particularly, the use of fish traps.

The Report further identifies that overfishing of herbivores, particularly parrotfish, has been the major drivers of reef decline in the Caribbean to date, concluding that management action to address overfishing at the national and local levels can have a direct positive impact on reef health now and for the future. *In some areas of the wider Caribbean (for example Bermuda and the Exuma Cays Land and Sea Park in the Bahamas, and more lately in Belize and Bonaire), active management including bans on fish traps, has led to increases in parrotfish numbers and consequent improvement in reef health and resilience to perturbations including hurricanes. This is in contrast to other areas within the Caribbean, where heavily fished reefs lacked the resilience to recover from storm damage.*
Positive impacts on reef health demonstrably have spill over effects on local economies, including the potential for alternative livelihoods to fishing, thanks to increased tourism revenues, replenishment of fish stocks and restoration of ecosystem services such as shoreline protection.

It is recognised that in the Caribbean there are varying levels of community reliance on fishing in general and the taking of parrotfish in particular. However, in light of the evidence now available, and in accordance with ICRI's Framework for Action cornerstone of 'integrated management' (which includes fisheries management), the International Coral Reef Initiative would like to highlight the benefits of strong management to protect reefs from overfishing, and urges immediate action to effectively protect parrotfish and similar herbivores.

Accordingly, the International Coral Reef Initiative urges Nations and multi-lateral groupings of the wider Caribbean to:

1. ***Adopt*** conservation and fisheries management strategies that lead to the restoration of parrotfish populations and so restore the balance between algae and coral that characterises healthy coral reefs;

2. ***Maximise*** the effect of those management strategies by incorporating necessary resources for outreach, compliance, enforcement and the examination of alternative livelihoods for those that may be affected by restrictions on the take of parrotfish;
3. ***Consider*** listing the parrotfish in the Annexes of the SPAW Protocol (Annex II or III) in addition to highlighting the issue of reef herbivory in relevant Caribbean fisheries fora;
4. ***Engage*** with indigenous and local communities and other stakeholders to communicate the benefits of such strategies for coral reef ecosystems, the replenishment of fisheries stocks and communities' economy.

Annex: *Executive Summary - Status and Trends of Caribbean Coral Reefs; 1970-2012, GCRMN Report*

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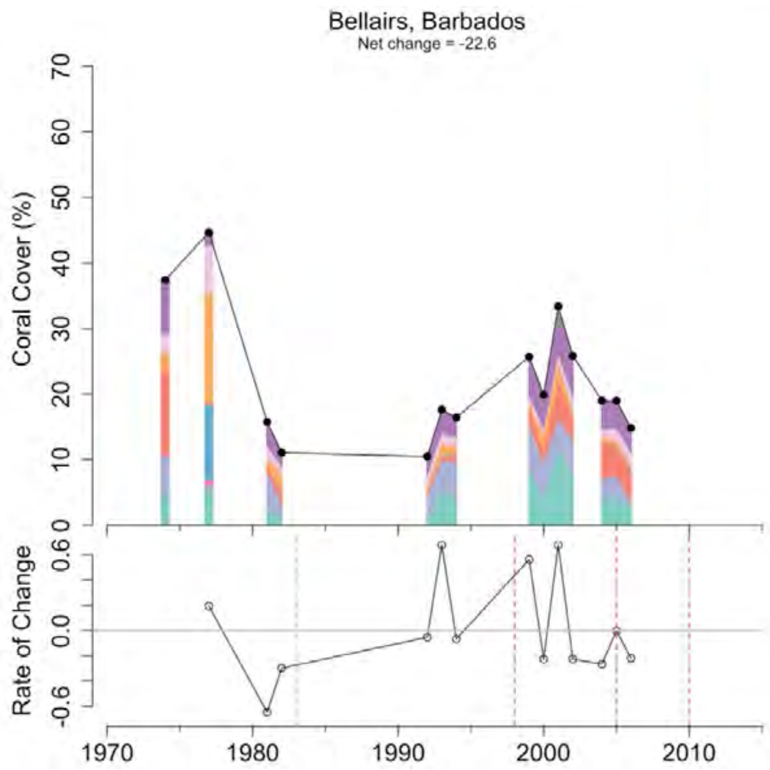
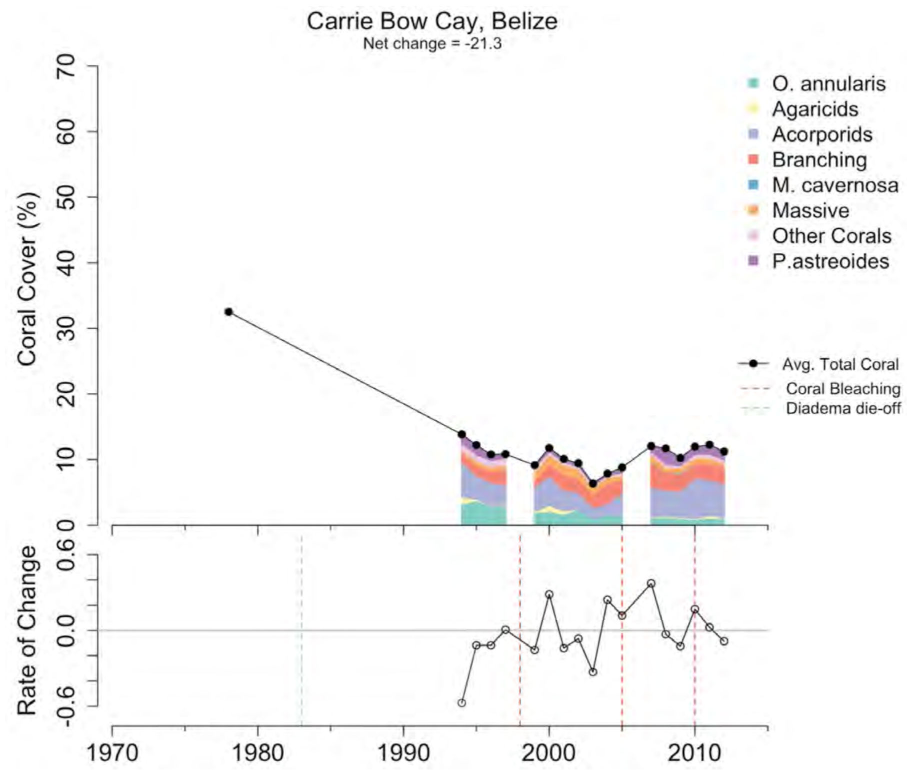
APPENDICES

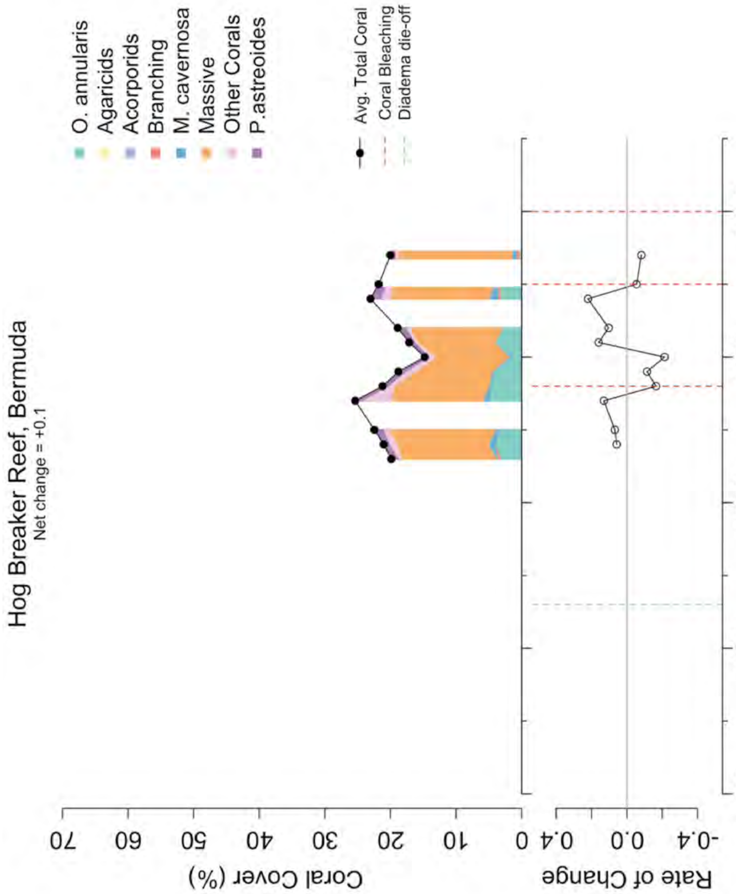
Appendix I: Database structure

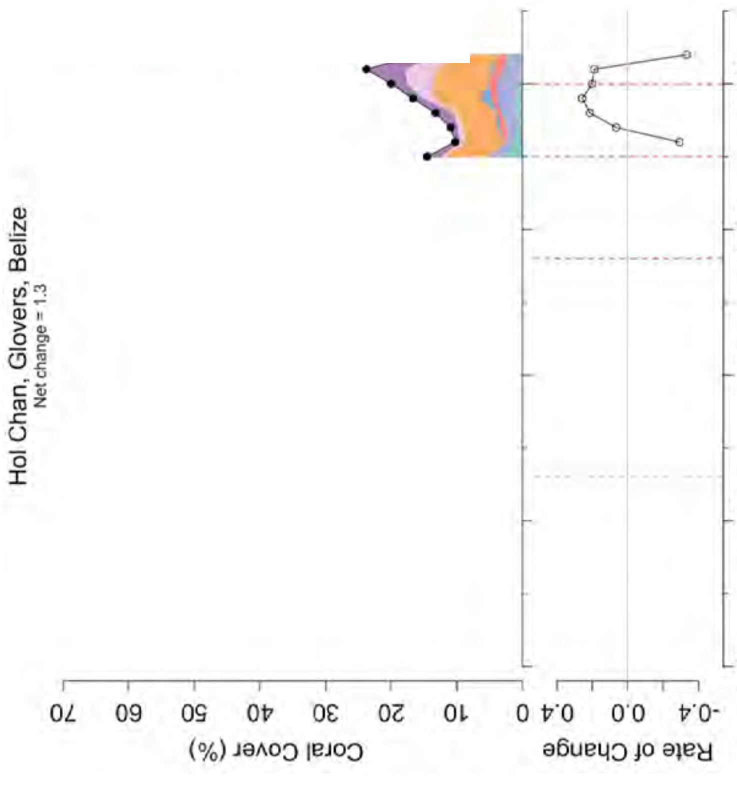
Field	Field Options
Contact Person	
DataLevel	Raw; Summarized with error; Summarized without error
Latitude	
Longitude	
Country	Antigua & Barbuda; Bahamas; Barbados; Belize; British Virgin Islands; Cayman Islands; Columbia; Costa Rica; Cuba; Dominica; Dominican Republic; French Antilles; Grenada; Guatemala; Haiti; Honduras; Jamaica; Mexico; Netherlands Antilles; Panama; Puerto Rico; St. Kitts & Nevis; St. Lucia; St. Vincent & the Grenadines; Trinidad & Tobago; Turks & Caicos; USA; USVI; Venezuela
Location	
ReefSite	
Replicate ID	
Management	No-take MPA; Restricted take MPA; Restricted take MPA; Restricted land-use MPA; No management; Other (explain)
StartYearManagement	
StartYear	
EndYear	
ReefType	Barrier Reef; Deep Reef; Fringing Reef; Hard Bottom; Patch Reef; Spur and Groove; Bank Reef; Atoll; Back Reef
ReefZone	A. cervicornis Zone; A. palmata Zone; Escarpment; Fore Reef Slope; Gorgonian Zone; Reef Crest; Reef Flat; Ridge; Trough; Lagoon
ReefSlope	Flat ; Gentle; Steep; Wall
WaveExposure	Exposed; Protected; Semi-protected
SampleDesign	Random; Selective; Haphazard; Stratified Random
SamplingMethod	Belt Transect; Chain Transect; Linear-Point Intersept Transect; Photo Quadrat; Photo Transect; Quadrat; Video Transect; Continuous transect; Visual estimate
Permanent	Yes; No
SamplingUnit	Single; Multiple
NoOfReplicates	
SampleArea	
SampleAreaUnit	meters; feet
NumberPointsSampled	
MinDepth	
MaxDepth	
MedianDepth	
DepthUnit	meters; feet
Published	Yes-reports; Yes-papers; No
Reference	
UrchinSamplingMethod	Belt Transect; Chain Transect; Linear-Point Intersept Transect; Photo Quadrat; Photo Transect; Quadrat; Video Transect
UrchinSamplingNoOfReplicates	Single; Multiple
UrchinAreaSurveyed	
UrchinAreaSurveyedUnit	meters; feet
PercentTotalCoralIncludesMillepora?	Yes; No

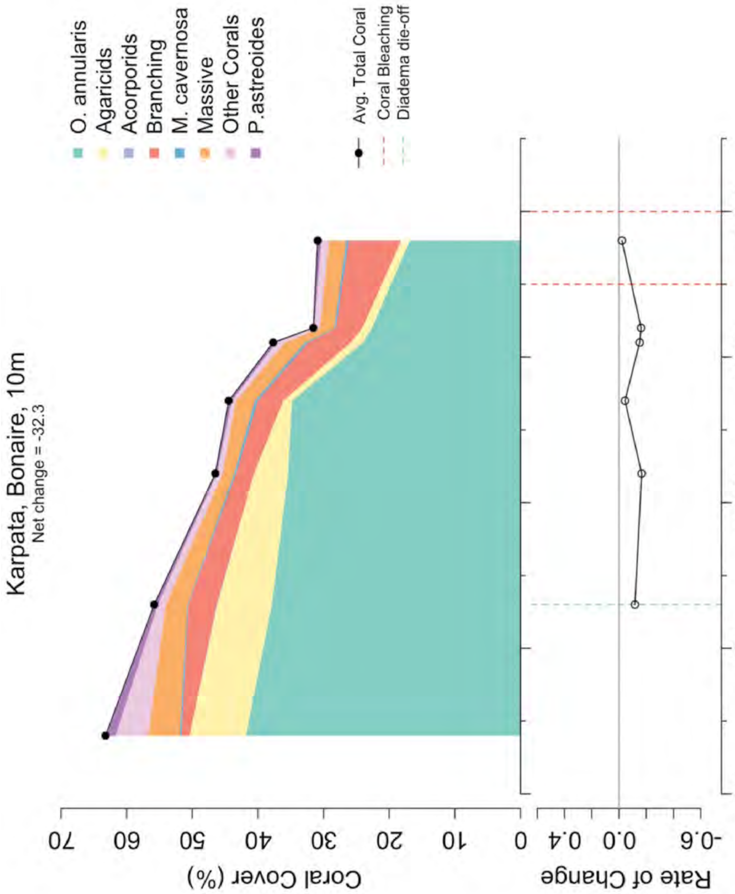
Appendix II: Timelines of coral cover and composition for 40 reef sites

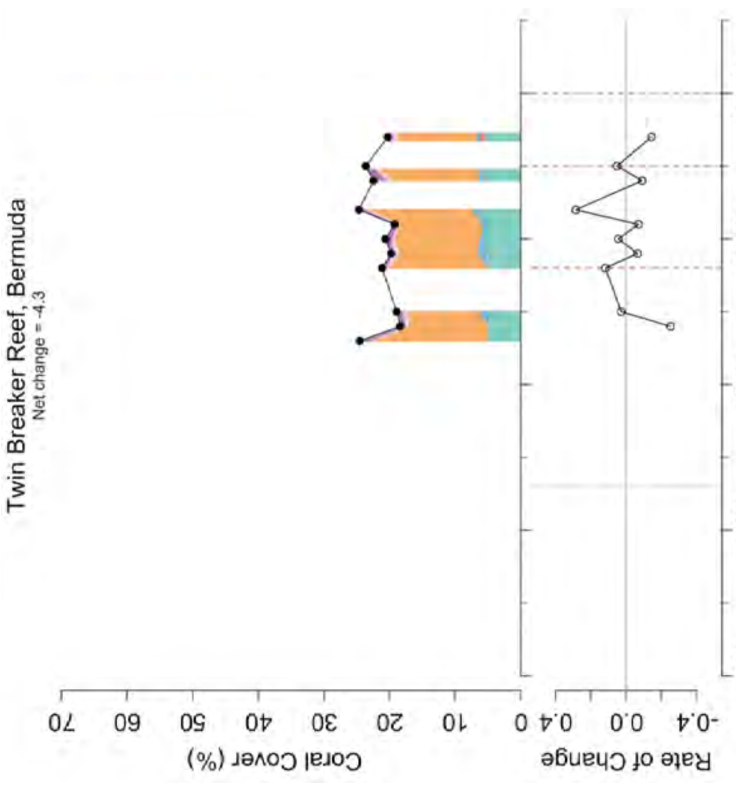


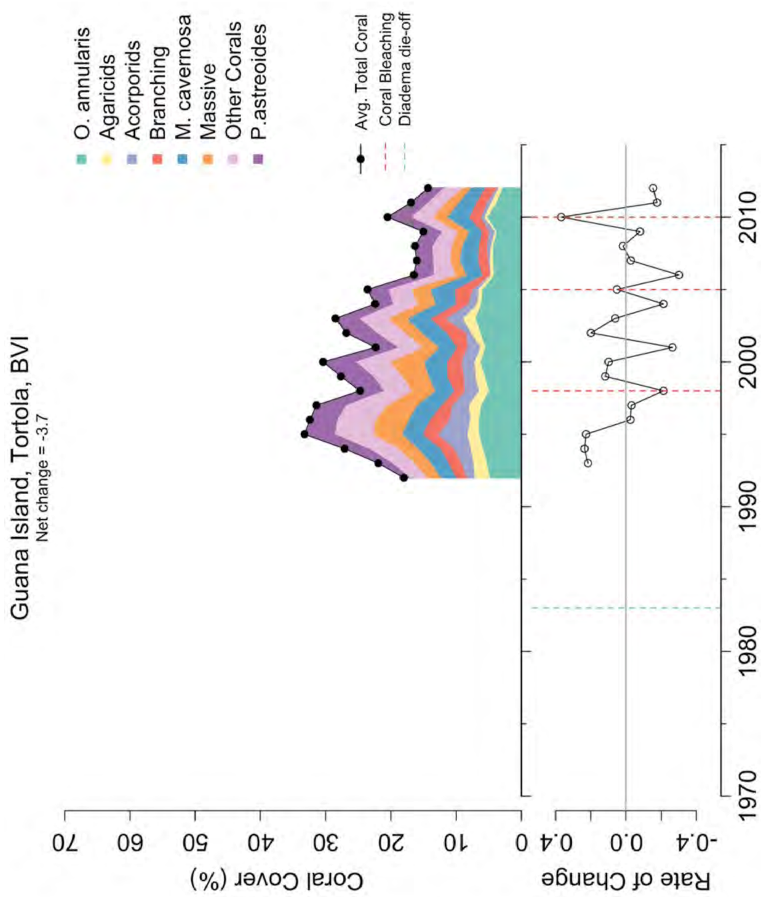


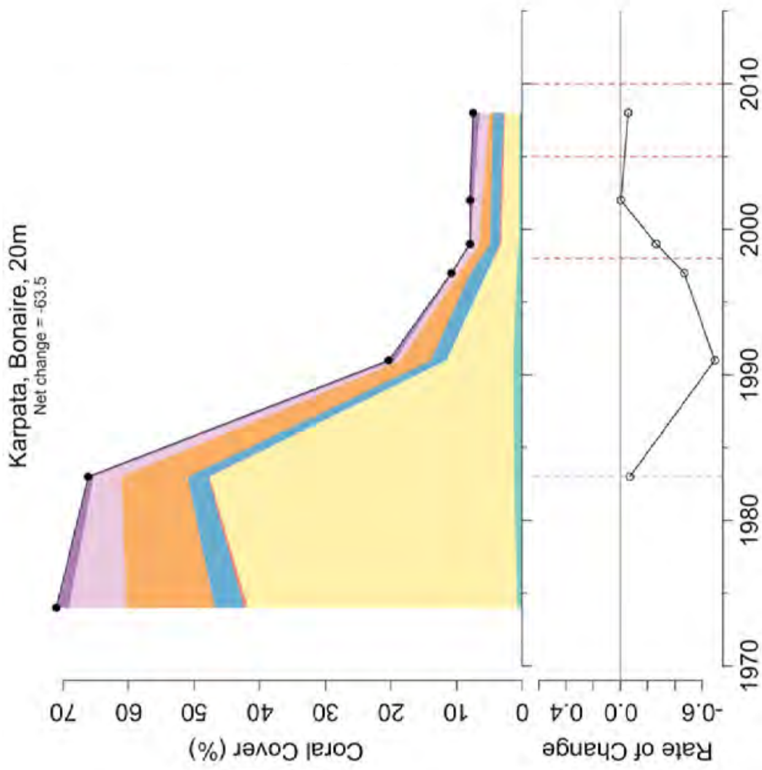

Hog Breaker Reef, Bermuda
Net change = +0.1
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

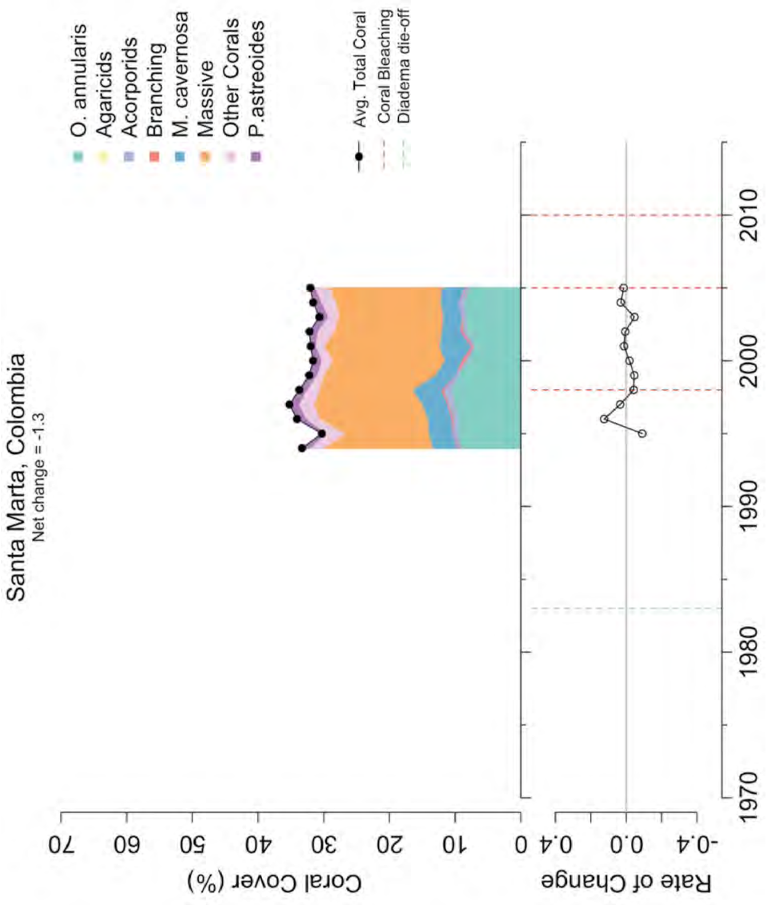

Hol Chan, Glovers, Belize
Net change = 1.3
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

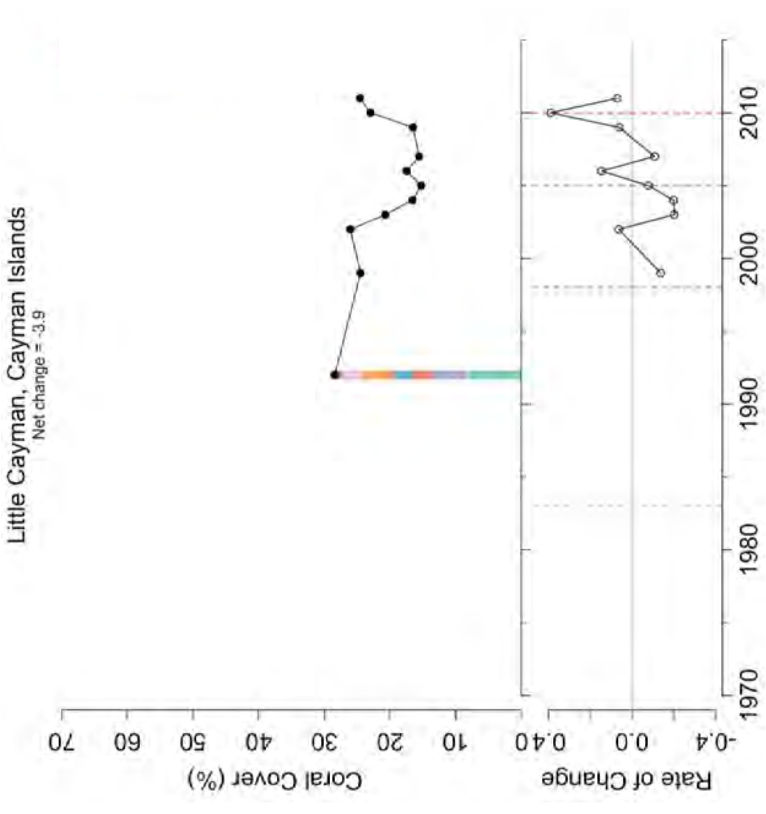

Karpata, Bonaire, 10m
Net change = -32.3
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

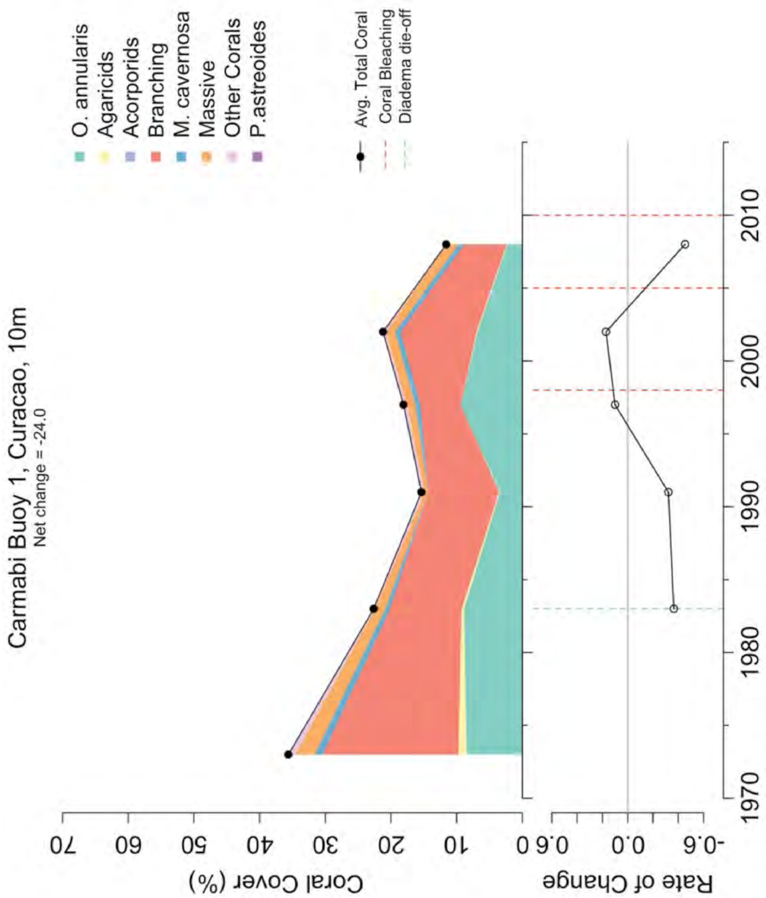

Twin Breaker Reef, Bermuda
Net change = -4.3
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

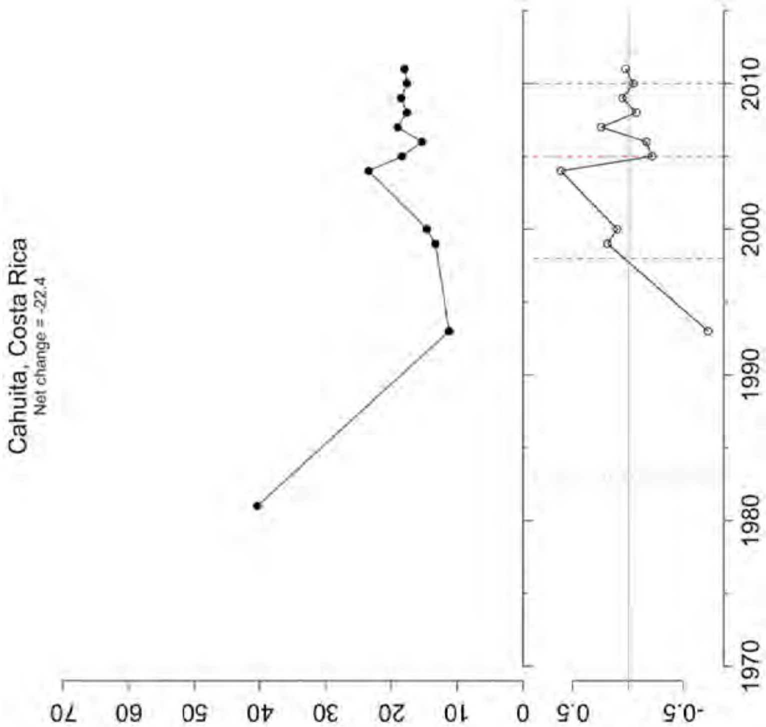

Guana Island, Tortola, BVI
Net change = -3.7
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

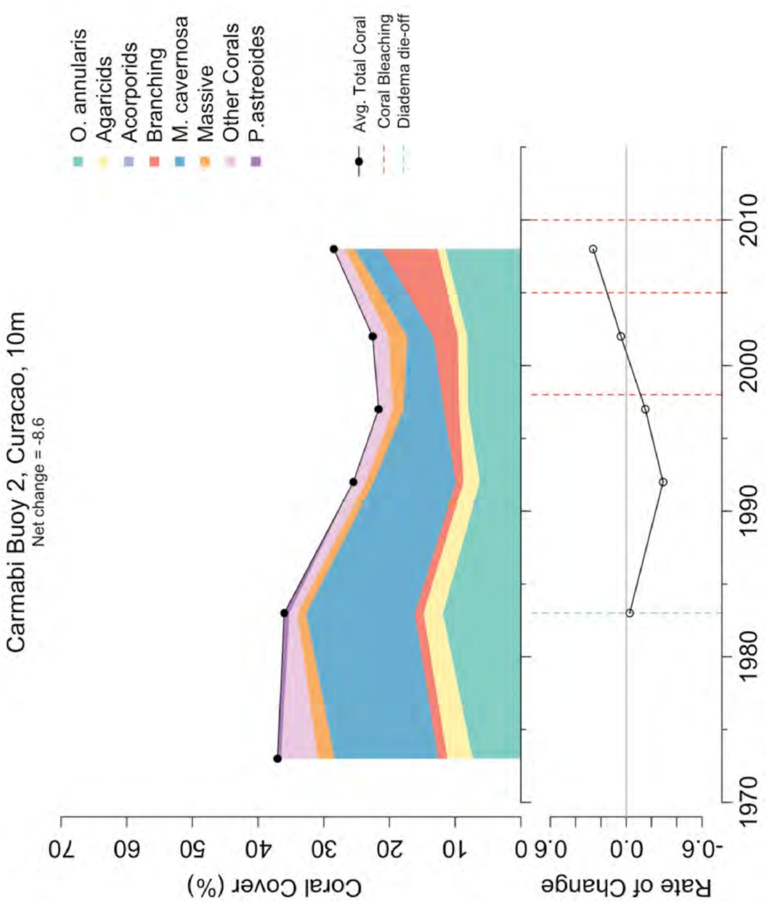

Karpata, Bonaire, 20m
Net change = -63.5
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

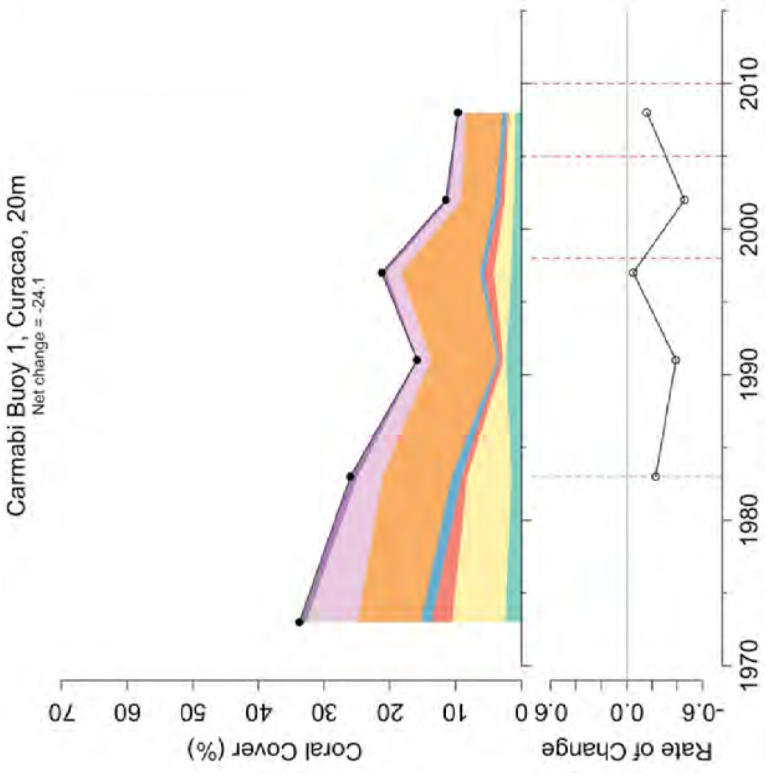

Santa Marta, Colombia
Net change = -1.3
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change

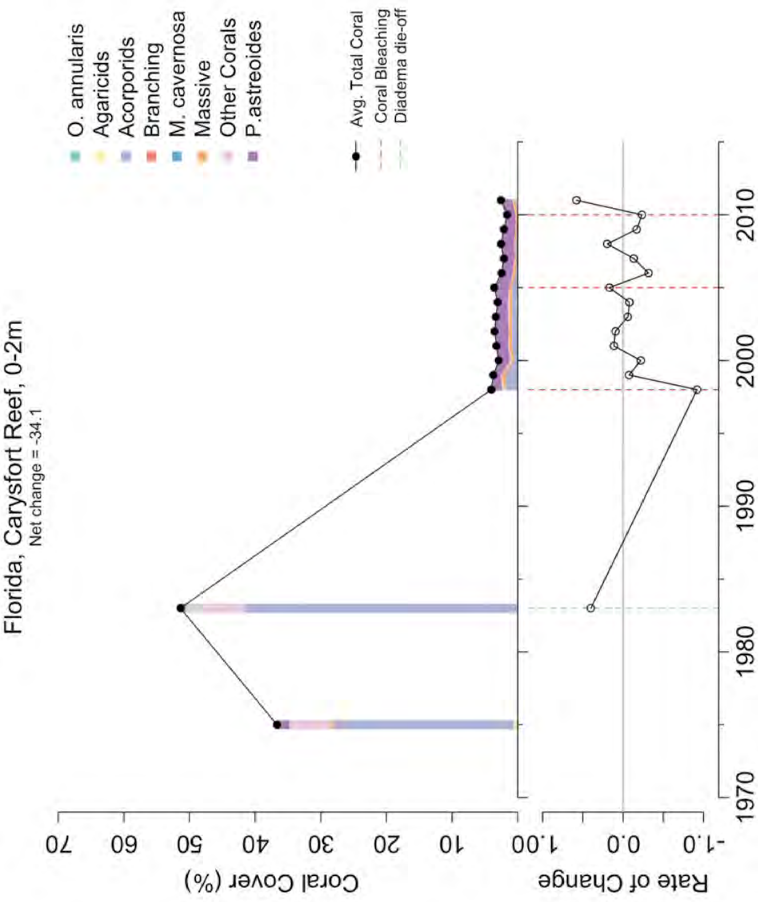

Little Cayman, Cayman Islands
Net change = -3.9
Coral Cover (%)
Rate of Change

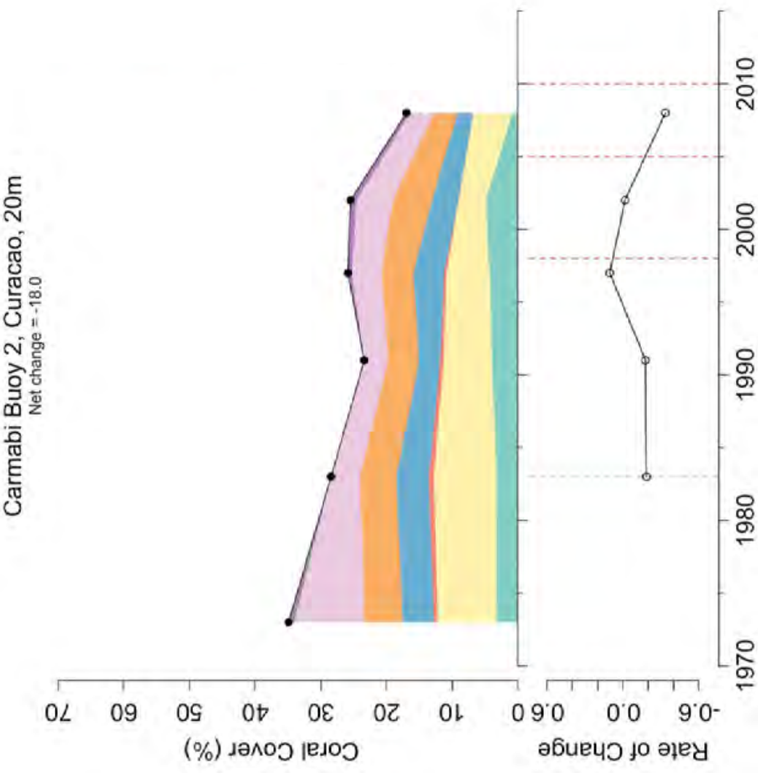

Carmabi Buoy 1, Curacao, 10m
Net change = -24.0
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

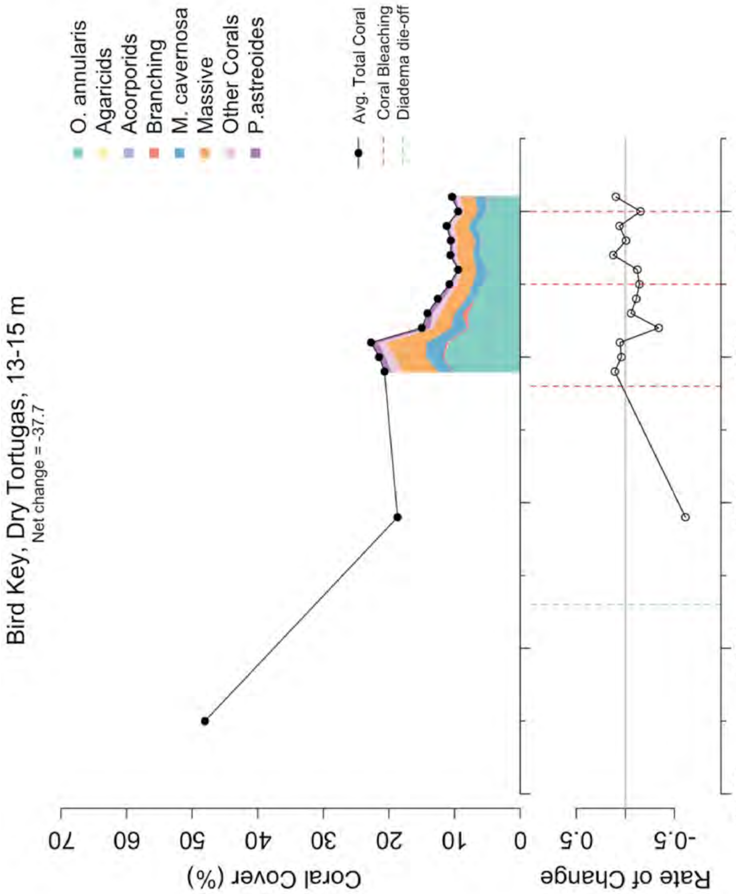

Cahuita, Costa Rica
Net change = -22.4
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

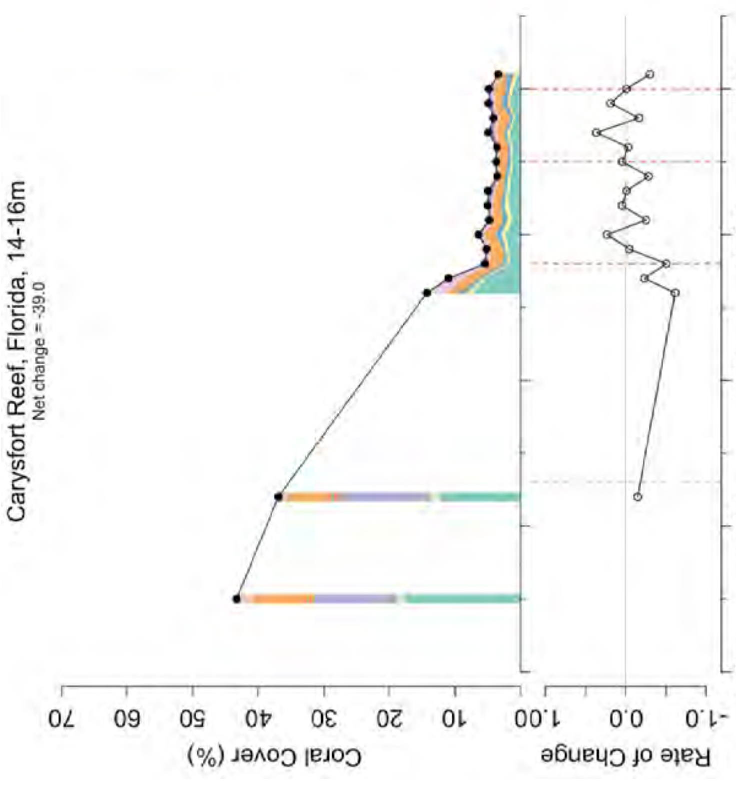

Carmabi Buoy 2, Curacao, 10m
Net change = -8.6
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
0.6
0.0
-0.6
1970
1980
1990
2000
2010

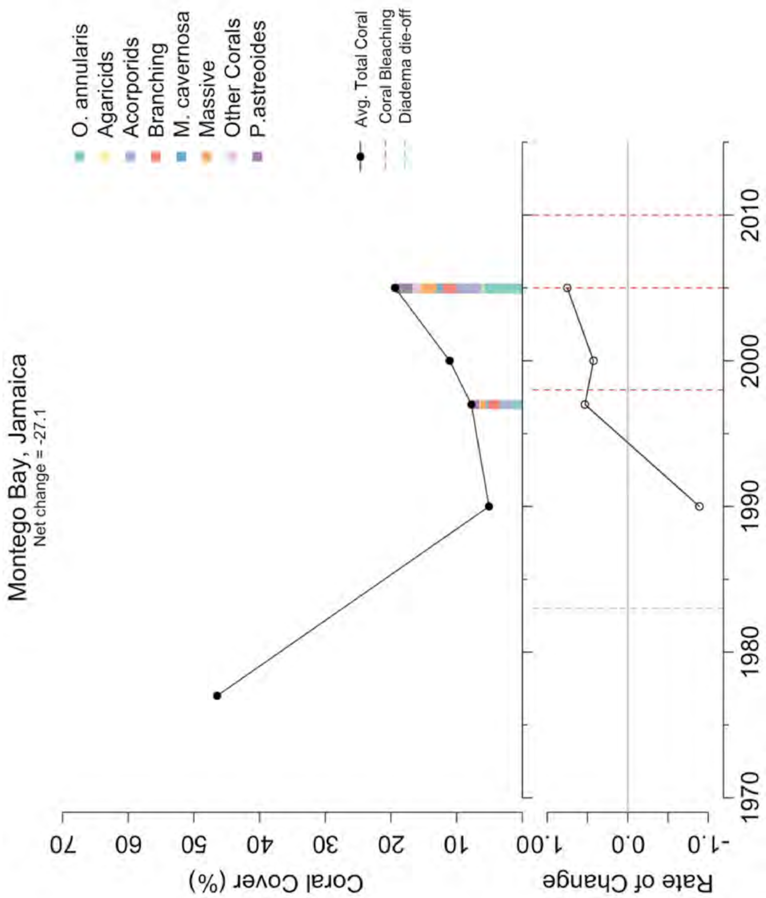

Carmabi Buoy 1, Curacao, 20m
Net change = -24.1
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
0.6
0.0
-0.6
1970
1980
1990
2000
2010

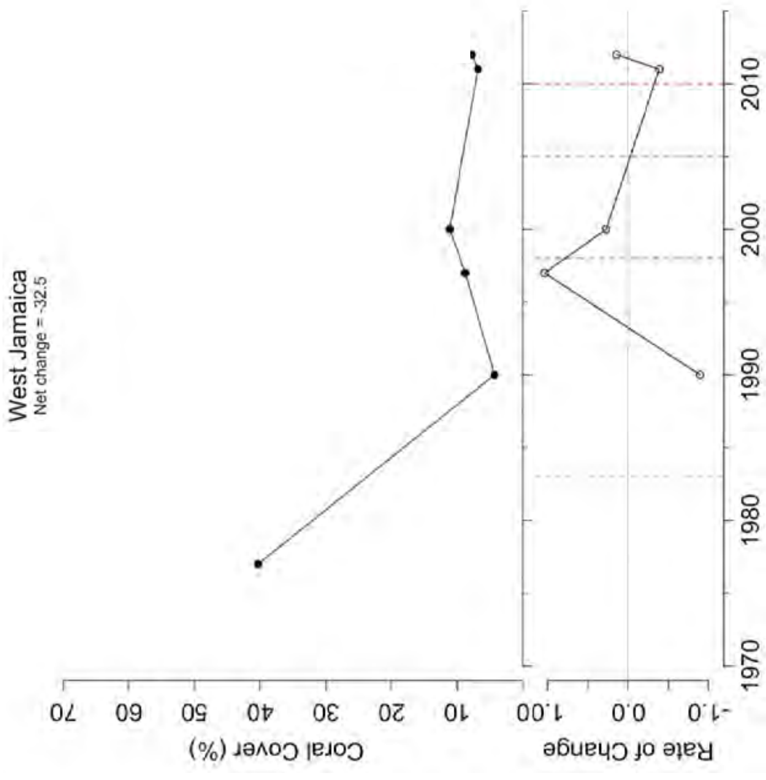

Florida, Carysfort Reef, 0-2m
Net change = -34.1
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change

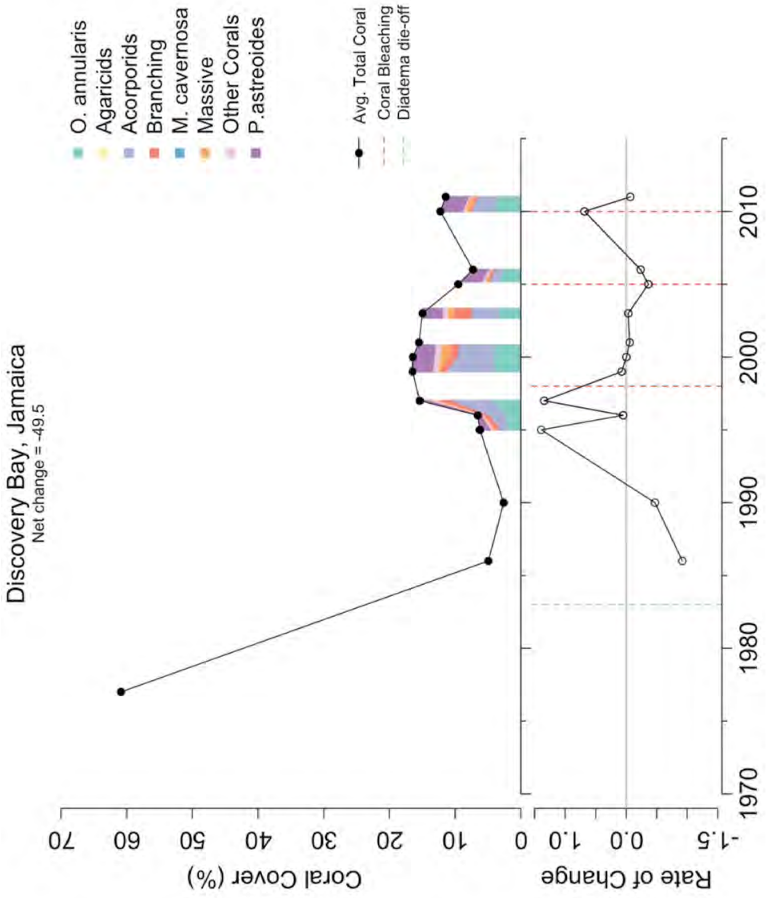

Carmabi Buoy 2, Curacao, 20m
Net change = -18.0
Coral Cover (%)
Rate of Change


Bird Key, Dry Tortugas, 13-15 m
Net change = -37.7
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

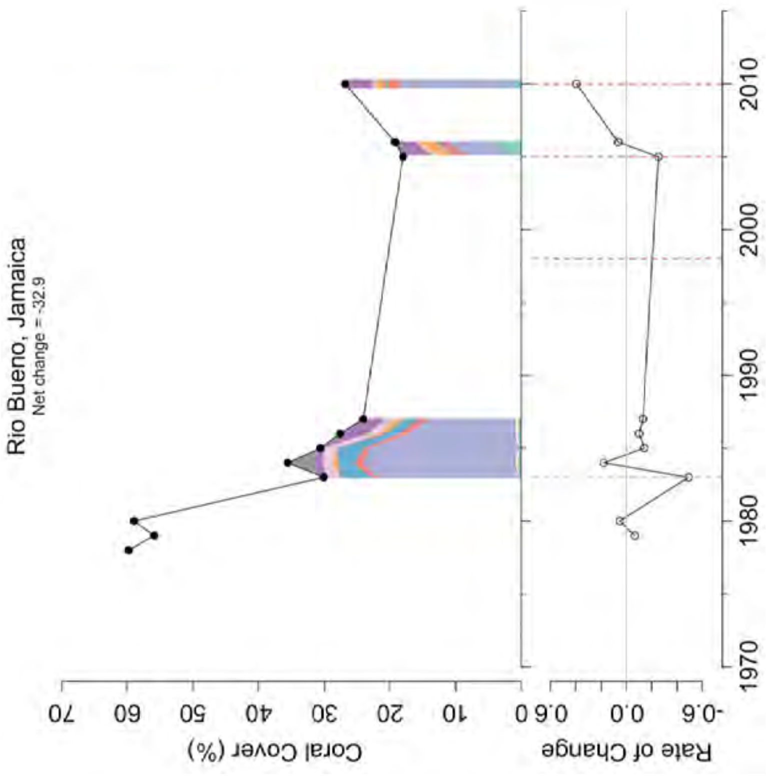

Carysfort Reef, Florida, 14-16m
Net change = -39.0
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010


Montego Bay, Jamaica
Net change = -27.1
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

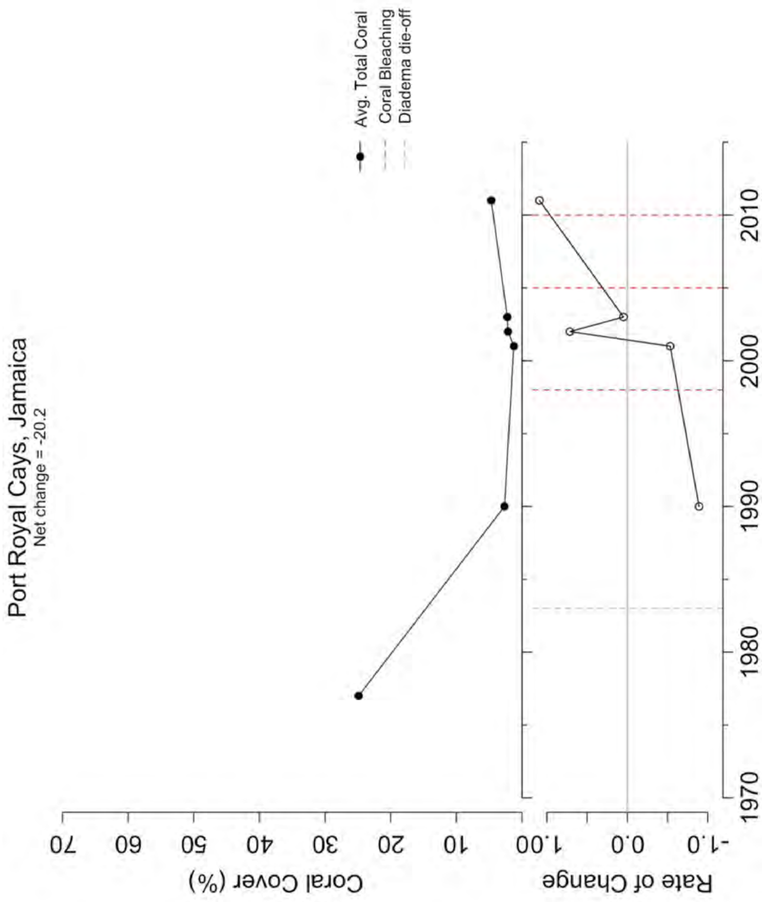

West Jamaica
Net change = -32.5
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

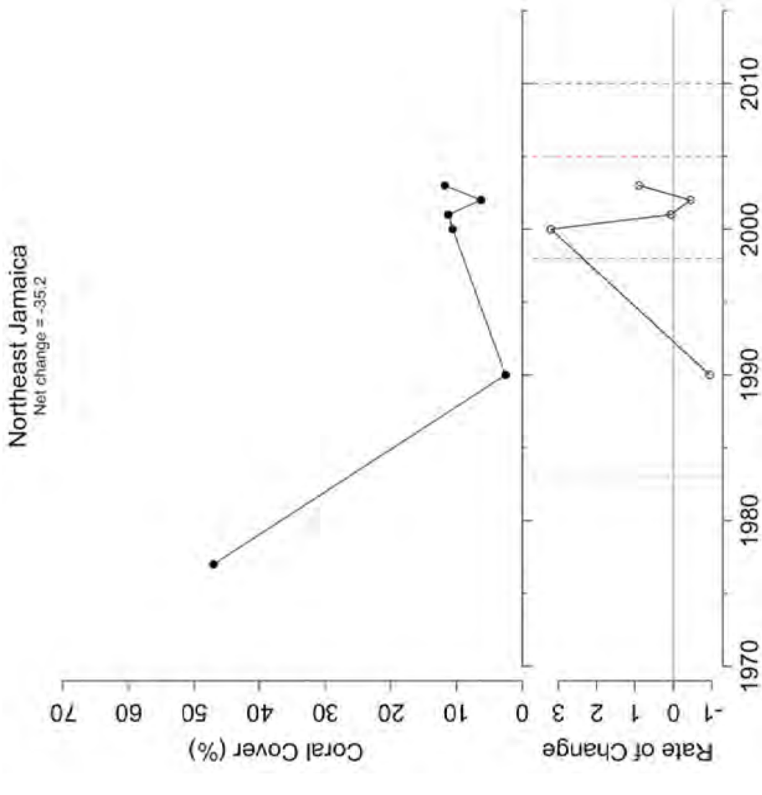


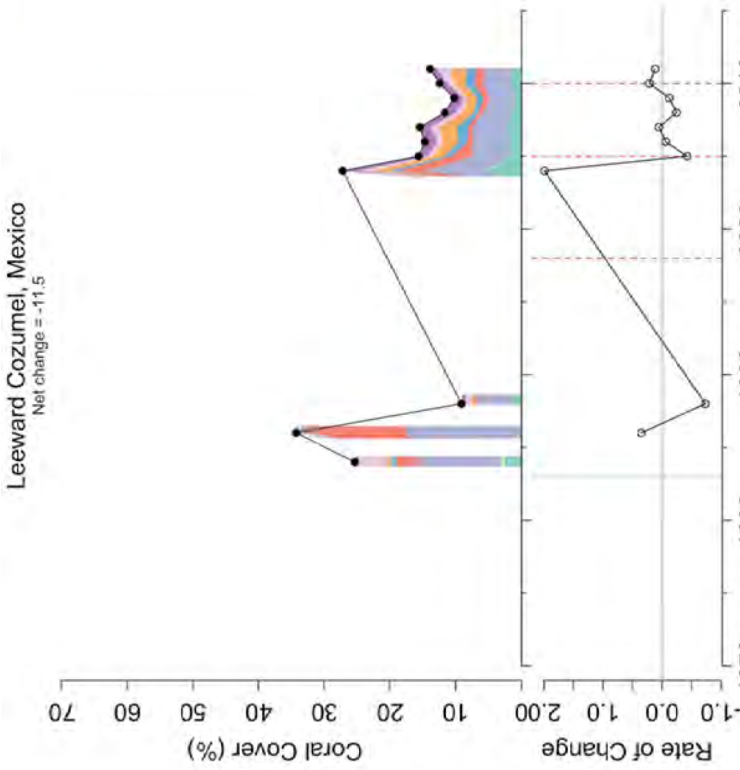
Discovery Bay, Jamaica
Net change = -49.5
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change

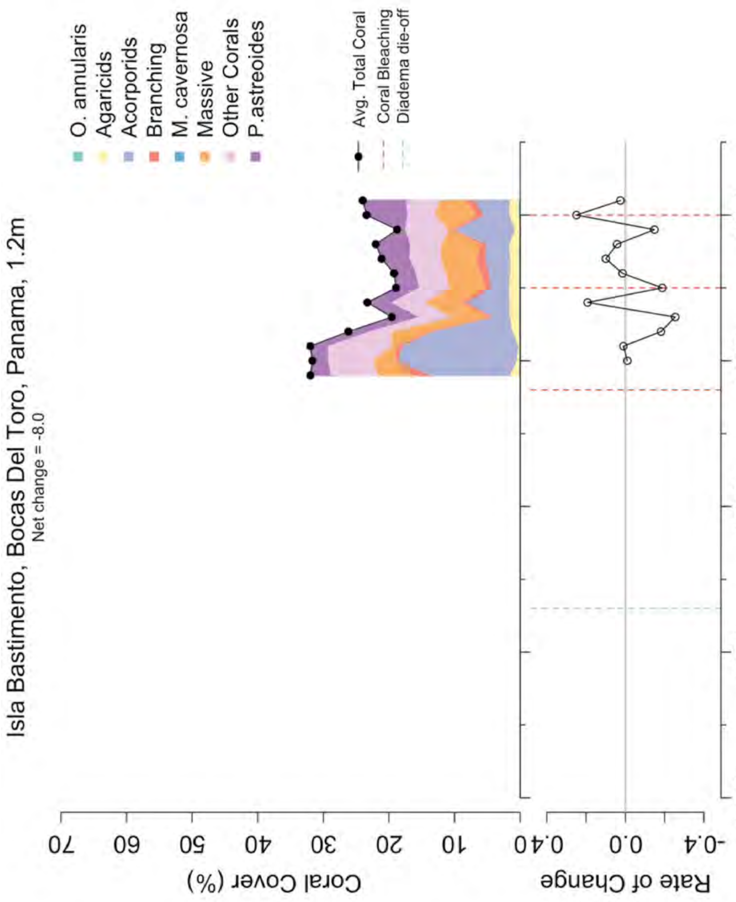


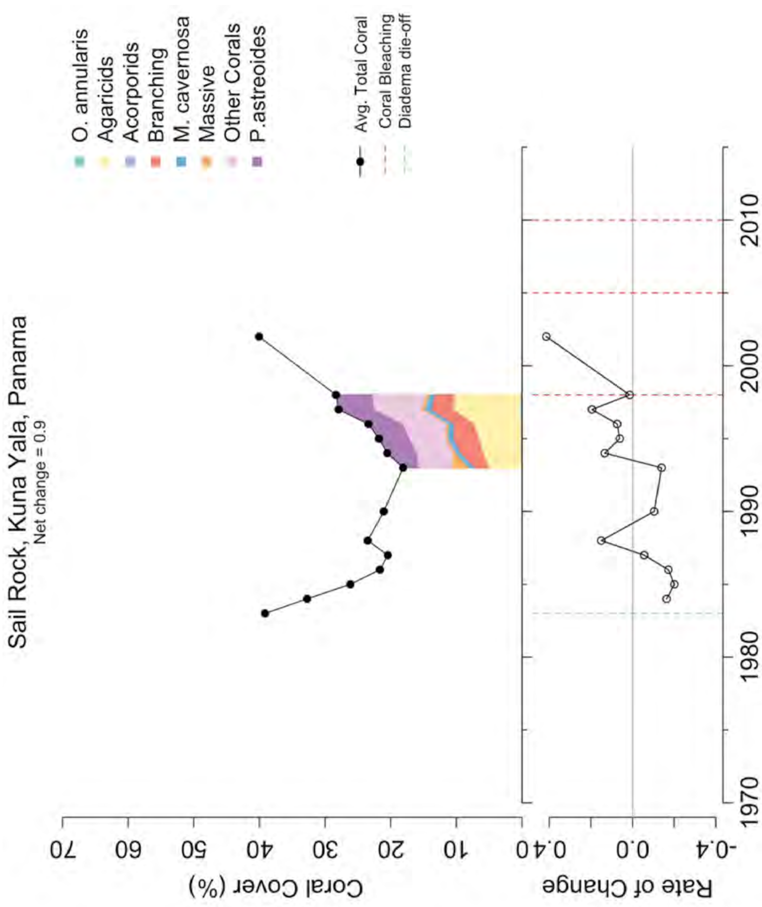
Rio Bueno, Jamaica
Net change = -32.9
Coral Cover (%)
Rate of Change

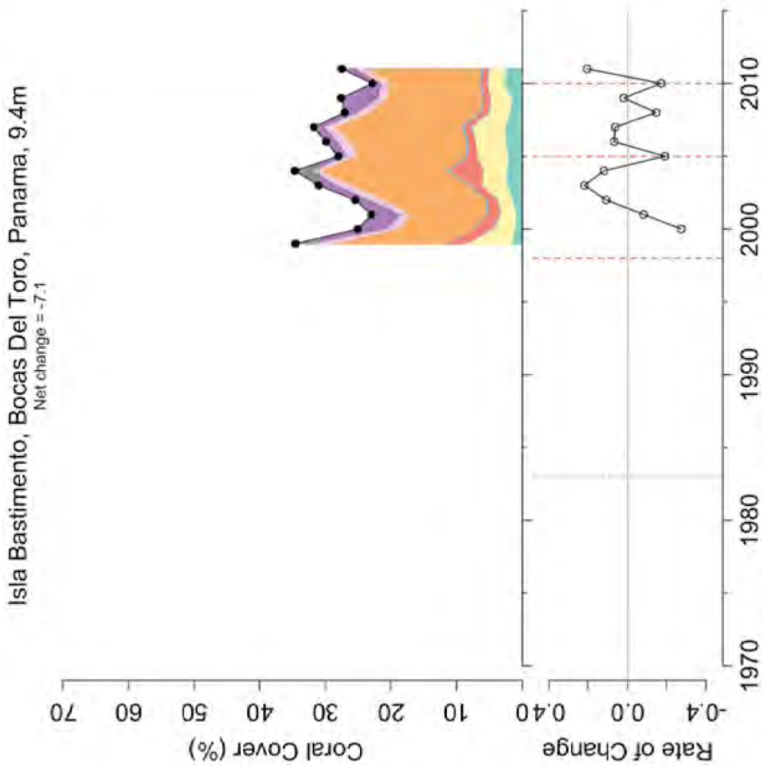

Port Royal Cays, Jamaica
Net change = -20.2
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

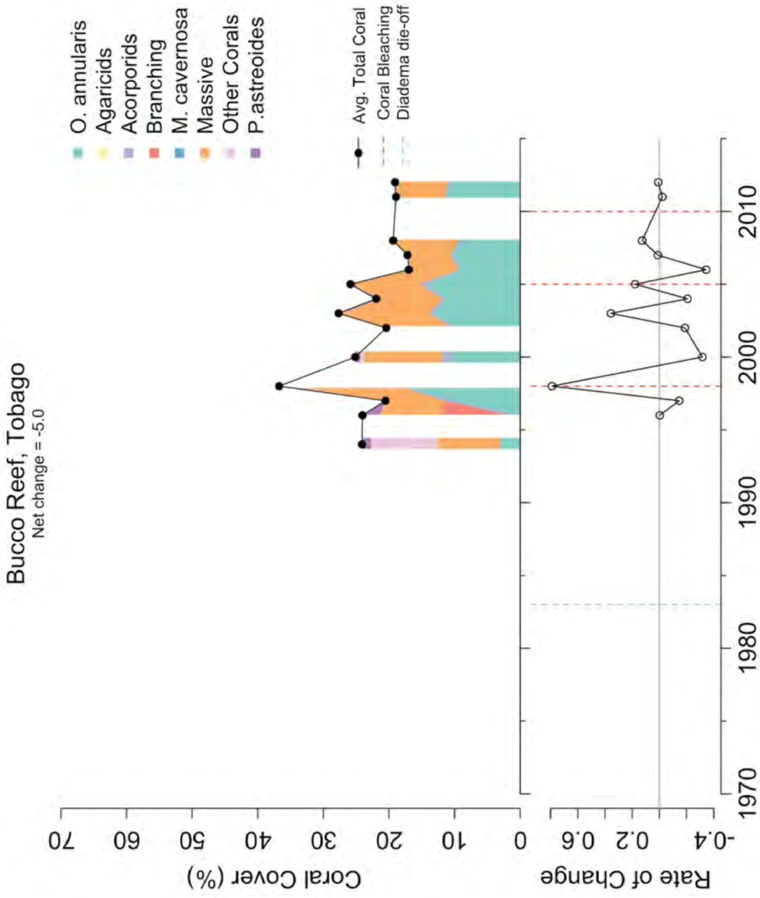

Northeast Jamaica
Net change = -35.2
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

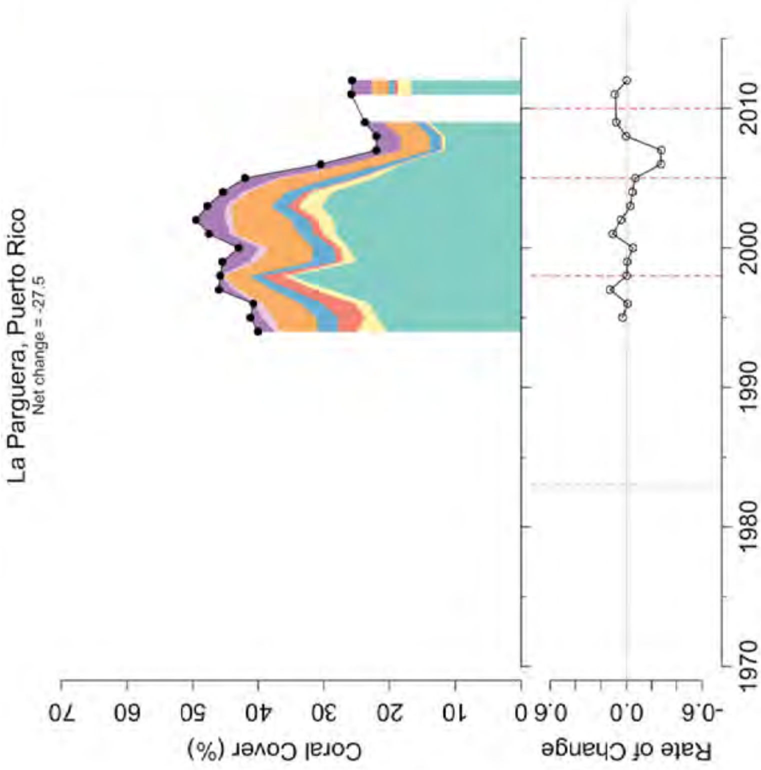

Leeward Cozumel, Mexico
Net change = -11.5
Coral Cover (%)
Rate of Change
0 10 20 30 40 50 60 70
-1.0 0.0 1.0 2.00
1970 1980 1990 2000 2010

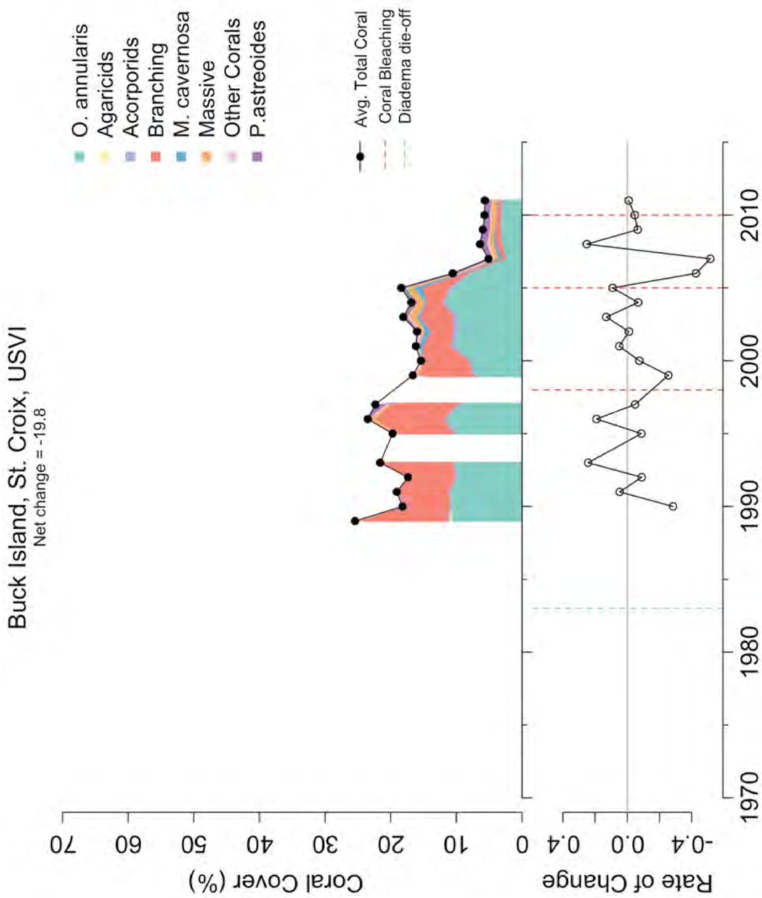

Isla Bastimento, Bocas Del Toro, Panama, 1.2m
Net change = -8.0
Coral Cover (%)
Rate of Change
0 10 20 30 40 50 60 70
-0.4 0.0 0.40
1970 1980 1990 2000 2010
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off

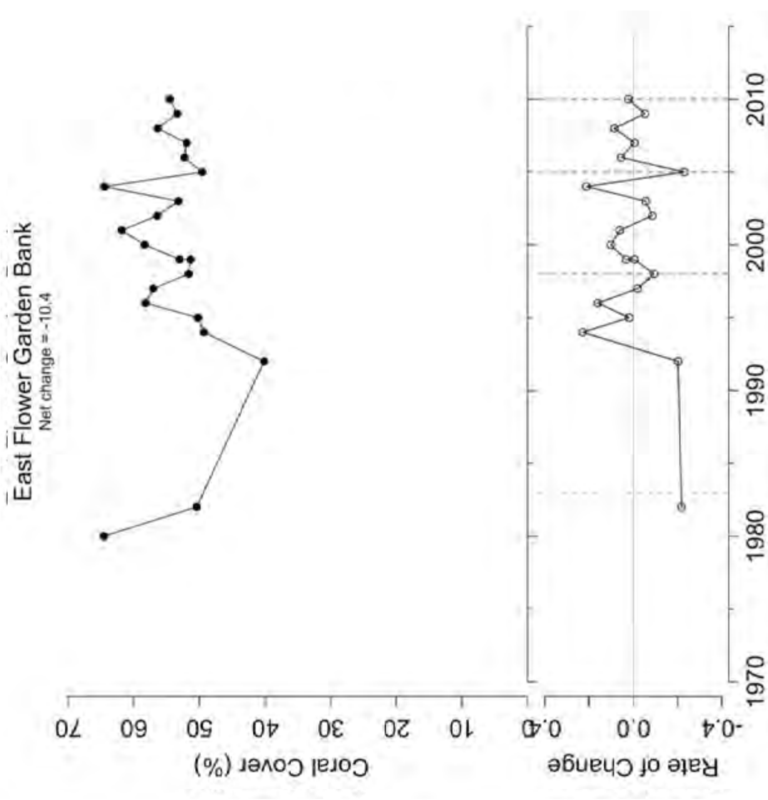

Sail Rock, Kuna Yala, Panama
Net change = 0.9
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
0.4
0.0
-0.4
1970
1980
1990
2000
2010

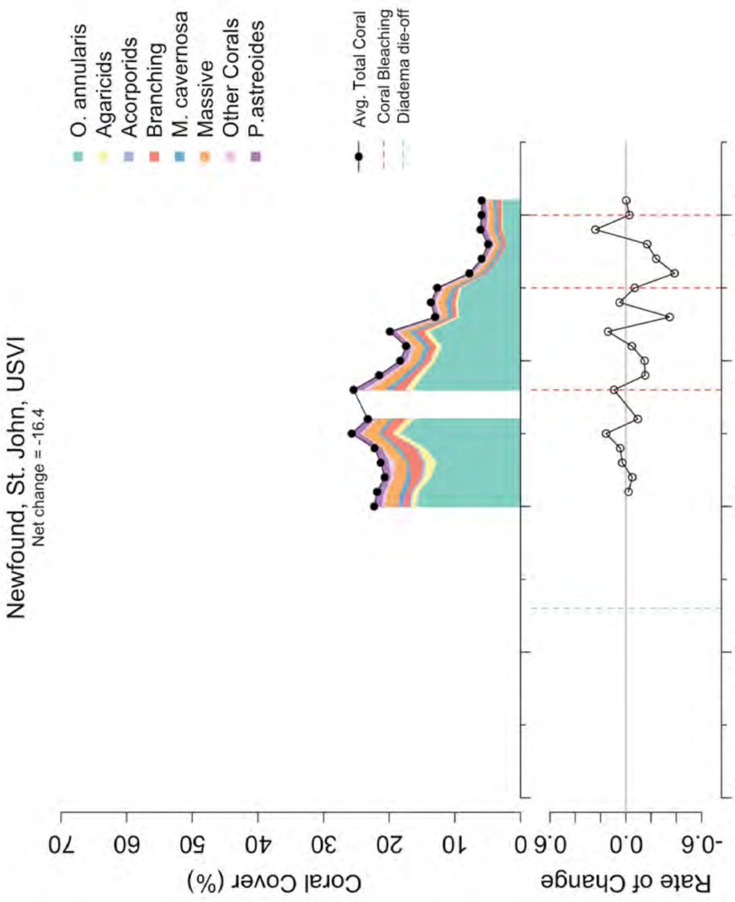

Isla Bastimento, Bocas Del Toro, Panama, 9.4m
Net change = -7.1
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
0.4
0.0
-0.4
1970
1980
1990
2000
2010

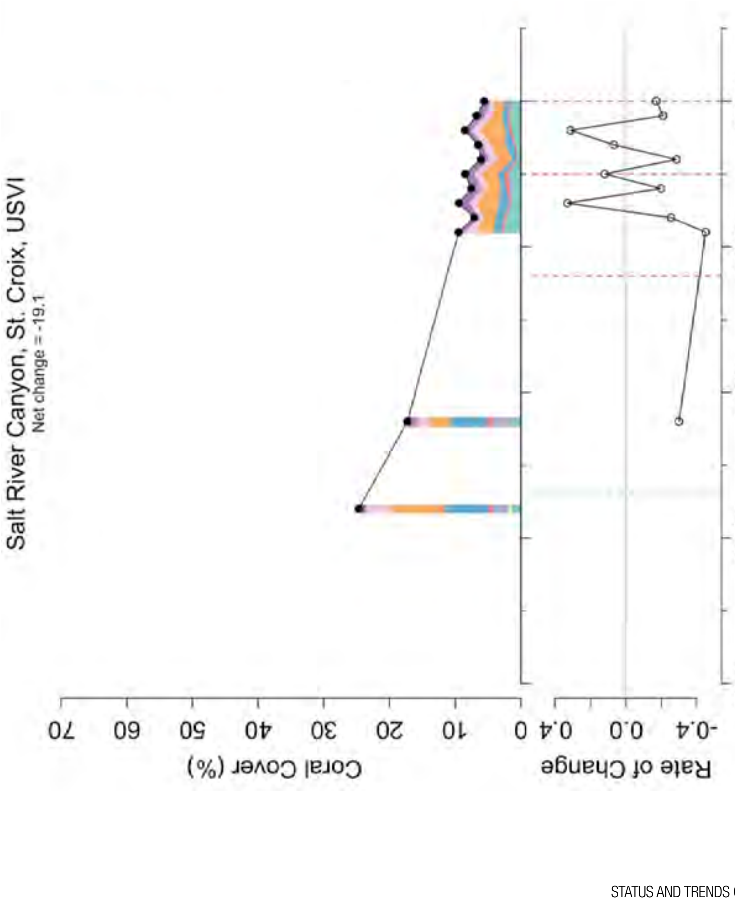

Bucco Reef, Tobago
Net change = -5.0
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

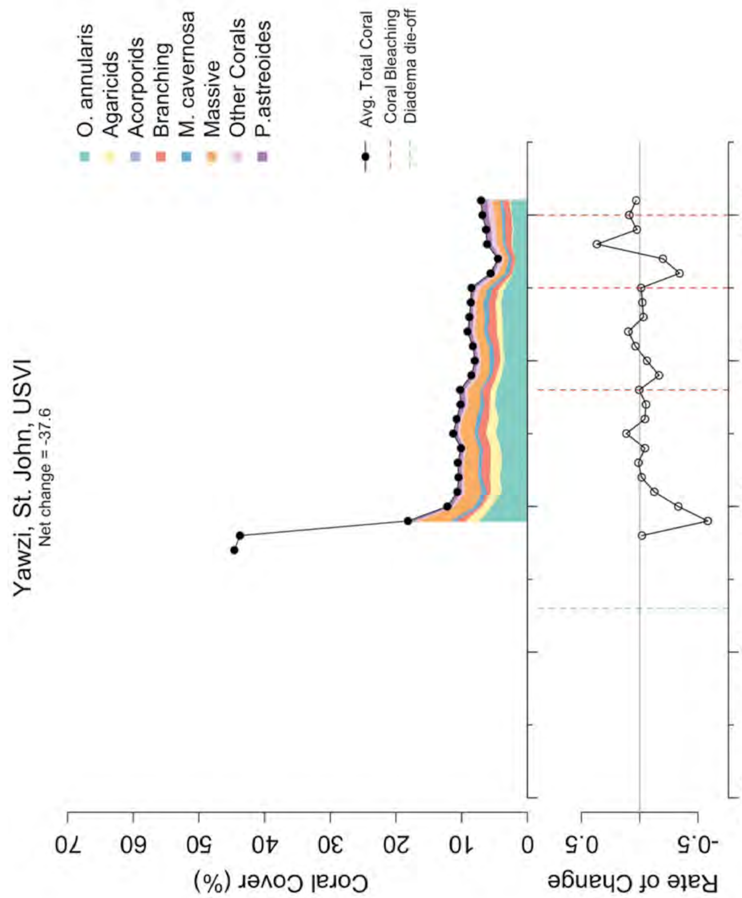

La Parguera, Puerto Rico
Net change = -27.5
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

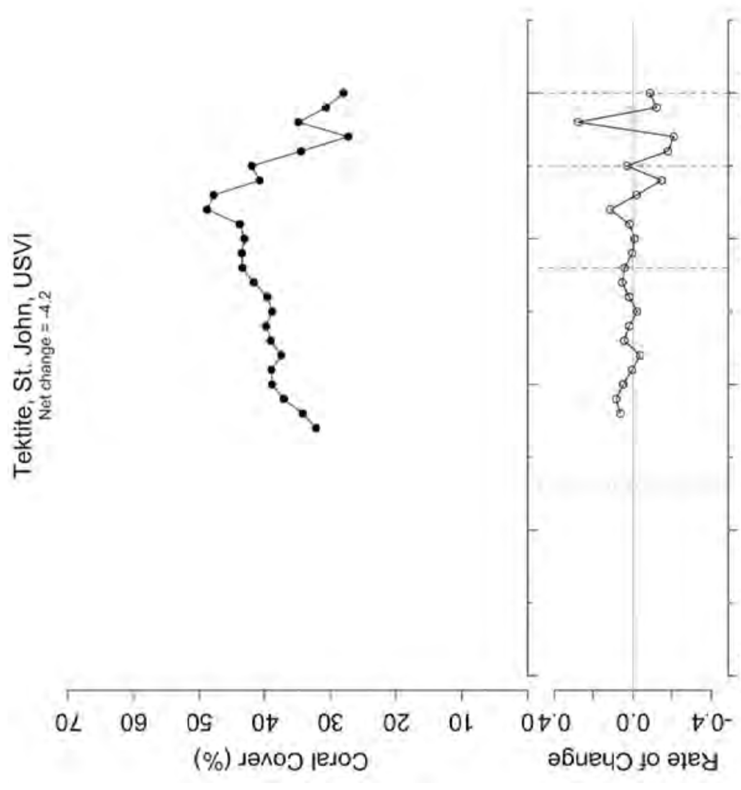

Buck Island, St. Croix, USVI
Net change = -19.8
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

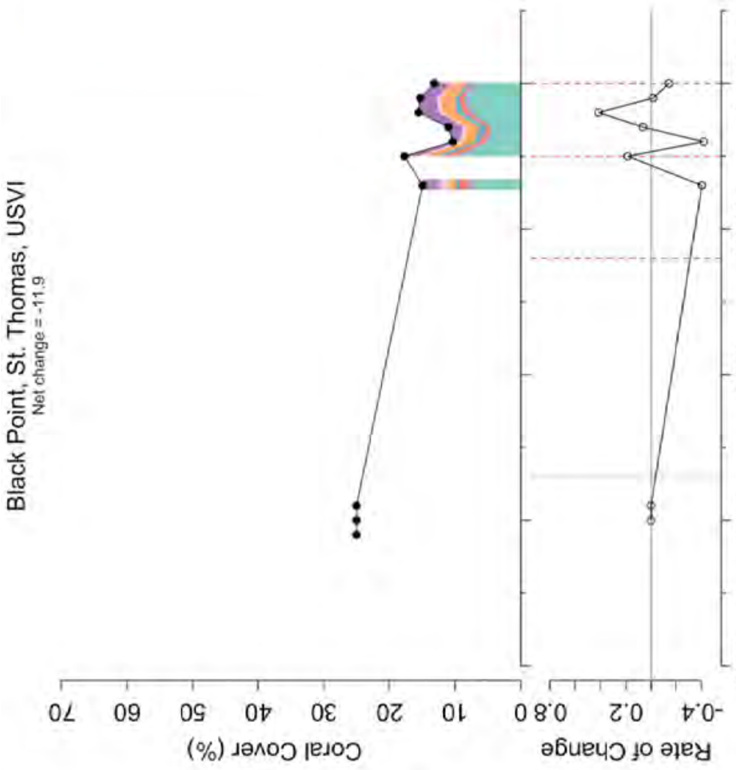

East Flower Garden Bank
Net change = -10.4
Coral Cover (%)
Rate of Change
1970
1980
1990
2000
2010

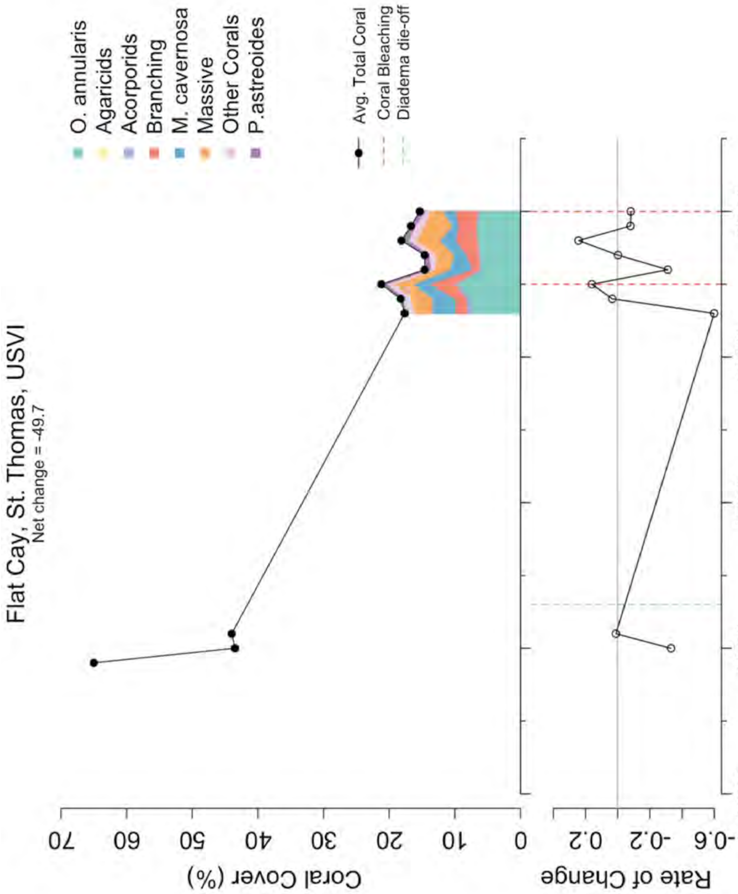

Newfound, St. John, USVI
Net change = -16.4
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
0.6
0.0
-0.6
1970
1980
1990
2000
2010

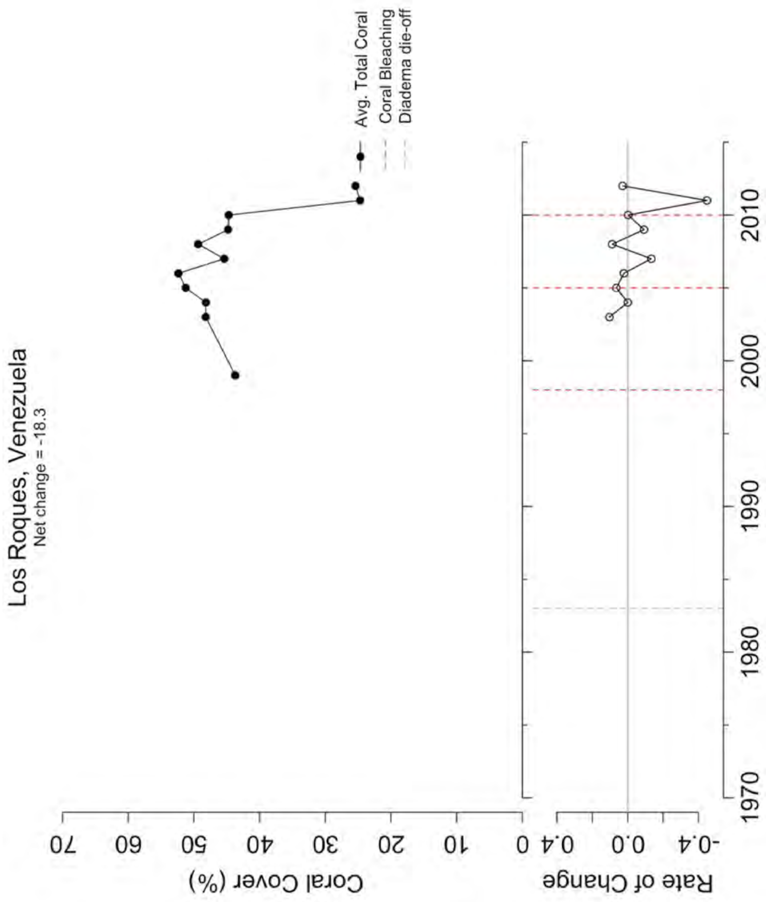

Salt River Canyon, St. Croix, USVI
Net change = -19.1
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
0.4
0.0
-0.4
1970
1980
1990
2000
2010

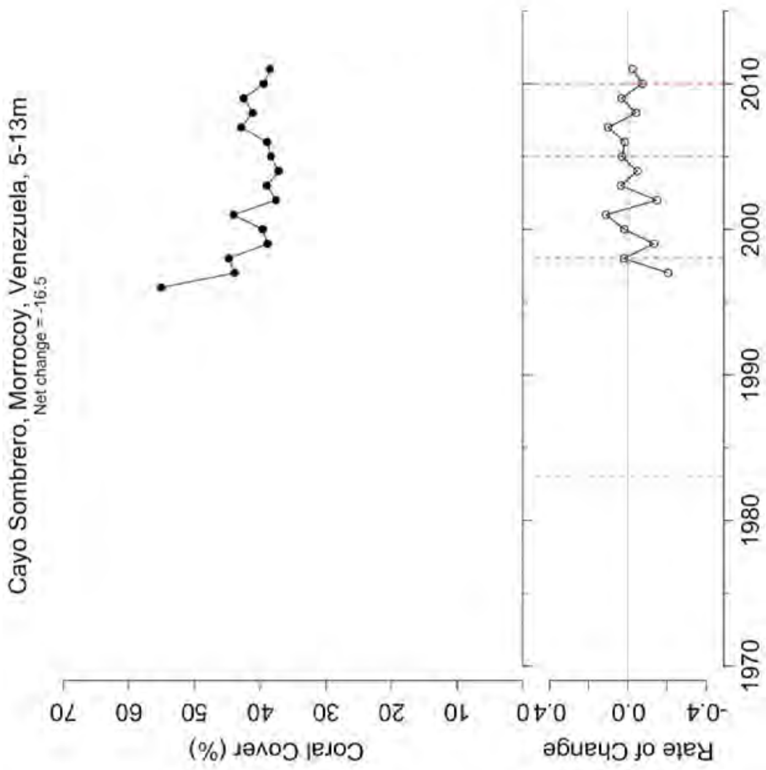

Yawzi, St. John, USVI
Net change = -37.6
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change


Tektite, St. John, USVI
Net change = -4.2
Coral Cover (%)
Rate of Change


Black Point, St. Thomas, USVI
Net change = -11.9
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
-0.4
0.2
0.8
1970
1980
1990
2000
2010


Flat Cay, St. Thomas, USVI
Net change = -49.7
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
-0.6
-0.2
0.2
1970
1980
1990
2000
2010
O. annularis
Agaricids
Acorporids
Branching
M. cavernosa
Massive
Other Corals
P.astreoides
Avg. Total Coral
Coral Bleaching
Diadema die-off


Los Roques, Venezuela
Net change = -18.3
Avg. Total Coral
Coral Bleaching
Diadema die-off
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
0.4
0.0
-0.4
1970
1980
1990
2000
2010


Cayo Sombrero, Morrocoy, Venezuela, 5-13m
Net change = -16.5
Coral Cover (%)
Rate of Change
0
10
20
30
40
50
60
70
0.4
0.0
-0.4
1970
1980
1990
2000
2010

Appendix III: Sources of qualitative data for Acropora decline

Location	Reference	Data Contributor	Start Year	End Year	Data Type
Antigua and Barbuda	Adey and Burke 1976		1974	1974	Descriptive text
Antigua and Barbuda	Hughes 1750		1750	1750	Descriptive text
Antigua and Barbuda	Wigley 1977		1976	1976	Descriptive text
Antigua and Barbuda		Creary, Marcia	2007	2008	Percent cover
Aruba	Roos 1971		1966	1966	Descriptive text
Aruba		Bak, Rolf	1986	1986	Percent cover
Bahamas	Dahl et al. 1974		1971	1971	Descriptive text
Bahamas	Milliman 1967		1964	1964	Descriptive text
Bahamas	Newell and Rigby 1957		1951	1964	Descriptive text
Bahamas	Storr 1964		1964	1964	Descriptive text
Bahamas		Bruno, John	2010	2011	Percent cover
Bahamas		Creary, Marcia	1994	2006	Percent cover
Bahamas		Lang, Judith	2011	2011	Percent cover
Bahamas		Mumby, Peter; Harborne, Alastair	2004	2007	Percent cover
Barbados	Allard 1994 Thesis		1992	1992	Percent cover
Barbados	Butsch 1939		1939	1939	Descriptive text
Barbados	Hughes 1750		1750	1750	Descriptive text
Barbados	Lewis 1960		1959	1959	Descriptive text
Barbados	Lewis 1984		1980	1980	Relative abundance
Barbados	Liddell and Ohlhorst 1988		1977	1977	Percent cover
Barbados	Ott 1975		1971	1971	Descriptive text
Barbados	Stearn et al. 1977		1976	1976	Descriptive text
Barbados	Tomascik and Sander 1987		1982	1982	Percent cover
Barbados	Williams and Polunin 2001		1997	1997	Percent cover
Barbados		Brathwaite, Angelique	1982	1982	Percent cover
Barbados		Oxenford, Hazel	1993	2006	Percent cover
Belize	Aronson et al. 1998		1980	1980	Descriptive text
Belize	Aronson et al. 2002		1975	1985	Descriptive text; Relative abundance
Belize	Dahl et al. 1974		1971	1971	Descriptive text
Belize	James and Ginsburg 1979		1978	1978	Descriptive text
Belize	Macintyre et al. 1981		1980	1980	Descriptive text
Belize	McClanahan and Muthiga 1998		1970	1996	Percent cover
Belize	Miller and Macintyre 1977		1976	1976	Descriptive text
Belize	Purdey et al 1975		1974	1974	Descriptive text
Belize	Rutzler and Macintyre 1982		1979	1979	Presence/absence
Belize	Stoddart 1962		1961	1961	Descriptive text
Belize	Williams and Polunin 2001		1998	1998	Percent cover
Belize		Bruno, John	2009	2011	Percent cover
Belize		Hardt, Marah; Paredes, Gustavo	2004	2004	Percent cover
Belize		Koltes, Karen	1994	2012	Percent cover

Location	Reference	Data Contributor	Start Year	End Year	Data Type
Belize		Mcfield, Melanie	1997	1999	Percent cover
Belize		Mcfield, Melanie	1997	1999	Percent cover
Bermuda	Dodge et al. 1982		1978	1978	Percent cover
Bermuda		Hochberg, Eric	1993	2007	Percent cover
Bermuda		Thaddeus Murdoch	2004	2012	Percent cover
Bermuda		Weil, Ernesto (CRTR Program)	2005	2009	Percent cover
Bonaire	Roos 1971		1966	1966	Descriptive text
Bonaire	Scatterday 1974		1974	1974	Descriptive text
Bonaire	Van Duyl 1985		1980	1980	Relative abundance
Bonaire	Van't Hof 1983		1982	1982	Descriptive text
Bonaire		Bak, Rolf; Nugues, Maggy	1974	2008	Percent cover
Bonaire		De Meyer, Kalli	1994	1997	Percent cover
Bonaire		Sommer, Brigitte; Scheffers, Sander; Harrison, Peter	2008	2009	Percent cover
Bonaire		Steneck, Bob	2002	2011	Percent cover
BVI	Adey and Burke 1976		1974	1974	Descriptive text
BVI	Dunne and Brown 1979		1978	1978	Relative abundance
BVI		Forrester, Graham	1992	2012	Percent cover
Cayman Islands	Rigby and Roberts 1976		1967	1967	Descriptive text
Cayman Islands	Roberts 1971		1967	1976	Descriptive text
Cayman Islands	Roberts 1974		1973	1973	Descriptive text
Cayman Islands	Roberts 1977		1977	1977	Descriptive text
Cayman Islands	Williams and Polunin 2001		1997	1997	Percent cover
Cayman Islands		Croy, McCoy; Bush; Philippe	1995	2001	Percent cover
Cayman Islands		Fenner, Douglas	1988	1988	Percent cover
Cayman Islands		Miller, Jeff	1992	1992	Percent cover
Cayman Islands		Manfrino, Carrie	1999	2011	Percent cover
Cayman Islands		Weil, Ernesto (CRTR Program)	2005	2009	Percent cover
Colombia	Erhardt & Werding 1975		1966	1966	Descriptive text
Colombia	Garzon-Ferreira and Kielman 1993		1982	1982	Percent cover
Colombia	Geister 1986		1969	1969	Descriptive text
Colombia	Kucurko 1977		1977	1977	Descriptive text
Colombia	Liddell and Ohlhorst 1988		1977	1977	Percent cover
Colombia	Milliman 1969		1966	1966	Descriptive text
Colombia		CARICOMP	1998	2006	Percent cover
Colombia		Friedlander, Alan	2000	2000	Percent cover
Colombia		Rodriguez-Ramirez, Alberto	1993	2005	Percent cover
Costa Rica	Cortes and Jimenez 1993		1981	1993	Percent cover
Costa Rica	Cortes and Risk 1983		1982	1982	Relative abundance; Presence/absence

Location	Reference	Data Contributor	Start Year	End Year	Data Type
Cuba	Kuhlmann 1971		1964	1964	Relative abundance
Cuba	Williams and Polunin 2001		1998	1998	Percent cover
Cuba		Alcolado, Pedro	1994	1997	Percent cover
Cuba		Bruno, John	2010	2011	Percent cover
Cuba		Hardt, Marah; Paredes, Gustavo	2005	2005	Percent cover
Curaçao	Bak 1976		1975	1975	Relative abundance
Curaçao	Bak and Luckhurst 1980		1973	1978	Relative abundance
Curaçao	Bries et al 2004		1971	1971	Descriptive text
Curaçao	Liddell and Ohlhorst 1988		1977	1977	Percent cover
Curaçao	Roos 1964		1961	1961	Relative abundance
Curaçao	Roos 1971		1966	1966	Descriptive text
Curaçao	Van der Horst 1927		1927	1927	Relative abundance
Curaçao	Van Duyl 1985		1980	1980	Relative abundance
Curaçao		Bak, Rolf; Nugues, Maggy	1973	2009	Percent cover
Curaçao		CARICOMP	1994	1995	Percent cover
Curaçao		Nagelkerkan, Ivan	1973	2003	Percent cover
Curaçao		Steneck, Bob	2009	2009	Percent cover
Curaçao		Vermeij, Mark	2003	2010	Percent cover
Curaçao		Weil, Ernesto (CRTR Program)	2005	2011	Percent cover
Dominica		Creary, Marcia	2007	2009	Percent cover
Dominican Republic	Geraldes and de Calventi 1978		1964	1974	Descriptive text
Dominican Republic		CARICOMP	1994	2001	Percent cover
Dry Tortugas	Agassiz 1883		1881	1881	Percent cover
Dry Tortugas	Dahl et al. 1974		1976	1976	Descriptive text
Dry Tortugas	Davis 1982		1975	1975	Descriptive text
Dry Tortugas	Jaap et al. 1989		1975	1991	Relative abundance
Dry Tortugas	LeCompte 1937		1936	1936	Descriptive text
Dry Tortugas	Porter et al. 1982		1979	1979	Relative abundance
Dry Tortugas		National Park Service, South Florida Caribbean Network	1999	2011	Percent cover
Dry Tortugas		Colella, Mike; Ruzicka, Rob	1975	1975	Percent cover
Dry Tortugas		Dustan, Phil	2005	2005	Percent cover
Dry Tortugas		Hardt, Marah; Paredes, Gustavo	1975	1976	Percent cover
Dry Tortugas		Jaap, Walter	1976	1977	Percent cover
Florida Keys	Agassiz 1880		1851	1851	Descriptive text
Florida Keys	Bright 1981		1979	1979	Descriptive text

Location	Reference	Data Contributor	Start Year	End Year	Data Type
Florida Keys	Burns 1985		1981	1981	Relative abundance
Florida Keys	Dustan 1985		1975	1975	Descriptive text
Florida Keys	Lirman & Fong 1997		1993	1994	Percent cover
Florida Keys	Shinn 1980		1979	1979	Descriptive text
Florida Keys	Shinn 1981		1979	1979	Descriptive text
Florida Keys	Wheaton 1981		1975	1975	Descriptive text
Florida Keys	Wheaton and Jaap 1988		1983	1983	Relative abundance
Florida Keys		National Park Service, South Florida Caribbean Network	2004	2011	Percent cover
Florida Keys		CARICOMP	2001	2004	Percent cover
Florida Keys		Chiappone, Mark	1999	2005	Percent cover
Florida Keys		Colella, Mike; Ruzicka, Rob	1996	2011	Percent cover
Florida Keys		Dustan, Phil	1975	1983	Percent cover
Florida Keys		Hardt, Marah; Paredes, Gustavo	2005	2005	Percent cover
Florida Keys		Pandolfi, John	1994	1996	Percent cover
Florida Keys		Weil, Ernesto	1994	1994	Percent cover
Flower Garden Banks	Bright et al. 1984		1974	1974	Percent cover
Flower Garden Banks		NOAA	2006	2011	Percent cover
Grenada	Adey and Burke 1976		1974	1974	Descriptive text
Grenada	Goodwin et al. 1976		1975	1977	Relative abundance
Grenada		Creary, Marcia; Mitchell, Jerry	2007	2009	Percent cover
Grenada		Weil, Ernesto (CRTR Program)	2005	2009	Percent cover
Guadaloupe	Adey and Burke 1976		1974	1974	Descriptive text
Guadaloupe	Battistini and Petit 1979		1979	1979	Descriptive text
Guadaloupe		Bouchon, Claude	2002	2011	Percent cover
Haiti	Beebe 1928		1927	1927	Descriptive text
Honduras		Fenner, Douglas	1987	1987	Percent cover
Jamaica	Bonem and Stanley 1977		1976	1976	Descriptive text
Jamaica	Dahl et al. 1974		1971	1971	Descriptive text
Jamaica	Goreau 1959		1955	1955	Descriptive text; Presence/absence
Jamaica	Goreau and Goreau 1973		1972	1972	Descriptive text
Jamaica	Knowlton et al. 1990		1982	1987	Percent cover
Jamaica	Liddell and Ohlhorst 1987		1977	1977	Percent cover
Jamaica	Liddell and Ohlhorst 1988		1980	1980	Percent cover
Jamaica	Rylaarsdam 1983		1976	1976	Relative abundance
Jamaica	Wapnick et al. 2004		1978	1979	Descriptive text
Jamaica	Williams and Polunin 2001		1997	1997	Percent cover
Jamaica	Woodley and Robinson 1977		1973	1973	Descriptive text
Jamaica		Dustan, Phil	1972	1973	Percent cover

Location	Reference	Data Contributor	Start Year	End Year	Data Type
Jamaica		Gayle, Peter; Charpentier, Bernadette	2011	2012	Percent cover
Jamaica		Hardt, Marah	2005	2005	Percent cover
Jamaica		Hughes, Terry	1977	1993	Percent cover
Jamaica		Loya, Yossi	1969	1969	Percent cover
Martinique	Adey et al 1977a		1976	1976	Descriptive text
Martinique	Battistini 1978		1974	1974	Presence/absence
Martinique		Bouchon, Claude	2001	2007	Percent cover
Mexico	Bonet 1967		1967	1967	Relative abundance
Mexico	Busby 1966		1959	1959	Descriptive text
Mexico	Chávez et al. 1970		1966	1969	Relative abundance
Mexico	Farrell et al. 1983		1982	1982	Presence/absence
Mexico	Freeland 1971		1968	1968	Descriptive text
Mexico	Heilprin 1890		1890	1890	Descriptive text
Mexico	Horta-Puga 2003		1999	1999	Relative abundance; Presence/absence
Mexico	Jordan and Martin 1987		1979	1979	Descriptive text
Mexico	Jordan et al. 1981		1980	1980	Relative abundance
Mexico	Kornicker and Boyd 1962		1960	1960	Descriptive text
Mexico	Kuhlmann 1975		1965	1965	Percent cover
Mexico	Liddell and Ohlhorst 1988		1985	1985	Percent cover
Mexico	Logan et al. 1969		1968	1968	Descriptive text
Mexico	Moore 1958		1955	1955	Descriptive text
Mexico	Murray 1991		1991	1991	Percent cover
Mexico	Rannefeld 1972		1971	1971	Relative abundance; Presence/absence
Mexico	Rigby and Macintyre 1966		1965	1965	Descriptive text
Mexico	Roy 2004		2000	2000	Percent cover
Mexico	Ruiz-Renteria et al. 1998		1981	1981	Presence/absence
Mexico	Secretaria de Marina 1987		1985	1986	Percent cover
Mexico	Villalobos 1971		1963	1963	Descriptive text
Mexico		Arias, Ernesto	2000	2008	Percent cover
Mexico		Fenner, Douglas	1984	1988	Percent cover
Mexico		Hardt, Marah; Paredes, Gustavo	2004	2004	Percent cover
Mexico		Reyes Bonilla, Hector	2005	2011	Percent cover
Mexico		Rodriguez-Martinez, Rosa	1993	2005	Percent cover
Nicaragua		CARICOMP	1993	1998	Percent cover
Panama	Dahl et al. 1974		1971	1971	Descriptive text
Panama	Ogden and Ogden 1996		1971	1971	Relative abundance
Panama	Robertson and Glynn 1977		1977	1977	Descriptive text

Location	Reference	Data Contributor	Start Year	End Year	Data Type
Panama		Cramer, Katie	1959	2008	Relative abundance; Percent cover
Panama		Guzman, Hector	1985	2011	Percent cover
Panama		Weil, Ernesto (CRTR Program)	2005	2006	Percent cover
Puerto Rico	Acevedo et al. 1989		1989	1989	Percent cover
Puerto Rico	Almy and Carrion-Torres 1963		1961	1961	Descriptive text; Relative abundance
Puerto Rico	Garrison et al. 2005 USGS Report		1991	1998	Percent cover
Puerto Rico	Macintryre at el. 1983		1978	1978	Descriptive text
Puerto Rico	Morelock et al. 1977		1976	1976	Descriptive text
Puerto Rico	Pressick 1970		1969	1969	Descriptive text
Puerto Rico	Szmant-Froelich 1972		1971	1971	Descriptive text
Puerto Rico		CARICOMP	1994	2012	Percent cover
Puerto Rico		NOAA	2007	2011	Percent cover
Puerto Rico		Weil, Ernesto	2003	2007	Percent cover
Saba		Buchan, Kenny	1993	2003	Percent cover
Saba Bank	Van der Land 1977		1972	1973	Presence/absence
SE Florida	Burns 1985		1981	1981	Relative abundance
SE Florida	Goldberg 1973		1972	1972	Relative abundance
SE Florida	Porter and Meier 1992		1989	1989	Percent cover
SE Florida		Colella, Mike; Ruzicka, Rob	2003	2011	Percent cover
SE Florida		Weil, Ernesto	1994	1994	Percent cover
St Kitts and Nevis		Creary, Marcia	2007	2009	Percent cover
St Kitts and Nevis		Lang, Judith	2011	2011	Percent cover
St Lucia	Roberts 1972		1971	1971	Descriptive text
St Lucia		Creary, Marcia	2007	2009	Percent cover
St Martin	Adey and Burke 1976		1974	1974	Descriptive text
St Vincent and the Grenadines	Adams 1968		1965	1965	Presence/absence
St Vincent and the Grenadines	Goodwin et al. 1976		1976	1976	Percent cover
St Vincent and the Grenadines	Lewis 1975		1972	1972	Descriptive text
St Vincent and the Grenadines		Creary, Marcia	2007	2009	Percent cover
St Barthelemy		Bouchon, Claude	2002	2011	Percent cover
Trinidad and Tobago	Kenny 1988		1972	1972	Descriptive text
Trinidad and Tobago		Alemu, Jahson	1994	2012	Percent cover
Turks and Caicos		CARICOMP	1999	1999	Percent cover
US Virgin Islands	Adey et al. 1977b		1977	1977	Descriptive text
US Virgin Islands	Adey et al. 1977c		1977	1977	Descriptive text; Relative abundance
US Virgin Islands	Dahl et al. 1974		1971	1971	Descriptive text

Location	Reference	Data Contributor	Start Year	End Year	Data Type
US Virgin Islands	Gladfelter et al. 1977 NPS Report		1976	1976	Percent cover
US Virgin Islands	Hubbard et al. 1994		1976	1976	Relative abundance
US Virgin Islands	Hubbard et al.2005		1979	1979	Descriptive text
US Virgin Islands	Macintyre and Adey 1990		1977	1977	Descriptive text
US Virgin Islands	Rogers et al. 1983		1979	1981	Relative abundance
US Virgin Islands		National Park Service, South Florida Caribbean Network	1999	2011	Percent cover
US Virgin Islands		Edmunds, Peter	1987	2010	Percent cover
US Virgin Islands		Lundgren, Ian; Zandy Hilis Starr	1989	2005	Percent cover
US Virgin Islands		Miller, Jeff	1989	2002	Percent cover
US Virgin Islands		NOAA	2001	2011	Percent cover
US Virgin Islands		Rogers, Caroline	1978	1981	Percent cover
US Virgin Islands		Smith, Tyler; Nemeth, Rick	2001	2010	Percent cover
US Virgin Islands		Steneck, Bob	1982	1988	Percent cover
Venezuela	Antonius 1980		1968	1968	Descriptive text
Venezuela	Weiss et al. 1978		1972	1972	Descriptive text
Venezuela		Bastidas, Carolina; Croquer, Aldo	2003	2008	Percent cover

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Appendix IV: Sources of data for *Diadema* abundance before 1984 in Table 8

Barbados Leeward

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Belize Central Barrier

Lewis SM, Wainwright PC (1985) Herbivore abundance and grazing intensity on a Caribbean coral reef. Journal of Experimental Marine Biology and Ecology 87: 215-228.

Bermuda

Bauer JC (1980) Observations on geographical variations in population density of the echinoid *Diadema antillarum* within the western North Atlantic. Bulletin of Marine Science 30: 509-515.

Bonaire Leeward

Stokes MD, Leichter JJ, Genovese SJ (2010) Long-term declines in coral cover at Bonaire, Netherlands Antilles. Atoll Research Bulletin 582: 1-21.

Costa Rica Cahuita

Alvarado JJ, Cortés J, Salas E (2004) Population densities of *Diadema antillarum* Philippi at Cahuita National Park (1977-2003), Costa Rica. Caribbean Journal of Science 40: 257-259.

Curacao Southwest

Carpay MJE (1985) De rol van *Diadema antillarum* Philippi in de bio-erosieve rifprocessen op Curaçao. MSc Thesis. Amsterdam: University of Amsterdam. 67 p.

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Bak RP, Eys G (1975) Predation of the sea urchin *Diadema antillarum* Philippi on living coral. Oecologia 20: 111-115.

Florida Upper Keys

Bauer JC (1980) Observations on geographical variations in population density of the echinoid *Diadema antillarum* within the western North Atlantic. Bulletin of Marine Science 30: 509-515.

Jamaica Montego Bay

Hughes TP, Keller BD, Jackson JBC, Boyle MJ (1985) Mass Mortality of the Echinoid *Diadema antillarum* Philippi in Jamaica. Bulletin of Marine Science 36: 377-384.

Jamaica North Central

Hughes TP, Keller BD, Jackson JBC, Boyle MJ (1985) Mass Mortality of the Echinoid *Diadema antillarum* Philippi in Jamaica. Bulletin of Marine Science 36: 377-384.

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Hughes TP (1989) Community structure and diversity of coral reefs - the role of history. Ecology 70: 275-279.

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Steneck RS (1994) Is herbivore loss more damaging to reefs than hurricanes? Case studies from two Caribbean reef systems (1978-1988). In: Ginsburg RN, editor. Proceedings of the Colloquium on Global Aspects of Coral Reefs: Health, Hazards and History. Florida: University of Miami. pp. C32-C37.

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Jamaica North East

Hughes TP (1994) Catastrophes, phase-shifts, and large-scale degradation of a Caribbean coral reef. Science 265: 1547-1551.

Jamaica Port Royal Cays

Hughes TP (1994) Catastrophes, phase-shifts, and large-scale degradation of a Caribbean coral reef. Science 265: 1547-1551.

Jamaica West

Hughes TP (1994) Catastrophes, phase-shifts, and large-scale degradation of a Caribbean coral reef. Science 265: 1547-1551.

Panama San Blas

Shulman MJ, Robertson DR (1996) Changes in the coral reefs of San Blas, Caribbean Panama: 1983 to 1990. Coral Reefs 15: 231-236.

Lessios HA (2005) *Diadema antillarum* populations in Panama twenty years following mass mortality. Coral Reefs 24: 125-127.

USVI St Croix

Carpenter RC (1990) Mass mortality of *Diadema antillarum* I. Long-term effects on sea urchin population-dynamics and coral reef algal communities. Marine Biology 104: 67-77.

Rogers CS, Carl Fitz III H, Gilnack M, Beets J, Hardin J (1984) Scleractinian coral recruitment patterns at Salt River Submarine Canyon, St. Croix, U.S. Virgin Islands. Coral Reefs 3: 69-76.

Steneck RS (1994) Is herbivore loss more damaging to reefs than hurricanes? Case studies from two Caribbean reef systems (1978-1988). In: Ginsburg RN, editor. Proceedings of the Colloquium on Global Aspects of Coral Reefs: Health, Hazards and History. Florida: University of Miami. pp. C32-C37.

USVI St John

Bauer JC (1980) Observations on geographical variations in population density of the echinoid *Diadema antillarum* within the western North Atlantic. Bulletin of Marine Science 30: 509-515.

USVI St Thomas

Hay ME, Taylor PR (1985) Competition between herbivourous fishes and urchins on Caribbean reefs. Oecologia 65: 591-598.

Appendix V: Summary of information on fishing activities and fish catch for reef locations in Table 8

Barbados

Fisheries in Barbados are varied and include both inshore and offshore fishing. The inshore fishery becomes very important during the offseason for pelagics consists mainly of fishing by pots, but handlines, nets and spearguns have been common. In 1947 gillnets were introduced and widely adopted in the 1950s (Hess 1966). The total catch of inshore fishes has been relatively stable since 1960 at about 500 tons despite a large increase in effort starting in the 1970s. Historical reconstruction of Barbados fisheries statistics found that the number of boats exploiting the inshore fishery increased by 176% between 1979 and 2000 and a dramatic 73% decline in catch rates between 1966 and 2000 (Mohammad *et al.* 2003). Frydl & Stern (1978) surveyed parrotfishes in 1975-76 and noted a low biomass on the fringing reef compared to the back reef which they attributed to trap fishing by local fishermen, spearfishing and/or large densities of *Diadema*, which outcompeted the parrotfishes for food. The authors spent seven months in the field in 1975 and 1976 at several locations and did not see a single individual of the larger parrotfish species (*Scarus coeruleus*, *S. guacamaia*, *S. coelestinus*) and noted that *S. rubripinne*, which can obtain sizes of up to 45cm, was rarely encountered. Total parrotfish biomass around the reefs surveys in 1975-76 was extremely low (2-5 g m^{-2}).

Belize

Belizean fish stocks were considered lightly to moderately fished by Koslow (1994) based on observations of continually available target species such as snappers and groupers and a small, dispersed human population. These observations were followed by a formal survey of the fishery where relatively low catch per unit area was measured and catch was comprised entirely of snappers and groupers (Koslow et al. 1994). This study showed that the dominant gear type was hook and line and the catch consisted almost exclusively of predators such as snapper and groupers, largely for the export markets. The reliance on line fishing and the limited amount of trap fishing prior to the 1990s suggests minimal extraction of herbivorous fishes at that time. Surplus production models for 1991 indicate the fishery operating at 10% of maximum sustainable yield for prime commercial species (snappers and groups). Additionally, several earlier studies of herbivore communities in the early 1980s found herbivorous fishes to constitute an important component of the grazing capacity on the reef (Hay 1984, Lewis and Wainwright1985). Increasing pressure on Belize's fisheries was noticed much later than the rest of the Caribbean, with lobster and conch stocks declining in the 1990s. This increasing pressure on fish stocks led to a shift in targeted species after 2002 to parrotfishes (Mumby 2012). The sudden rise in catch of herbivorous species led to concerns that allowed for a ban on herbivorous fish harvesting in Belize in 2009.

Bermuda

Stevenson and Marshall 1974 noted that overall fishing effort was low in Bermuda and the 1960s and 1970s and stated that in 1956 only serranids were retained for market . Bermuda's fishery consists of two distinct sectors, an offshore- pelagic fishery and a near shore-reef fishery. The primary targets of the reef fishery are snapper and grouper, with jacks constituting most of the remaining portion. Fish production was consistent at about 5,000 tons, and most fish were caught locally until the 1980s (Butler *et al.* 1993). In the early 1980s catch began to increase and by 1990 the composition of the catch changed substantially. From 1951 to 1991 the percent of catch comprised of reef fishes, including parrotfishes and surgeonfishes, changed from 1 to 31 percent. Parrotfishes were not taken for food until 1977 and by 1986 comprised 36 percent of reef fish catch (Butler *et al.* 1993). In response to the concerns for declining fish stocks, in 1990 the Bermuda government prohibited the use of all nets and pots and strict regulations on trap fishing began in 1992. A survey of the landings data between 1970 and 1990 found a prominent increase in the landings of parrotfishes with a drop in landings following the trap regulations with a 81% decline from 1980 to 1990 (Luckhurst and Ward 1991).

Bonaire
Bonaire Island Government declared the entire coast around Bonaire and Klien Bonaire a marine park in1979. The fisher population is relatively small and consists of mostly recreational and subsistence fishers. Historically, fisherman did not target parrotfish, yet their population density has declined since 1999 (Steneck & Arnold 2011). In 2011 the park established a ban of all parrotfish take.

Cahuita, Costa Rica
Cahuita National Park was first declared a marine monument in 1970 and was later ratified to be a national park in 1978. The park covers 600 ha of reef that is the only well-developed reef on the Caribbean coast of Costa Rica. The park is a large tourist destination and is also close to the towns of Cahuita and Hone Creek. Residents from these towns have historically fished the reefs of Cahuita (Cortes, thesis).

Curacao
Curacao's main fishery is for pelagics, although reef fishes are caught in the artisanal fishery with a catch of 90-180 tons/year estimated in 1988 (Woodley 1997). In 1976 spearfishing and coral collection were prohibited but enforcement has never been realized (Woodley 1997, Buckner & Buckner 2003). In 1998 underwater surveys conducted island-wide found parrotfishes to be present at all sites and in some cases were the dominant species numerically (Buckner & Buckner 2003).

Florida Keys
Extensive commercial and recreational fisheries have existed in Florida Keys for hundreds of years (McClenachan 2009). Temporal studies of fisheries have revealed strong decreases in catch per unit effort of targeted species (McClenachan 2009, Ault 1998). But, the Florida Keys is one of the few places in the Caribbean where herbivorous fishes are not targeted in the catch (Harper 2000). A recent study on the impact of grazing by herbivorous fishes found that herbivory levels were sufficient to maintain low macroalgal cover on the offshore reefs in the upper Florida Keys (Paddock 2006).

Jamaica
Jamaica's fisheries have been heavily exploited for hundreds of years with a decline in reef fish populations by 1940, with future declines due to shifts in gear technology and government subsidies throughout the 20th century (Hardt 2009). Snappers, groupers, and large parrotfishes (Scaridae) were noted to be abundant off Jamaica in 1800s (Gosse, 1851) but have virtually disappeared from most reef areas by the early 1990s (Koslow et al. 1994). By the 1950s the dominant fish caught in traps included parrotfishes and surgeonfishes as large groupers, snappers, and grunts became extremely rare (Munro 1971, 2003 cited in Hardt 2009). Remote Pedro Bank was lightly exploited in the late 1960s but increasing fishing pressure over the next 15 years, presumably due to overexploitation of more accessible locations, led to a 75% decline in catch rates around the banks (Koslow et al. 1989). Declines in a wide spectrum of fishes (e.g., grunts, groupers, surgeonfishes, and triggerfishes), as well as "virtual elimination of larger species such as large parrotfishes, large groupers, and snappers" were noted throughout Jamaican waters by the 1980s (Koslow et al. 1989). In recent times, the overall density and biomass of all fishes and particularly herbivorous fishes in Jamaica is considerably lower than all other places in the Caribbean (Newman 2006, Paddock 2008, this study). A formal survey of Jamaica's fisheries in 1990 found that the landings were comprised of many low-value species, including small parrotfishes, surgeonfishes, porgies, wrasses, goatfishes, and soldierfishes (Koslow 1994).

U.S. Virgin Islands
The effects of intensive fishing pressure have been felt throughout the USVI with dramatic declines in fish populations over the past 30-40 years (Beets 1997, Beets and Friedlander 1999, Rogers and Beets 2001, Beets and Rogers 2002). As far back at the late 1950s, Randall (1963) noted that the limited fringing reef area around the USVI received nearly all of the fishing effort, and as a consequence the effects of overfishing were evident. Large predatory fishes such as groupers and snappers are now far less abundant, the relative abundance of herbivorous fishes has increased, individuals of many fish species are smaller, and some spawning aggregations have been decimated (Beets and Friedlander 1992, 1999, Beets 1997, Beets and Rogers 2002). The reef

fish assemblage in the US Virgin Islands has suffered the loss of large predators and a virtual absence of large parrotfishes resulting from years, if not decades, of overfishing (Friedlander and Beets 2008). This release from top-down control has likely increased the importance of bottom-up processes, such as disturbance events and habitat loss (Rogers et al. 2008).

San Blas, Panama

Reefs of San Blas, Panama have historically been fished for subsistence by the local population and until recently were considered in relatively good condition (ref). A study comparing the health of the reef in 1983 and 1996 found changes in the composition of the benthos, but did not attribute this change to herbivorous fishes, which were considered to be in relatively high numbers (Shulman and Robertson 1996).

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LIST OF ACRONYMS

AGGRA	Atlantic and Gulf Rapid Reef Assessment
AIC	Akaike Information Criterion
BBD	Black Band Disease
BIC	Bayesian Information Criterion
BVI	British Virgin Islands
CARICOMP	Caribbean Coastal Marine Productivity Program
CREMP	Coral Reef Evaluation and Monitoring Project
CDR	NOAA Data Climate Record
CRW	NOAA Coral Reef Watch
DHW	Degree Heating Week
FWC	Florida Fish and Wildlife
GCFI	Gulf and Caribbean Fisheries Institute
GCRMN	Global Coral Reef Monitoring Network
GHRSST	Group for High Resolution Sea Surface Temperature
GIS	Geographic Information System
GLMM	Generalized linear mixed effect model
I&M of NPS/SFCN	Inventory and Monitoring Program, National Park Service/South Florida Caribbean Network
ICRI	International Coral Reef Initiative
ICRS	International Coral Reef Symposium
IUCN	International Union for the Conservation of Nature
LOF	Living Oceans Foundation
MACC	Mainstreaming Adaptation to Climate Change
MDS	Multidimensional Scaling
NOAA	National Oceanographic and Atmospheric Administration
PCA	Principal Component Analysis
PC 1, 2, 3	Principle Component Axis 1, 2, and 3
R	Software Program R, Version 2.15 (R Development Core Team 2011)
SPAW-RAC	CEP Regional Activity Centre for Specially Protected Areas and Wildlife
SST	Sea Surface Temperature
TNC	The Nature Conservancy
UNEP	United Nations Environment Programme
USVI	United States Virgin Islands
WBD	White Band Disease
WPD	White Pox Disease
YBD	Yellow Band Disease

PART II: REPORTS FOR INDIVIDUAL COUNTRIES AND TERRITORIES

1. INTRODUCTION

Part II provides more detailed coverage of the status and trends of coral reefs in individual countries and territories with references to all the compiled sources of data.

Each report contains 6 sections in a standardized format.

1. **Geographic Information:** Provided by the World Resource Institute (WRI) including length of coastline, land area, maritime area, local population size, reef area and number of MPAs. Updated statistics are included for certain countries based on research contributions and information was omitted for countries where total population wasn't representative of the population near coral reefs such as large continental countries.

2. **Map of individual surveys**: The maps indicate the location of survey data for corals and macroalgae only, based on geographical coordinates of individual studies. Points on the maps are numbered to correspond to a table listing data sources. Points are distributed at or near the actual geographic location with one label per buffered area. This labeling system is employed to avoid overlap and ease readability. Thus, the number of labels per dataset does not reflect the number of samples in that dataset, only the rough position of the surveys in a given area. Some map codes in the table are not always present in the map due to missing coordinates or other crucial metadata. Location labels correspond to those defined by this study.

3. **Data sources**: Summary tables listing all data sources with locations, survey dates, and numbers of years of data for percent cover of corals and macroalgae, density of *Diadema*, and numbers or biomass of fishes. All data provided to GCRMN staff with information on these metrics are included in the table.

4. **Status and trends**: This includes graphs of average changes in coral and macroalgal cover, abundance of *Diadema*, and biomass of parrotfishes and groupers based on quantitative surveys. Results presented are for depths 0-20.9 meters and are averages by dataset and location. LOESS smoothers were applied to the data, which is a weighted linear regression that incorporates adjacent values into the fit.

   The detail and number of the graphs depends on the geographic extent of each country and the amount of data available. Each data point is displayed with a number or letter corresponding to the map code in the table of data sources. Data for all four variables were not always available or usable from every country, in which case no graph is presented. The Caribbean regional average of each variable is drawn as a light gray line through every figure. Regional trends in coral and macroalgal cover exclude data collected from random stratified sampling programs that surveyed non-reef habitats. This was not an issue for *Diadema* abundance and fish biomass,

5. **Timeline**: This is a selective list of local events affecting coral reefs. This information was gathered from individuals working in each country or territory.

6. **References**: These include (1) basic references on local coral reefs and fisheries and (2) published data sources in the GCRMN database. For a more detailed bibliography of the early literature see Wells (1998).

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COUNTRIES, STATES, AND TERRITORIES

Antigua & Barbuda
Bahamas
Barbados
Belize
Bermuda
Bonaire
British Virgin Islands
Cayman Islands
Colombia
Costa Rica
Cuba
Curaçao
Dominica
Dominican Rep.
Flower Garden Banks
Florida Keys
French Antilles
Grenada
Guatemala
Honduras
Jamaica
Mexico
Navassa Island
Nicaragua
Panama
Puerto Rico
Saba, St. Maarten, and St. Eustaius
St. Kitts & Nevis
St. Lucia
St. Vincent & the Grenadines
Trinidad & Tobago
Turks & Caicos
US Virgin Islands
Venezuela

ANTIGUA & BARBUDA

Coauthors: Marilyn Brandt, MACC, AGRRA and Reef Check

Geographic Information

Coastal Length:	285 km
Land Area:	463 km^2
Maritime Area:	109,845 km^2
Population:	69,886
Reef Area:	116 km^2
Number of hurricanes in the past 20 years:	6

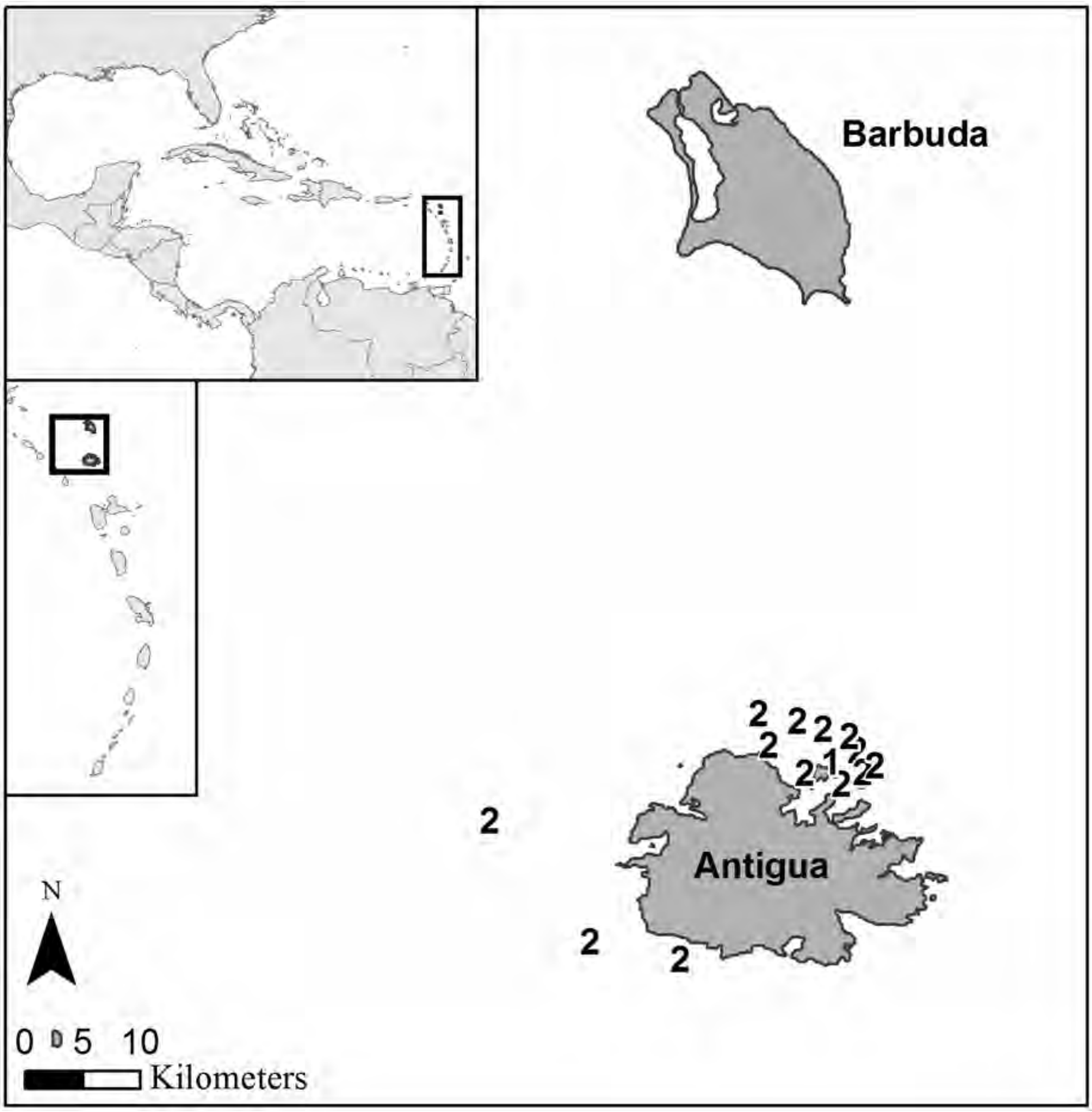

Fig. 1.1 Map of Antigua & Barbuda, codes represent studies listed in Table 1.1. Missing map code(s) due to unavailable coordinates.

Table 1.1 Collected data sources from Antigua & Barbuda, codes represent individual studies. Refer to Fig. 1.1 for locations; * denotes original data; full references found in published data sources.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	MACC*[1]	2007-2008	2	X		X	
2	Brandt, Marilyn/ AGRRA*[2]	2005	1	X	X		X
3	Bauer 1980[3]	1979	1		X		
4,5	Reef Check*	2003-2004	2		X		

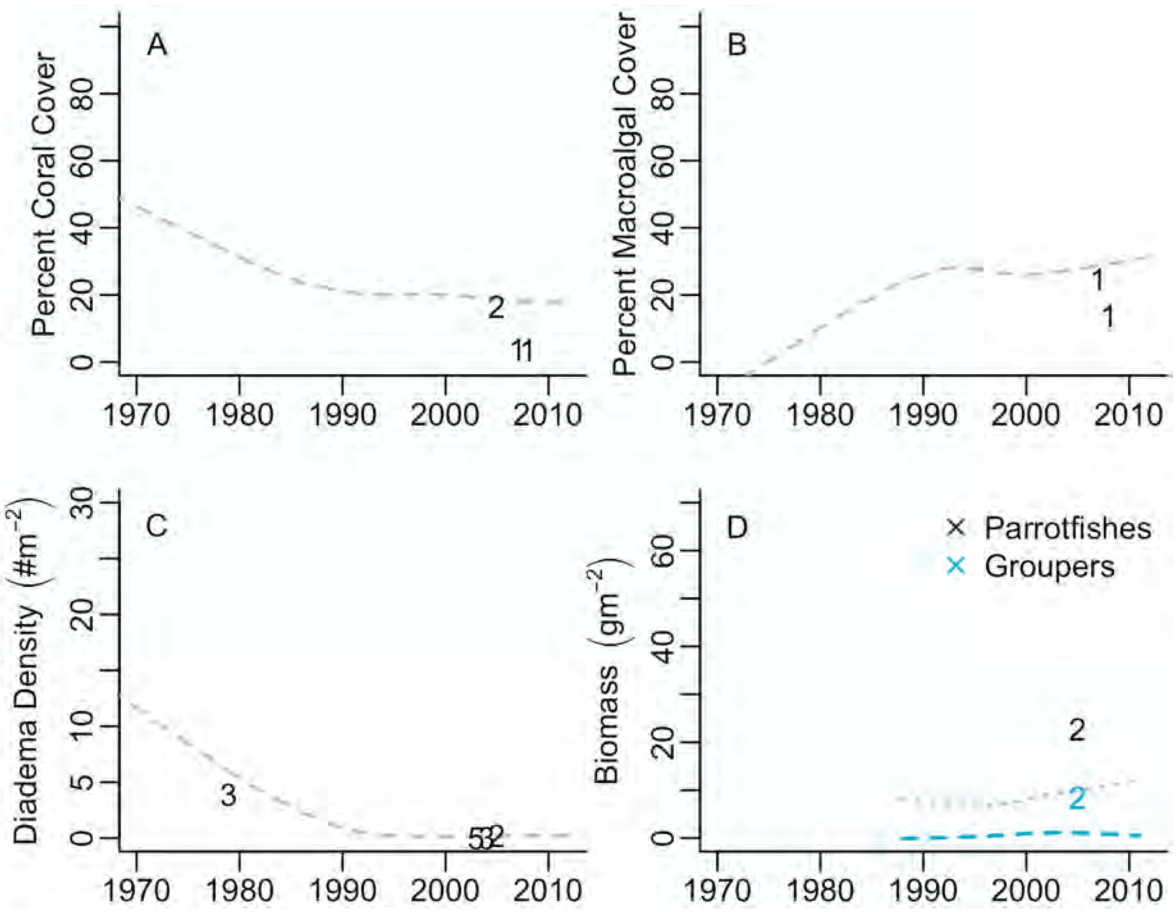

Fig. 1.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C), and biomass of parrotfishes and groupers (D) in Antigua & Barbuda. Dotted line represents the average of Caribbean data collected for this report (codes as in Table 1.1 and Figure 1.1).

General Literature

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BAHAMAS

Coauthors: Dan Brumbaugh, John Bruno, Mark Chiappone, Craig Dahlgren, Phil Dustan, Brooke Gintert, Alastair Harborne, Mark Hixon, Allison King, Lindy Knowles, Patricia Kramer, Philip Kramer, Judy Lang, Casuarina McKinney-Lambert, Peter Mumby, Ivan Nagelkerken, Eric Pante, Kathleen Sullivan Sealey, Chris Stallings, AGRRA, CARICOMP, Khaled bin Sultan Living Oceans Foundation and Reef Check

Geographic Information	
Coastal Length:	11,145 km
Land Area:	13,370 km^2
Maritime Area:	622,273 km^2
Population:	304,107
Reef Area:	4,081 km^2
Number of hurricanes in the past 20 years:	13

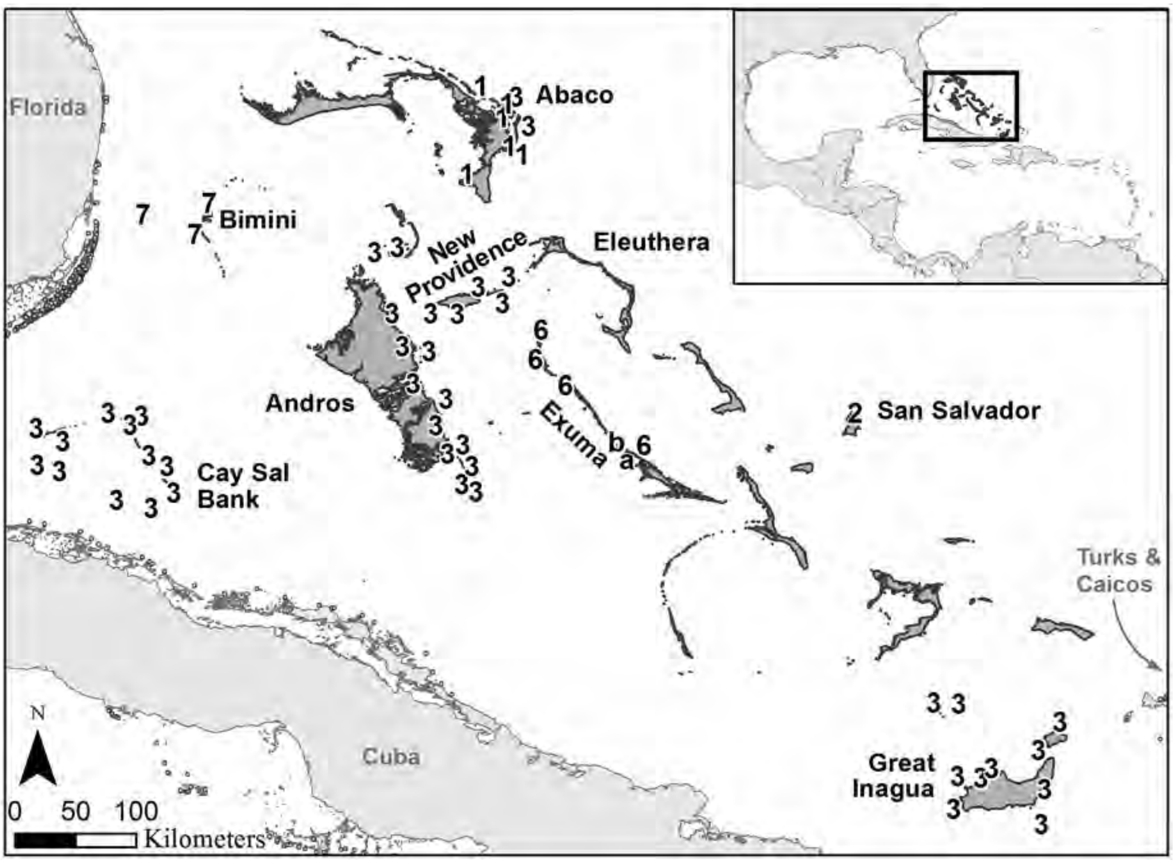

Fig. 2.1 Map of Bahamas, codes represent studies listed in Table 2.1. Missing map code(s) due to unavailable coordinates.

Table 2.1 Collected data sources from Bahamas, codes represent individual studies. Refer to Fig. 2.1 for locations; * denotes original data; full references found in published data sources.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Bruno, John*	Abaco	2010-2011	2	X	X	X	
2	CARICOMP*[1]	Fernandez Bay	1994-1998, 2001, 2003-2006	10	X	X	X	
3	AGRRA/ LOF*[2,3,4,5,12,13,14,15]	Abaco; Andros; Cay Sal Bank; Inaguas; New Providence	1998-1999, 2008, 2011	4	X	X	X	X
6	Harborne, Alastair; Mumby, Peter*[6,7,8]	Exuma Cays Land and Sea Park (ECLSP)	2004, 2007	2	X	X		X
7	Nagelkerkan, Ivan*	Bimini	2006	1	X			
8	Hay 1984[9]	Eleuthera	1981	1		X		
9	Reef Check*	Andros; Paradise Island	1999-2000, 2002-2007	8		X		
a	Hixon, Mark; Stallings, Chris*	Southern Exumas	1993-2005	13				X
b	Dustan, Phil; King, Allison; Pante, Eric*[10]	Iguana Cay, Exuma	1991, 2004	2	X			
c	Bauer 1980[11]		1978	1			X	
d	AUTEC/Patricia Kramer	Andros	2002-2011	7	X			

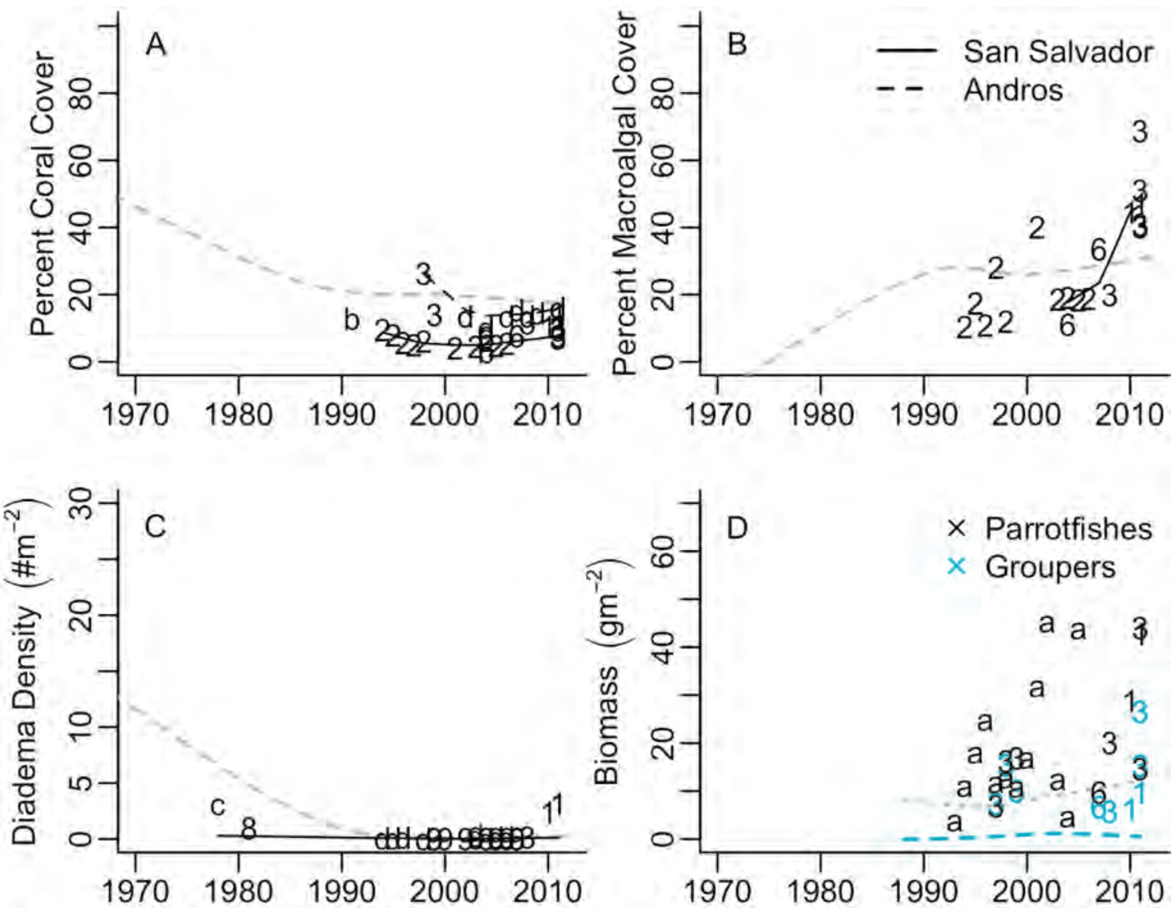

Fig. 2.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C), and biomass of parrotfishes and groupers (D) in Bahamas. Dotted line represents the average of Caribbean data collected for this report (codes as in Table 2.1 and Figure 2.1).

Timeline

950-1500:	Fishing by dugout canoe using spears, nets, hook and line and traps (Blick 2007)
1890-1930s:	First coral reef studies of Andros Island (Agassiz 1895; Ray 1998)
1950:	Development of mass tourism on New Providence (Cleare 2007; Palmer 1994)
1958-1959:	Exuma Cays Land and Sea Park and Bahamas National Trust established (Ray 1998)
1968-1969:	Minor coral bleaching on Andros Island, causing ~8% coral mortality, significant coral disease observed at Andros Island (Gintert 2011)
1970s:	Fleshy macroalgae common on some Andros Island fore reefs (Gintert 2011)
1972:	Pelican Cays Land and Sea Park established
1983:	Bleaching event at Gingerbread Grounds, Grand Bahama Bank (Wells 1988)
1984:	Mass die-off of *Diadema antillarum* (Lessios et al. 1984)
1985:	White band disease recorded (Rogers 1985)
1986:	Exuma Cays Land and Sea Park made fully no-take
1987:	Minor coral bleaching on Andros Island, Lee Stocking Island
1990:	Minor bleaching event
1991:	Lee Stocking Island patch reef coral cover 13% (Pante et al. 2008)
1993:	Minor bleaching event, Lee Stocking Island (Anthony et al. 1997)
1994:	CARICOMP monitoring at San Salvador: coral cover 9.6%, algal cover 17.5% (Wilkinson 2000)
1995:	Minor bleaching event (Linton et al. 2002); In the Exuma Cays, coral cover on shallow (<10m) channel reefs was 2-44%, algal cover was 22-80%; on shallow fringing reefs, coral cover was 8-37% and algal cover was 37-77% (Chiappone et al. 1997)
1997-1998:	Massive, widespread coral bleaching: On Andros reefs, 10-80% of corals bleached (Kramer et al. 2003a); More than 60% shallow water (<15m) corals bleached at Walker's Cay, Chub Cay, New Providence Island, Sweetings Cay, Egg Island, San Salvador, Little San Salvador, and Little Inagua, and other reports up to 80% of bleached corals between 15-20m depth (Wilkinson 2000); Abaco (Feingold et al. 2003) and Samana Cay reefs were not significantly affected; CARICOMP monitoring at San Salvador: coral cover 6.1%, algal cover 39.3% (Wilkinson 2000)
1998:	First short-term fishing closure (lasting 1-2 weeks) at High Cay, Andros focused on conserving a Nassau grouper spawning aggregation. In subsequent years, similar short-term closures were continued at High Cay and other spawning sites at Long Island.
2000-2002:	Andros Island live coral declined nearly 50% on some reefs after 1998 bleaching/disease event (Kramer & Kramer 2003); First major decline in Andros Island coral cover since 1970s (Gintert 2011)
2000:	Commitment to establish five no-take fisheries reserves (Dahlgren 2004)
2002:	Expansion of the National Park System, including Walker's Cay National Park, North and South Marine Parks on Andros, West Side National Park, and Little Inagua National Park
2004:	The start of nationwide annual partial to near complete seasonal bans on fishing for Nassau groupers during winter spawning aggregation months; invasive Pacific red lionfish (*Pterois volitans*) established on Bahamian reefs (Darling et al. 2011); Hurricanes Frances and Jeanne affected reefs in Exuma Cays (Bahamas Biocomplexity Project 2003) and severely damaged Abaco reefs; Lee Stocking Island patch reef coral cover 3%- decline appears to be due to 1998 bleaching event, bioerosion, and hurricanes (Pante et al. 2007)
2005:	Minor coral bleaching, but not as severe as reported in eastern Caribbean mass bleaching event (Eakin et al. 2010), low coral mortality on Andros reefs and other reefs had only 17% of corals bleached, with little mortality (Kramer & Kramer 2007)
2007:	Minor coral bleaching event, low coral mortality on Andros reefs (Kramer & Kramer 2008)
2008:	The Bahamas launched the Caribbean Challenge with other Caribbean countries, committing to protecting 20% of nearshore environment by 2020; New Providence and Rose Island - live coral cover varied 3.5% on hard bottom and 14.5% on fore reefs (median values); macroalgal cover varied from 11.5% in reef crests to 24.5% in patch reefs; and total fish biomass varied from ~2500 g/100 m^2 on hard bottom to ~8700 g/100 m^2 in the fore reefs (Lang et al. 2008)
2009:	Continued recovery observed on Andros reefs after 1998 bleaching event with increases in coral cover (2-27%), Diadema abundance, and fish biomass (including commercially significant fish and herbivorous fish); however, lionfish biomass increased since 2007 (Kramer & Kramer 2010); All turtle species fully protected in Bahamas and South Berry Islands Marine Reserve established.
2010:	No Name Marine Reserve, Abaco established.
2011:	Bahamas-wide shark sanctuary established. West Side National Park expanded to 1.3 million acres; Coral cover, macroalgal cover and total fish biomass were 8%, 64%, and 11,362 g/100m^2 at Cay Sal, 12%, 26%, and 15,843 g/100m^2 at Great Inagua, 5%, 49%, and 9,844 g/100m^2 at Hogsty Reef, 11%, 33%, and 11,911 g/100m^2 at Little Inagua and 10%, 38% and 21,480 g/100m^2 at southern Andros (Bruckner 2011); Hurricane Irene caused disturbance to some large sponges and bioeroded corals, and decreased, for a limited time, macroalgae cover on reefs in the Exuma Cays.

2012: In the Exuma Cays, coral cover on shallow (<10m) channel reefs was 21-48%, algal cover was 12-41%; on shallow fringing reefs, coral cover was 4-32% and algal cover was 35-73% (Brumbaugh et al. 2013); Fowl Cays National Park, Conception Island Marine Environment (an extension of the previous National Park).

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BARBADOS

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Geographic Information

Coastal Length:	96 km
Land Area:	443 km^2
Maritime Area:	186,827 km^2
Population:	282,819
Reef Area:	62 km^2
Number of hurricanes in the past 20 years:	0

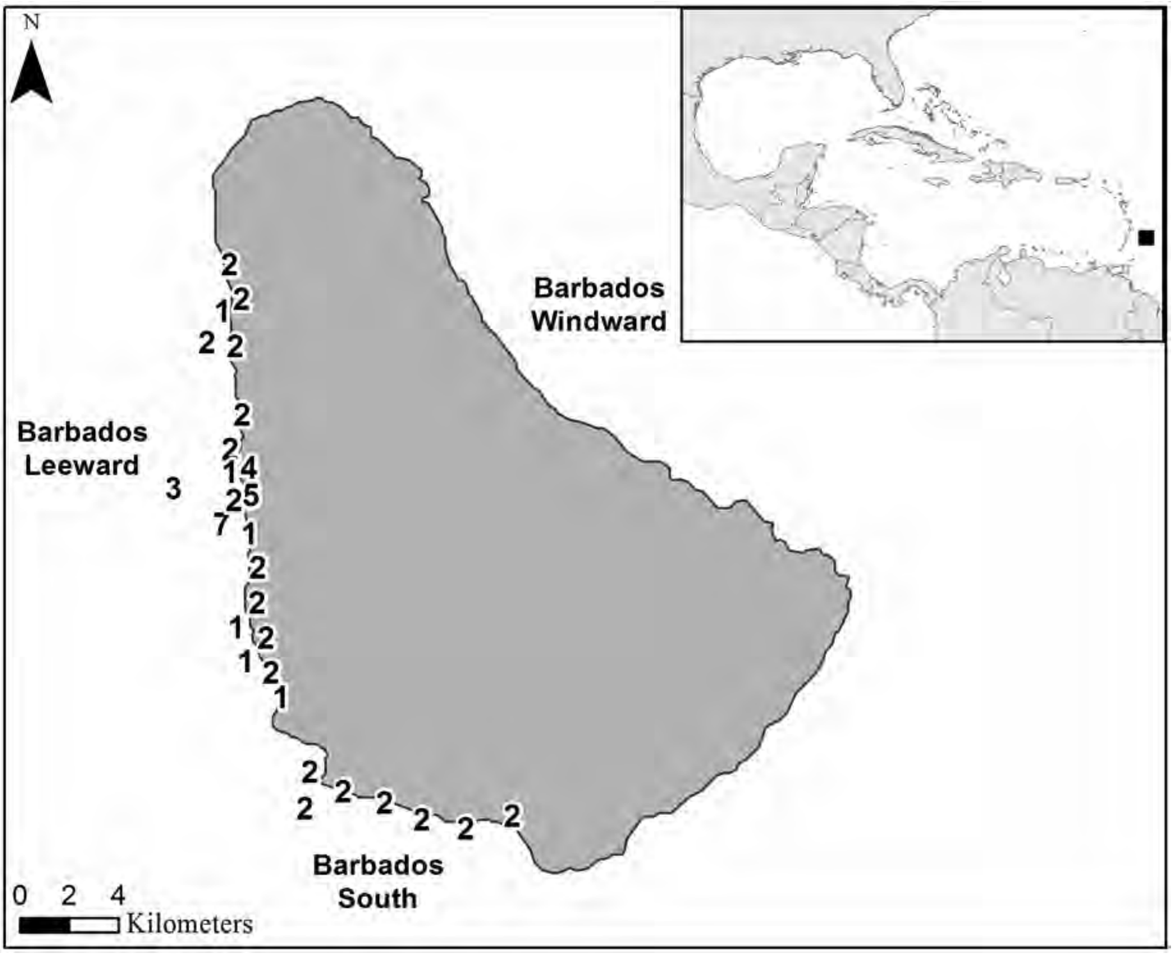

Fig. 3.1 Map of Barbados, codes represent studies listed in Table 3.1. Missing map code(s) due to unavailable coordinates.

Table 3.1 Data sources from Barbados used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 3.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Allard 1994[1]	1992	1	X			
2	Brathwaite, Angelique*[2]	1982, 1987, 1992, 1997, 2002, 2007	6	X	X	X	
3	Liddell & Ohlhorst 1988[3]	1977	1	X		X	
4	Oxenford, Hazel/ CARICOMP*[4]	1993-1994, 1998-2002, 2004-2006	10	X	X	X	
5	Scoffin 1993[5]	1974, 1981, 1992	3	X			
6	Tomascik & Sander 1987[6]	1982	1	X	X	X	
7	Williams, Ivor; Polunin, Nicholas*[7]	1997	1	X		X	
9	Hawkins & Lewis 1982[8]	1975-1976	2		X		
a	Hunte et al. 1986[9]	1983-1984	2		X		
b	Hunte & Younglao 1988[10]	1983-1985	3		X		
c	Reef Check*	1997, 2001-2005	6		X		
d	Scoffin 1980[11]	1978	1		X		
e	Bauer 1980[12]	1978	1		X		

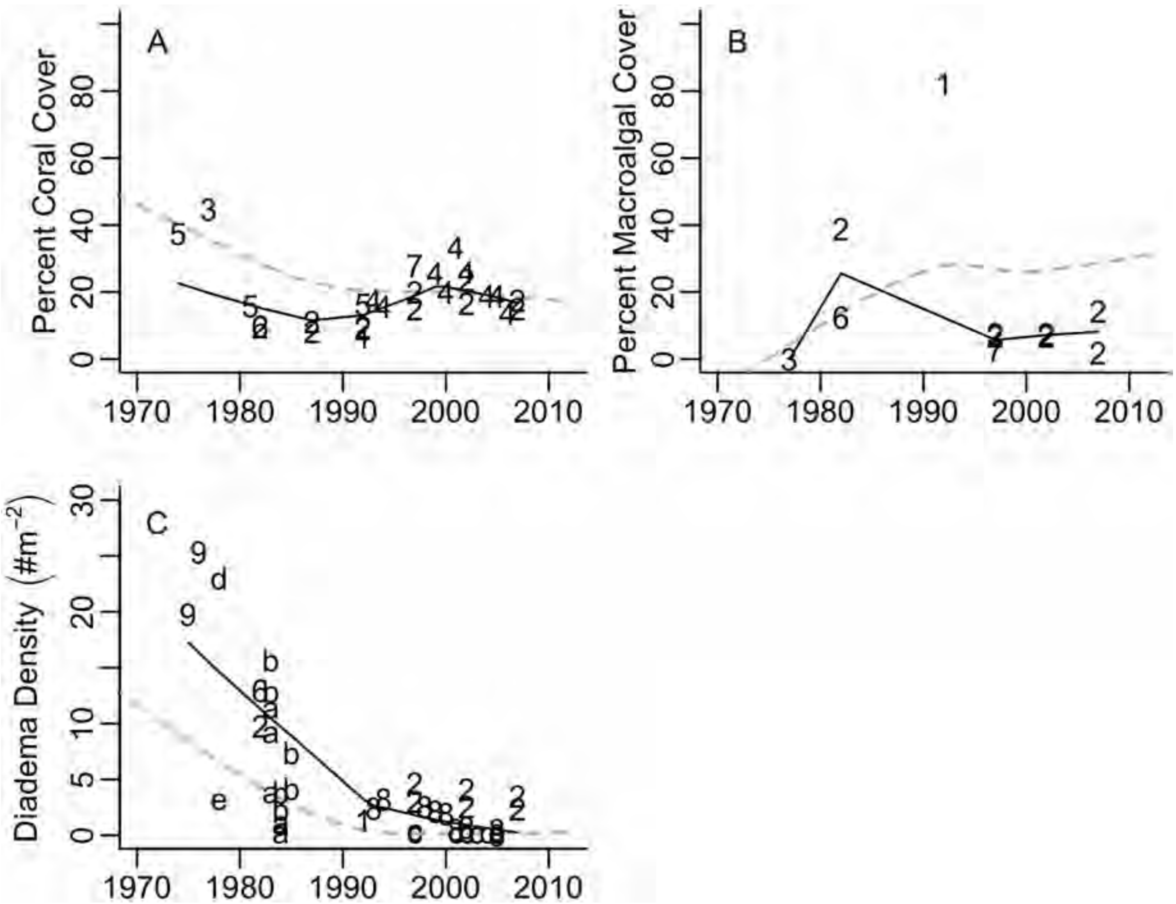

Fig. 3.2 Average percent cover of live corals (A) and macroalgae (B), and density of *Diadema antillarum* (C) in Barbados. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 3.1 and Figure 3.1)

Timeline

1640s:	Clearing of land for sugarcane cultivation resulted in the removal of forests and massive amounts of erosion and sedimentation. Resulted in final demise of *Acropora* reefs on the SE Coast
1950s:	Expansion of the fishing fleet
1955:	Hurricane Janet (Category 5)
1950s-60s:	Tourism boom begins
1960:	Bellairs Research Institute, a field station for McGill University was established, and some of the earliest records of reef health recorded.
1960:	Lewis categorizes platforms at Six Mens Bay on the west coast of Barbados
1974:	Cave Hill Campus of the University of the West Indies established and stimulates reef research
Early 1980s:	Acroporid mass mortality from White Band Disease
1982:	Coastal Zone Project Unit established and starts the National Coral Reef Monitoring Programme, with local and international scientists; First documentation of a eutrophication gradient on the west coast of the island (Tomascik & Sander 1985); Establishment of the Bridgetown Sewage Treatment Plant
1983:	*Diadema antillarum* mass mortality
1987:	Second monitoring event for the National Coral Reef Monitoring Programme
1992:	Third monitoring event for the National Coral Reef Monitoring Programme
1997:	Fourth monitoring event for the National Coral Reef Monitoring Programme
1998:	Bleaching event
2001-2002:	First coral disease survey carried out, low but pervasive levels recorded
2002:	Establishment of the South Coast Sewerage Project, largely as a result of proving that corals deteriorated along a eutrophication gradient on the west coast; Fifth monitoring event for the National Coral Reef Monitoring Programme
2003 – present:	Sporadic outbreaks of yellow band disease and black band disease observed, while white plague disease, dark spot disease and Aspergillosis remain at low levels.
2005:	Most severe mass bleaching event recorded, with an average of 70.6% corals bleached in October, and high bleaching associated mortality over the following year resulting in a loss of 25.9% live coral cover
2007:	Sixth monitoring event for the National Coral Reef Monitoring Programme
2010:	Mass bleaching, not as severe as in 2005 with an average of 37.3% corals bleached in October, and bleaching associated mortality resulting in the loss of around 8% coral cover within the following year
2011:	First lionfish observed
2012:	Seventh monitoring event for the National Coral Reef Monitoring Programme

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BELIZE

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Geographic Information

Coastal Length:	1,984 km
Land Area:	22,153 km^2
Maritime Area:	34,701 km^2
Reef Area:	1,668 km^2
Number of hurricanes in the past 20 years:	5

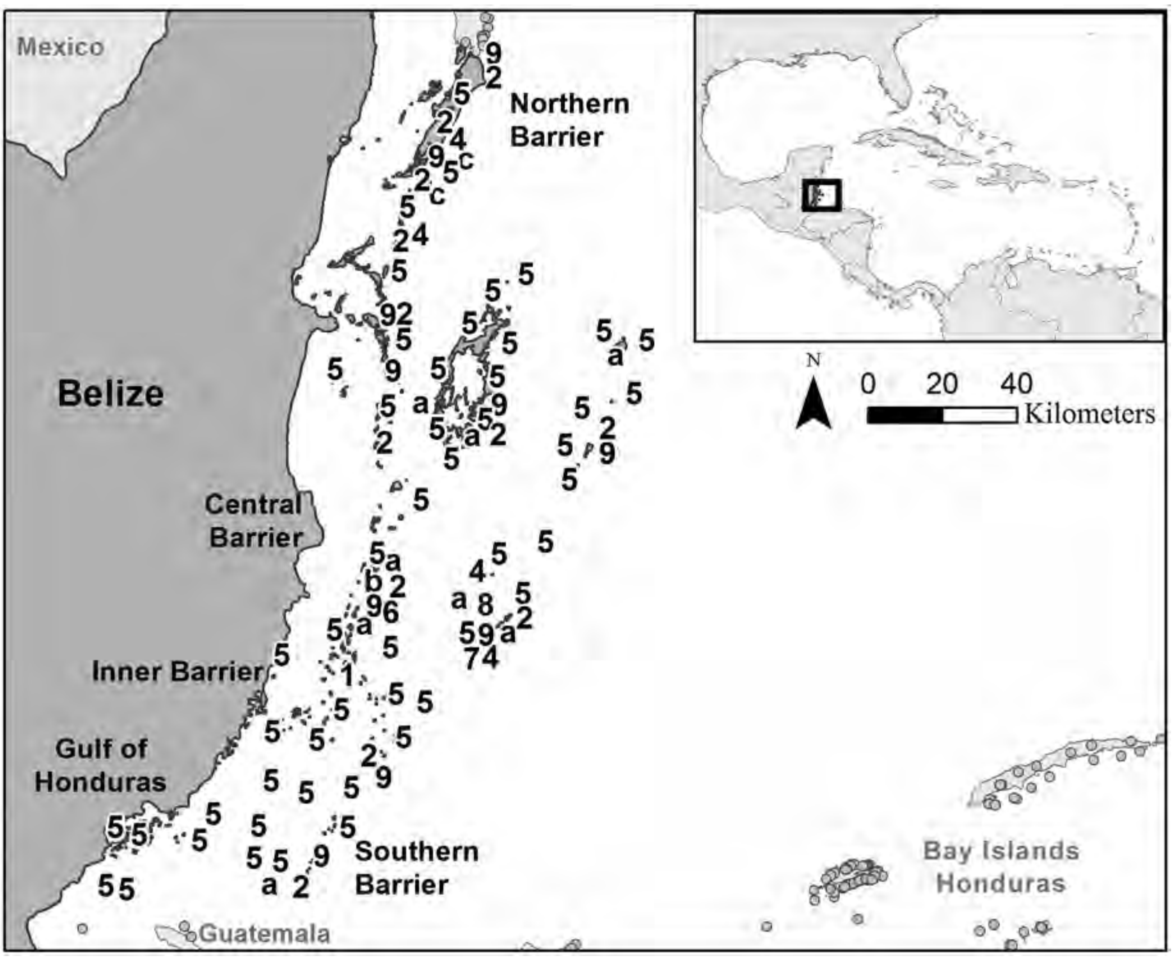

Fig. 4.1 Map of Belize, codes represent studies listed in Table 4.1. Missing map code(s) due to unavailable coordinates.

Table 4.1 Data sources for Belize used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 4.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Aronson et al. 2000[1]	Barrier	1994-2001	8	X	X	X	
2	Bruno, John*	Atoll, Barrier	2009-2011	3	X		X	
3	Forman, Kirah*[2,3,4,5,6]	Barrier	2005-2012	8	X		X	
4	Hardt, Marah; Paredes, Gustavo*[7]	Atoll, Barrier	2004	1	X		X	X
5	Heathy Reefs Initiative & AGRRA*[8]	Atoll, Barrier (197 sites)	1999-2000, 2006, 2008-2009	5	X	X	X	X
7	Koltes, Karen; Tschirky, John/ CARICOMP*[9]	Barrier	1994-2012	17	X	X	X	
6	McClanahan et al. 2001[10]	Atoll	1998-1999	2	X		X	
8	McClanahan & Muthiga 1998[11,12]	Atoll	1970, 1996	2	X		X	
9	Mcfield, Melanie*[13,14,15,16]	Barrier	1997-1999	2	X		X	X
a	Mumby, Peter*[17,18]	Atoll, Barrier	2002	1	X		X	
b	Rützler & Macintyre 1982[19]	Barrier	1978	1	X		X	
c	Williams, Ivor; Polunin, Nicholas*[20]	Barrier	1998	1	X		X	X
d	Reef Check*	Atoll, Barrier (36 sites)	1997, 2001, 2004-2007	6		X		
e	Hay 1984[21]	Atoll, Barrier	1981	1		X		
f	Brown 2007[22]	Barrier	2003	1		X		
g	Lewis & Wainright 1985[23]	Barrier	1983	1		X		
h	Bood, Nadia*		2001, 2005	2		X		
i	Majil, Isaias*		2004-2005	2		X		
j	Steneck, Bob*		2002-2003, 2005, 2008, 2010	5	X		X	
k	Dustan, Phil*[24]	Atoll	1993	1	X		X	
m	Lessios 1988[25]	Barrier	1983-1984	2		X		

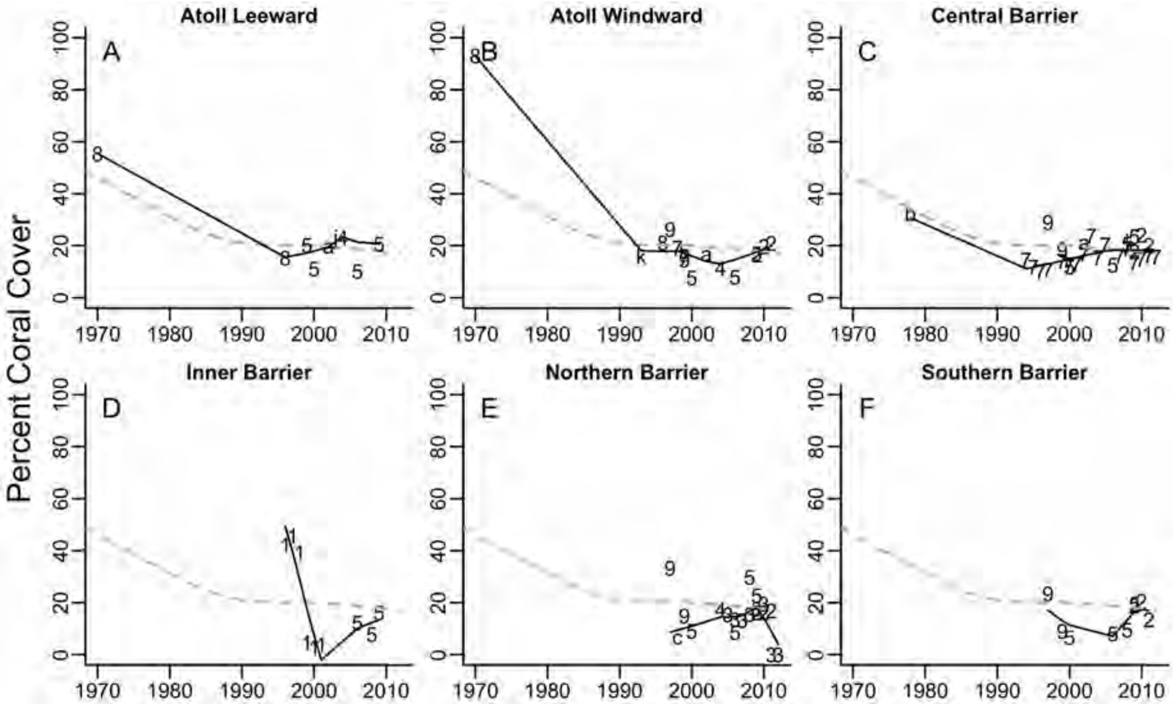

Fig. 4.2 Average percent cover of live corals for 6 locations in Belize: Atoll Leeward (A), Atoll Windward (B), Central Barrier (C), Inner Barrier (D), Northern Barrier (E) and Southern Barrier (F). Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 4.1 and Figure 4.1)

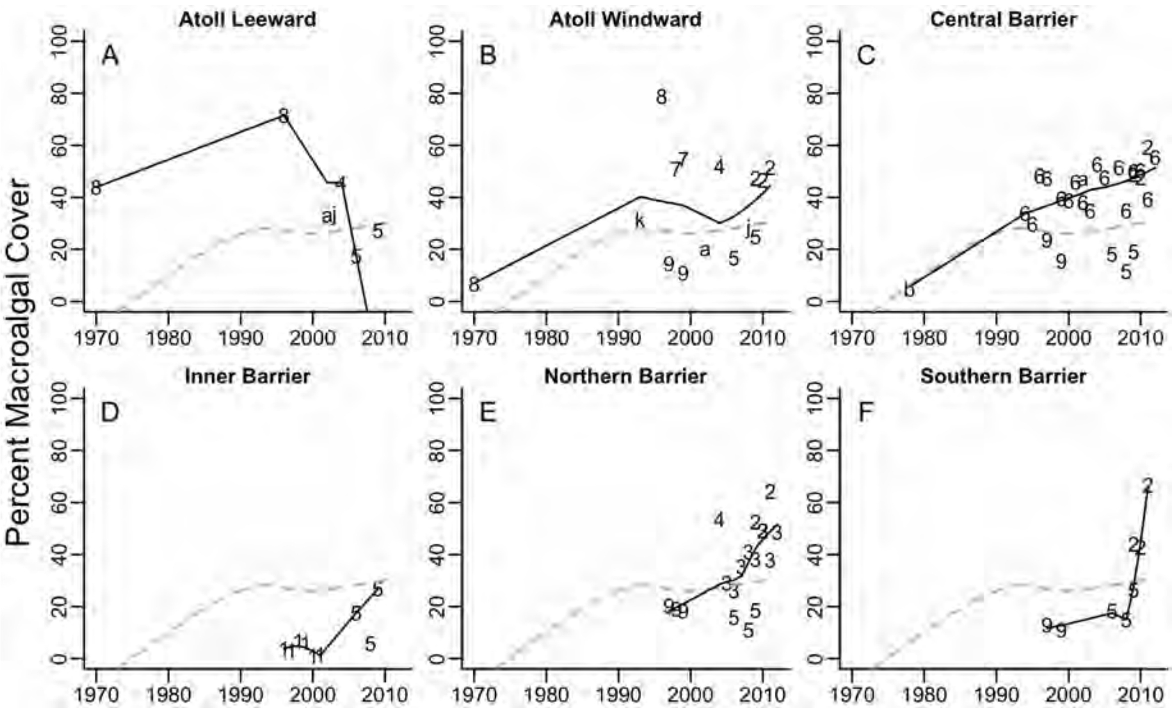

Fig. 4.3 Average percent cover of macroalgae for 6 locations in Belize: Atoll Leeward (A), Atoll Windward (B), Central Barrier (C), Inner Barrier (D), Northern Barrier (E) and Southern Barrier (F). Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 1 and Figure 1)

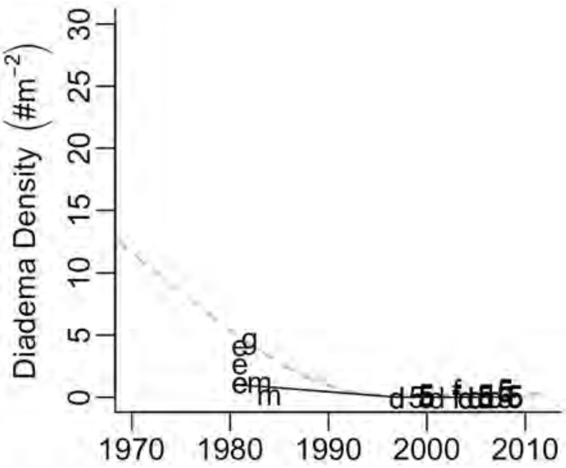

Fig. 4.4 Average density of *Diadema antillarum* for all Belize locations combined. Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 4.1 and Figure 4.1)

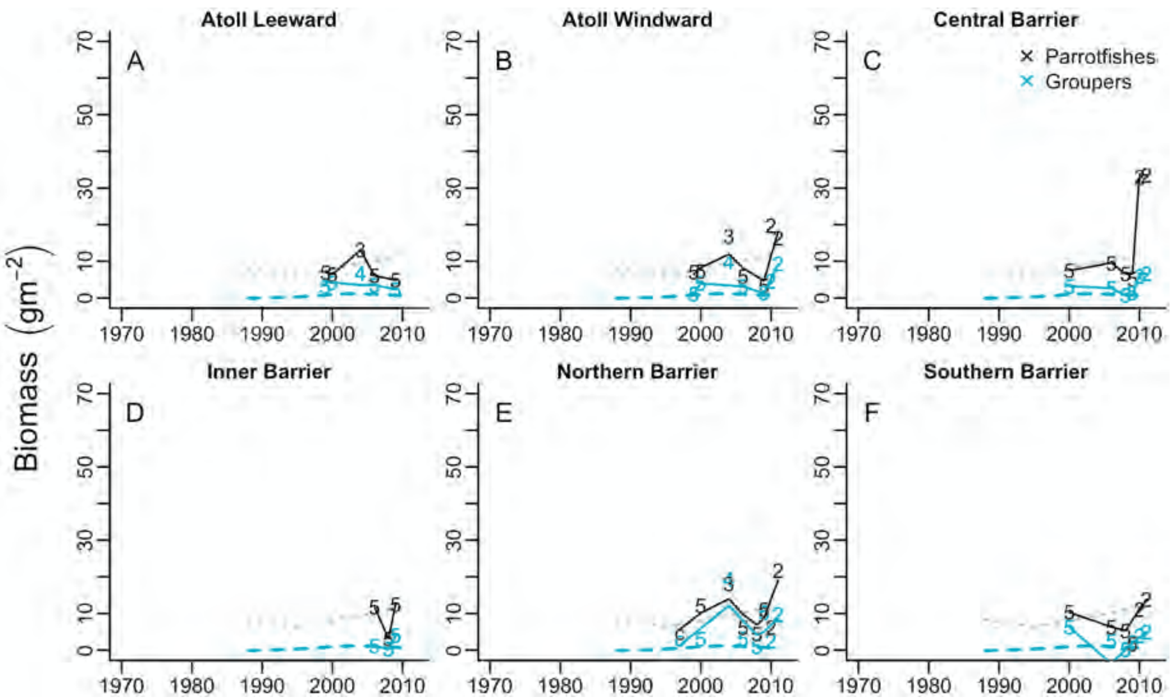

Fig. 4.5 Average biomass of parrotfishes and groupers for 6 locations in Belize: Atoll Leeward (A), Atoll Windward (B), Central Barrier (C), Inner Barrier (D), Northern Barrier (E) and Southern Barrier (F). Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 4.1 and Figure 4.1)

Timeline

1950s:	Development of commercial fishing
1960s:	Forming of fishing cooperatives
1972:	Research started at Carrie Bow Cay, coral cover around 60%
1981:	British Honduras becomes Belize
1981:	Half Moon Caye Natural Monument declared (first MPA)
1983-1984:	Mass mortality of Diadema

1987:	Hol Chan Marine Reserve established, coral cover <70%
1990:	Coastal Zone Management Unit (CZMU) established by the Government
Early 1990s:	Rapid increase of tourism
1995:	First mass coral bleaching event
Late 1990s:	Rapid clearing and development, dredging and filling of Cayes begin
1998:	Bleaching event (began late-summer)
1998:	Hurricane Mitch (Category 5, Oct/Nov – affected whole reef)
2001:	Hurricane Iris (Category 4, isolated path in southern reef)
2008:	Lionfish first reported in Belize
2009:	National ban on harvesting herbivores
2010:	Fisheries Act revisions begin; Belize population quadrupled and tourist numbers increased by 20-fold compared to 1960s
2011:	Unprecedented phytoplankton bloom

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BERMUDA

Coauthors: Mike Colella, Anne Glasspool, Ross Jones, Sheila McKenna, Jessie M.H. Murdoch, Thaddeus Murdoch, Ivan Nagelkerken, Tim Noyes, Joanna Pitt, Struan R. Smith, Wolfgang Sterrer, Gerardo Toro Farmer, Jack Ward, Ernesto Weil and CARICOMP

Geographic Information	
Coastal Length:	184 km
Land Area:	54 km^2
Maritime Area:	449,463 km^2
Population:	59,991
Reef Area:	672 km^2
Number of hurricanes in the past 20 years:	1

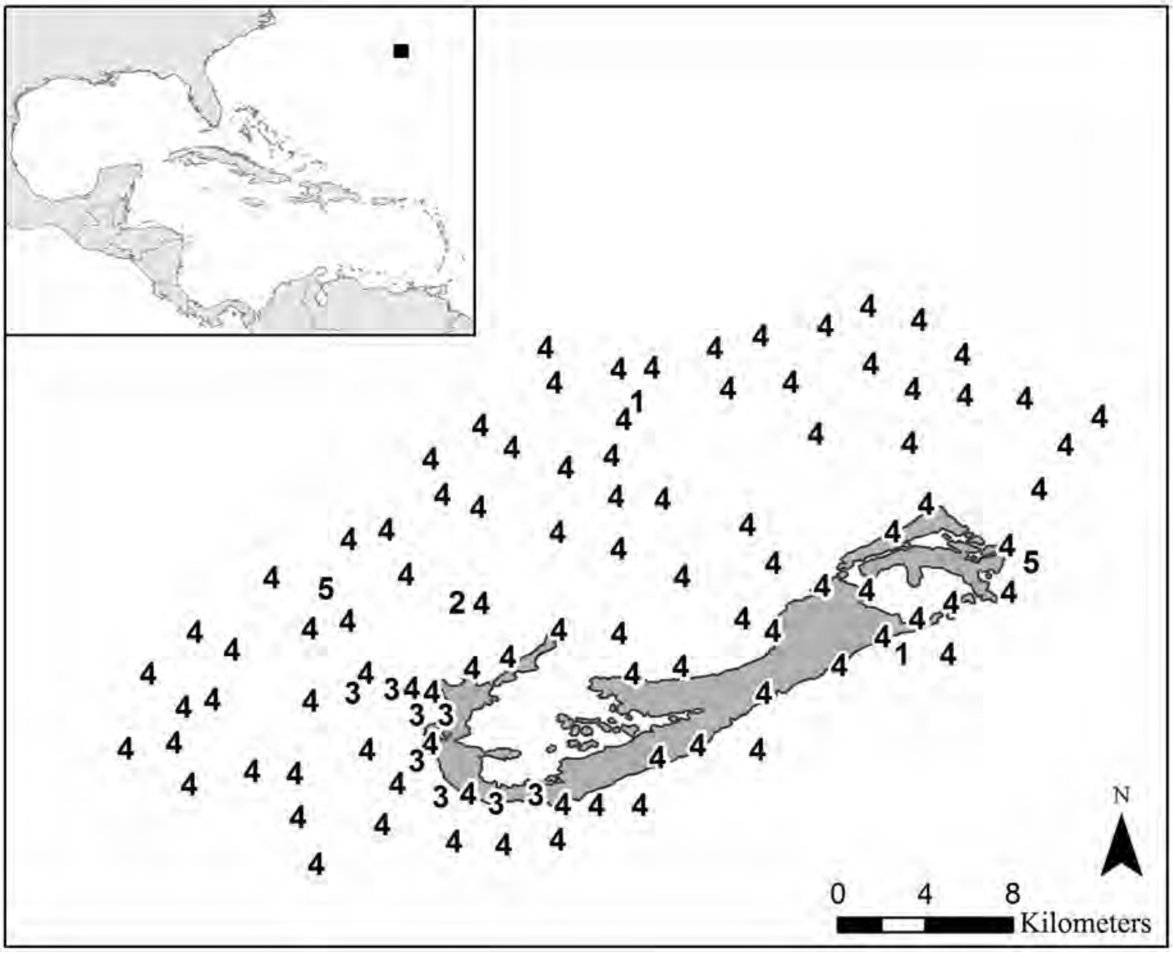

Fig. 5.1 Map of Bermuda, codes represent studies listed in Table 5.1. Missing map code(s) due to unavailable coordinates.

Table 5.1 Data sources from Bermuda used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 5.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Dodge et al. 1982[1]	1978	1	X			
2	CARICOMP*[2]	1993-1995, 1997-2002, 2004-2005. 2007	12	X	X	X	
3	Nagelkerken, Ivan*[3]	2005	1	X			X
4	Murdoch, Thaddeus*[4,5]	2004-2007, 2010-2012	7	X	X	X	X
5	Weil, Ernesto*	2005-2006, 2009	3	X		X	
6	Bauer 1980[6]	1977	1				
7	Pitt, Joanna[7]	2004	1				X

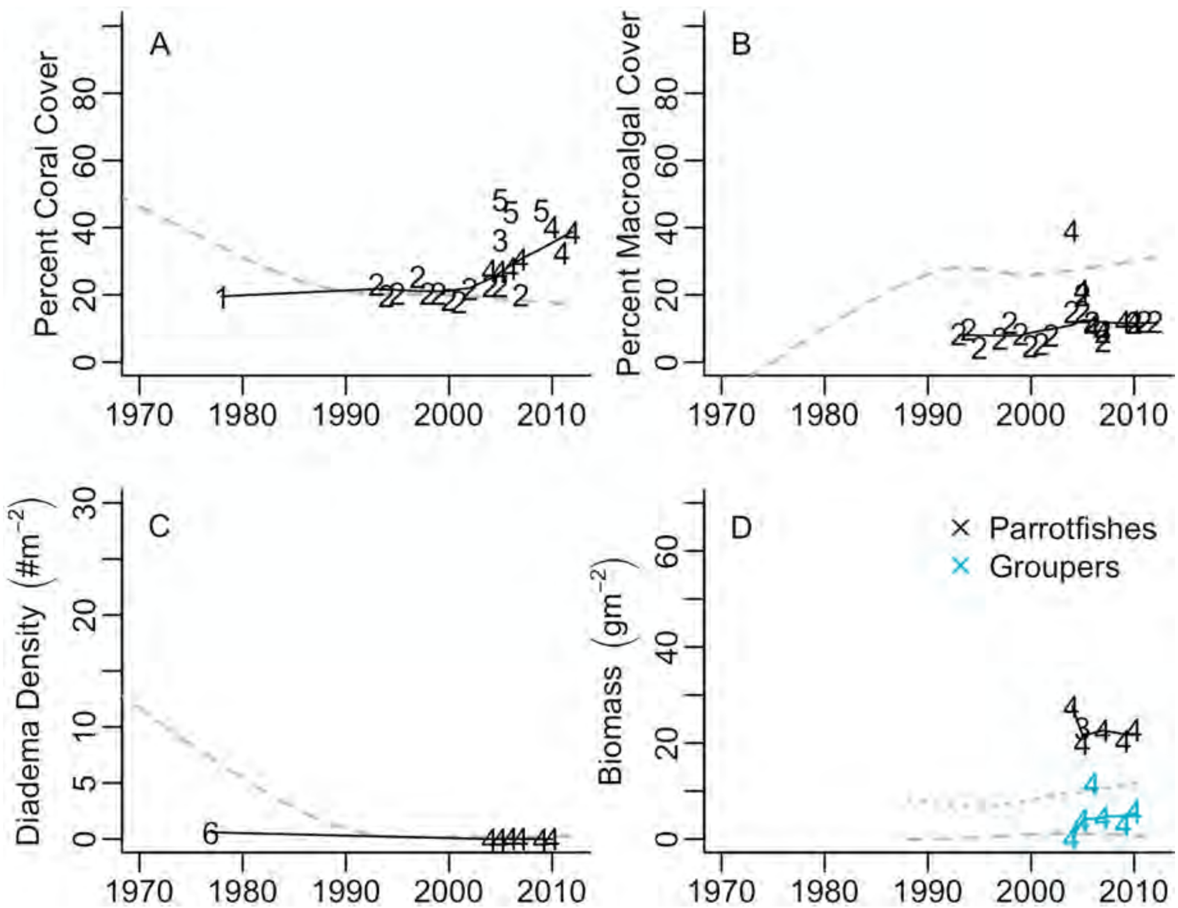

Fig. 5.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C), and biomass of groupers and parrotfishes (D) in Bermuda. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented (Codes same as in Table 5.1 and Figure 5.1).

Timeline

1791: First law passed to prohibit the use of fish pots

1789: Lt. Thomas Hurd and the Royal Navy initiates the first very high accuracy mapping of Bermuda coral reefs across the entire platform, which was completed in 1798

1912: First Fisheries Act passed, with restrictions on taking of certain species, imposition of minimum sizes, closing of specific areas to fishing

1941: Massive land-reclamation projects, via blasting and dredging, began in Castle Harbour and removed 24 hectares of coral reefs and 18 hectares of seagrass beds in Castle Harbour, local extinction of 10 fish species

1947: Fisheries Regulations established controls on fishing nets, taking of lobster during the breeding season, minimum size and weight restrictions for certain species, restriction on use of fish pots in inshore areas

1963: Fisheries Regulations updated with restriction on fishing, registration of fishermen, and control of spear-fishing that prohibited the use of "aqualungs"

1966: North shore and south shore coral reef preserves created; netting of parrotfishes along the south shore was prohibited. Taking of corals in preserves is prohibited

1972: Second Fisheries Act and Fisheries Regulations imposing restriction on the use of fish pots, licensing of fishermen and fishing vessels and reporting of catch statistics

1975: First coral disease, Black Band Disease reported
Red hind spawning grounds closed to fishing May through August under the Fisheries (Protected Areas) Order

1978: Fisheries (Protected Species) Order enacted to protect all corals, conchs, specific bivalves and gastropods1980: Moratorium on the issuance of permits for new fish pots

1983: *Diadema antillarum* die-off recorded

1987: Hurricane Emily (Category 3)

1988: First coral bleaching event reported, recurring bleaching events have occurred sporadically since 1988 but have never resulted in a substantive mortality

1990: Fish pot fishery is banned, *ex-gratia* payments made to fishermen; fisheries (Protected Areas) Order establishes eleven marine protected areas

1992: CARICOMP monitoring programme for coral reefs, seagrass beds and mangroves initiated by S.R. Smith. Coral cover remained stable from 1992 to 2002

1993: All parrotfishes (Scaridae) added to the Fisheries (Protected Species) Order

1995: Minor coral bleaching noted

1996: Fisheries (Protected Areas) Order establishes another marine protected area
The Bermuda Biodiversity Project (BBP), a collaboration of the Bermuda Natural History Museum and the Bermuda Zoological Society, was initiated by Wolfgang Sterrer and Anne GlasspoolVarious large grouper species (*Epinephelus spp.* and *Mycteroperca spp.*) added to the Fisheries (Protected Species) Order

1997: Fisheries (Protected Areas) Order establishes another marine protected area
Jack Ward of the Bermuda Aquarium, Museum and Zoo, and Anne Glasspool of BBP, commission the first digital island-wide high-resolution georeferenced aerial mosaic of the entire shallow reef platform

1998: First coral disease surveys initiated by C. McKinney and S.R. Smith; documented low levels of incidence and minor coral bleaching
Annie Glasspool (BBP) directs the Biodiversity Strategy and Action Plan for Bermuda, with UK DEFRA (Darwin) funding

2000: Fisheries (Protected Areas) Order establishes 16 more marine protected areas, bringing the total to 29. Seasonal closure of spawning areas extended; CARICOMP monitoring program documents complete loss of seagrasses at study sites
Anne Glasspool (BBP) directs the Biodiversity Strategy and Action Plan for Bermuda, with UK DEFRA (Darwin) funding
Anne Glasspool (BBP) and Jack Ward (Department of Conservation
Services) initiate Atlantic Gulf Rapid Reef Assessment (AGRRA) surveys as part of BBP
Judie Clee (Bermuda Zoological Society) establishes first REEF (Reef Environmental Education Foundation) field station in Bermuda at Bermuda Aquarium, Museum and Zoo
First invasive lionfish (*Pterois volitans*) reported

2003: Hurricane Fabian (Category 4), preceded by high SSTs and coral bleaching, that diminished after the hurricaneProtected Species Act 2003 enacted affording protection to listed marine species and critical marine habitats in accordance with IUCN criteria. (Five grouper species, two species of seahorse, four marine turtle species, whale shark and queen conch listed); Reef surveys, including coral disease and bleaching, initiated by R.J. Jones continue annually until 2011

2004: Thaddeus Murdoch initiates the Bermuda Reef Ecosystem Assessment and Mapping (BREAM) Program at BZS, as the marine component of the BBP. BREAM maps to GIS all reefs and other marine habitats. BREAM initiates biodiversity assessment of lagoonal and rim reef habitats

2005: Minor coral mortality event related to UV damage during low tides in April; minor coral bleaching event in August; Disease incidence remains low but four new diseases syndromes detectedBoat bottom paints containing organotin, irgarol or diuron compounds are prohibited
Bermuda Government presents White Paper on the Marine Environment and the Fishing Industry in Bermuda

2006: Bermuda Benthic Habitat Mapping, Monitoring and Assessment Programme (BBMAP) established by S. Manuel to study 17 permanent seagrass and water quality sites, providing the first comprehensive platform-wide time series

2007: Massive loss of lagoonal seagrass beds reported by Murdoch et al., expanding scope from the CARICOMP study-BREAM receives NOAA funding to assess MPA sites across the island for diver impact
Eagle ray population ecology studies initiated by Matthew Ajemian of the Dauphin Island Sea Lab AL, USA, with BREAM support

	First seasonal closure for blue-striped grunt (Haemulidae) spawning aggregation area off St. Georges. Fishing prohibited May-June Chris Flook initiates Bermuda Lionfish Project to investigate extent of invasive lionfish and develop mitigation measures
2008:	First extended closure of the immediate area around the eastern black grouper spawning aggregation. Fishing prohibited 90 days from September 1st, under the Fisheries Act 1972
2009:	BREAM initiates assessment of 100 MPA and forereef sites to 30m depth for fishes and benthic biota
2010:	First extended closure of the immediate area around a western black grouper spawning aggregation. Fishing prohibited 90 days from September 1st, under the Fisheries Act 1972; Invasive lionfish become more common on fore-reef (20-60m) Fisheries Regulations amended again. Licenses now required for free-diving pole-spear fishing. Other spear-fishing gear and use of SCUBA remain illegal Recently discovered historic map of Bermuda's clearly shows loss of reefs to channel dredging and base construction-Spotted eagle rays added to Protected Species list, in response to fishing pressure
2011:	Graham Maddocks establishes Ocean Support Foundation building on Bermuda Lionfish Project's work to tackle invasive Lionfish
2012:	Protected Species Order 2012 adds five species of seagrass, two scallop species, American and European eels and humpback and sperm whales

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BONAIRE

Coauthors: Rolf Bak, Paul Hoetjes, Ayana Elizabeth Johnson, Ramon de Leon, Kalli de Meyer, Gabriel Grimsditch, Peter L. Harrison, Erik Meesters, Ivan Nagelkerken, Gerard Nieuwland, Maggy Nugues, Stuart Sandin, Sanders R. Scheffers, Brigitte Sommer, Bob Steneck, AGRRA, CARICOMP and Reef Check

Geographic Information

Coastal Length:	120 km
Land Area:	279 km^2
Maritime Area:	3,569 km^2
Population:	13,453
Reef Area:	68 km^2
Number of hurricanes in the past 20 years:	0

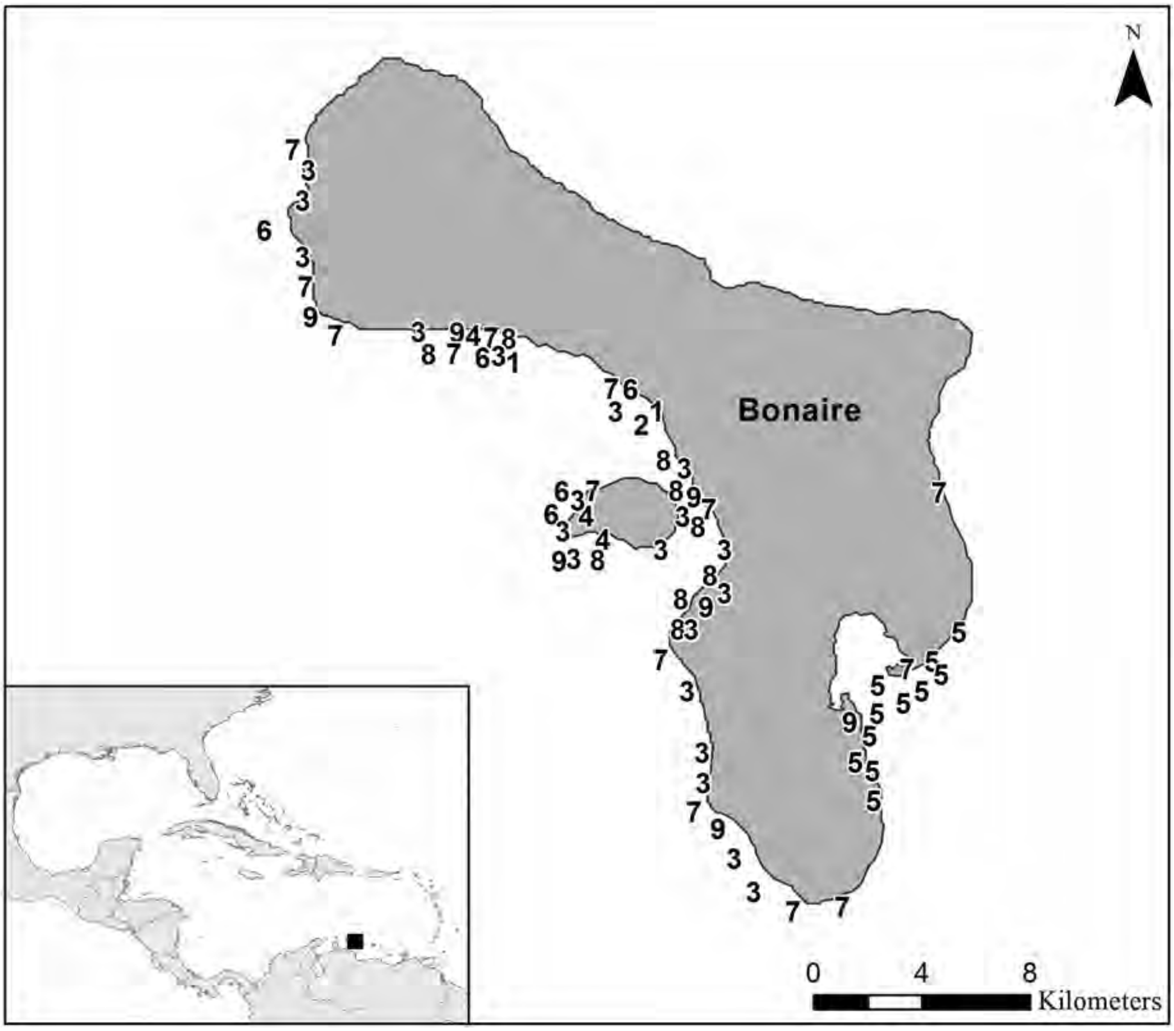

Fig. 6.1 Map of Bonaire, codes represent studies listed in Table 6.1. Missing map code(s) due to unavailable coordinates.

Table 6.1 Data sources for Bonaire used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 6.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Bak, Rolf,; Nugues, Maggy; Niewland, Gerard, Meesters, Erik*[1,2,3,4]	Leeward	1973-74, 1979, 1983, 1989, 1991-1993, 1997-1999, 2001-2003, 2006, 2008-09	17	X		X	
2	De Meyer, Kalli/ CARICOMP*[5]	Leeward	1994-1997	4	X	X	X	
3	Grimsditch, Gabriel*[6]	Leeward	2009	1	X		X	X
4	Hawkins et al. 1997[7]	Leeward	1991, 1994	2	X			
5	Nagelkerken, Ivan*[8]	Windward	2005	1	X			X
6	AGRRA*[10]	Leeward	1999	1	X	X		X
7	Sommer, Brigitte*[11]	14 sites island wide	2008-2009	2	X		X	
8	Steneck, Bob*[12,13,14,15,16]	Leeward	1999, 2002-2003, 2005, 2007, 2009, 2011	7	X	X	X	X
9	Stokes et al. 2010[17]	7 sites island wide	1982, 1988, 2008	3	X		X	
a	Sandin, Stuart*[18]	6 locations island wide	2001	1				X
b	Reef Check*	Leeward	1997-1998, 2000-2003	6		X		

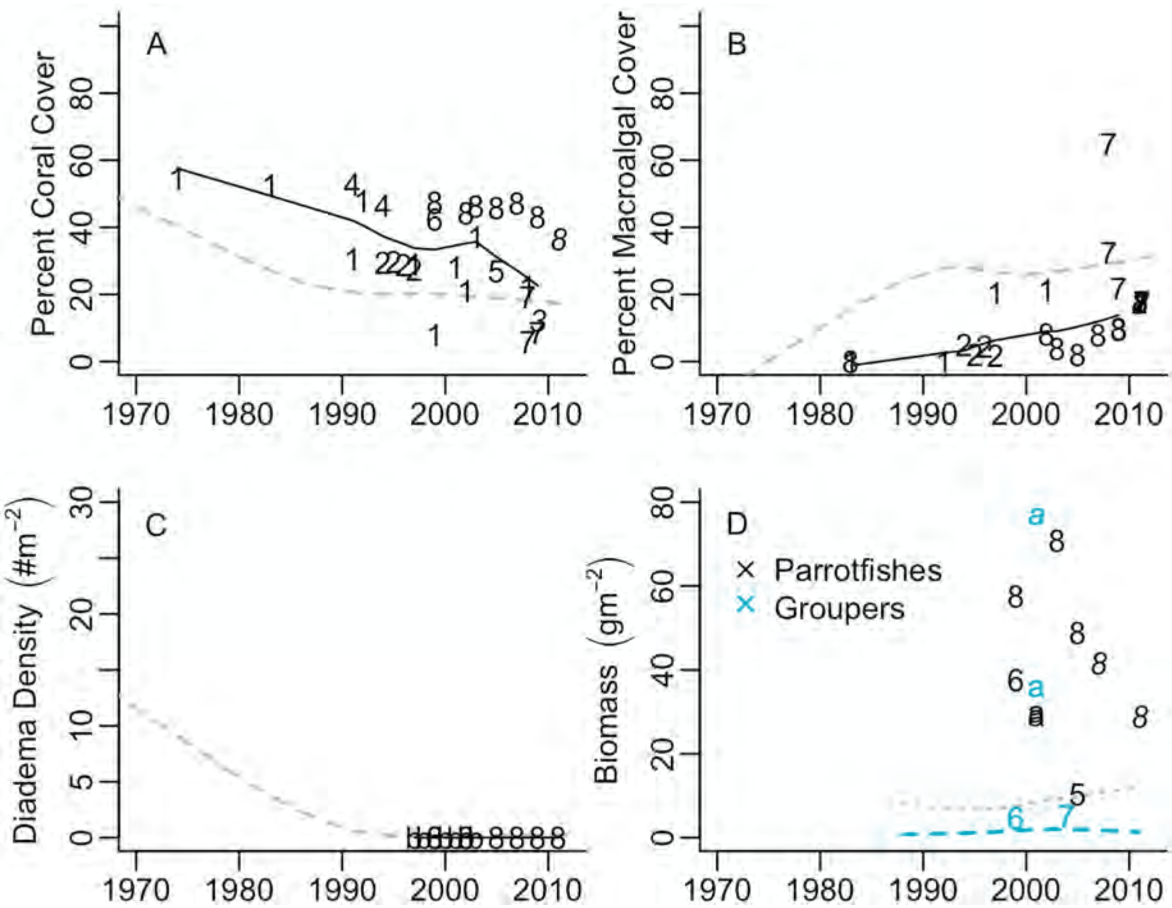

Fig. 6.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C), and biomass of groupers and parrotfishes (D) in Bonaire. Dotted line represents the average of Caribbean data collected for this report, solid lines are drawn through data presented for Leeward Bonaire; additional data from Windward Bonaire are indicated by italic text. (Codes same as in Table 6.1 and Figure 6.1)

Timeline

1950s: Synthetic fish lines introduced

1960: Large fishes commonly seen

1961: Minimum catch size for lobsters & regulation protecting sea turtles, sea turtle eggs and nesting areas on Bonaire

1963: Regulation of the use of dragging nets

1971: Use of spear guns banned

1975: Harvesting of corals banned

1979: Bonaire Marine Park established

1980: Joined RAMSAR Convention on Wetlands

1983: White band disease wipes out Acropora cervicornis

1983: Mass mortality of *Diadema antillarum*

1988: Tropical storm Joan

1993: Very little or no macroalgae found on Bonaire's reefs (Bob Steneck, pers. Comm.)

1999: AGRRA survey finds Bonaire has the highest coral cover (nearly 50%) and lowest algal cover in the Caribbean (Kramer 2003); Hurricane Lenny (Category 3) hits the normally sheltered SW coast with localized effects (Rolf Bak, pers. comm.); no obvious phase shift to macroalgae (Bob Steneck, pers. comm.)

2008: No fishing areas established in Bonaire and predator populations began to increase; Hurricane Omar with localized effects

2009: First lionfish detected, removal schemes pre-approved; Population reached 140,000

2010: Parrotfish catches banned; fish traps licensed for phase out; new permit system for fish nets; warm temperatures caused beaching in 10-20% corals, death of 10% coral; algal cover increased from 4% to 8%; Tropical Storm Tomas

2011: Algae increased from 5% to 15% (Bob Steneck, pers. comm.)

2012: Official lionfish removals commence

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BRITISH VIRGIN ISLANDS

Coauthors: Graham Forrester, Shannon Gore, Nancy Woodfield Pascoe, Ken Pemberton, AGRRA and Reef Check

Geographic Information

Coastal Length:	342 km
Land Area:	178 km^2
Maritime Area:	80,630 km^2
Population:	22,802
Reef Area:	266 km^2
Number of hurricanes in the past 20 years:	7

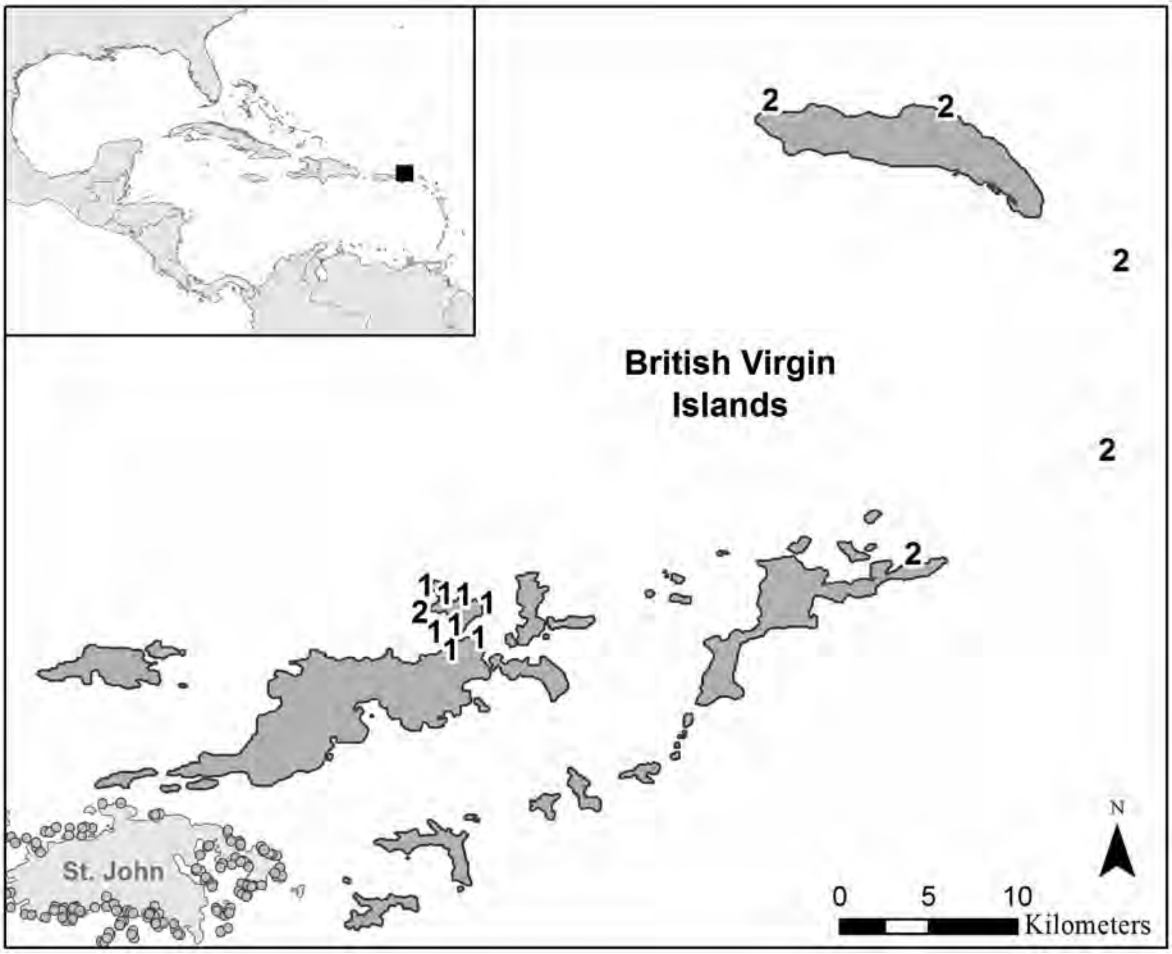

Fig. 7.1 Map of British Virgin Islands, codes represent studies listed in Table 7.1. Missing map code(s) due to unavailable coordinates.

Table 7.1 Data sources from the British Virgin Islands used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 7.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Forrester, Graham*1,2,3	Guana Island	1992-2012	21	X		X	X
2	AGRRA*4	5 sites in Eastern BVI	1999-2000	2	X	X		X
3	Bauer 1980*5	Peter Island	1977-1978	2		X		
4	Reef Check*	Tortola	1997-2007	11		X		

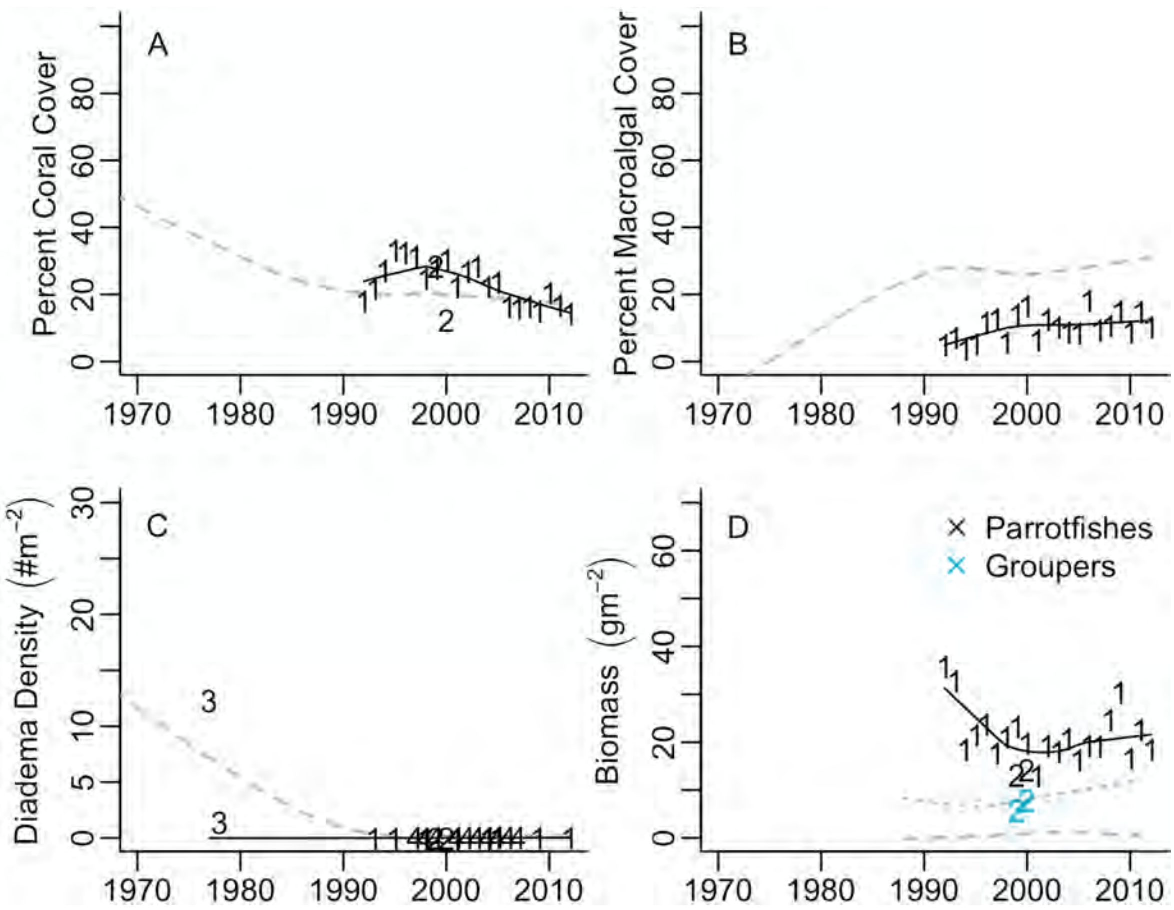

Fig. 7.2 Average percent cover of live corals and macroalgae, density of *Diadema antillarum*, in the British Virgin Islands. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented (Codes same as in Table 7.1 and Figure 7.1).

Timeline

1980s:	Mass mortality of *Acropora* species
1983:	Mass mortality of *Diadema antillarum*
1987-88:	Bleaching event
1989:	Hurricane Hugo (Category 4)
1995:	Hurricanes Luis and Marilyn (Category 4 and 2 respectively)
1999:	Hurricane Lenny (Category 4)
2000s:	Increasing damage to reefs from yacht anchoring, increasingly frequent sedimentation from road construction and coastal development
2005-6:	Bleaching event, affecting 20-50% coral cover followed by disease outbreaks
2008:	Lionfish *Pterois volitans* first documented

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CAYMAN ISLANDS

Coauthors: Timothy Austin, Phillippe Bush, Douglas Fenner, Carrie Manfrino, Croy McCoy, Jeff Miller, Ivan Nagelkerken, Nicholas Polunin, Ernesto Weil, Ivor Williams, AGRRA, CARICOMP and Reef Check

Geographic Information

Coastal Length:	217 km
Land Area:	287 km^2
Maritime Area:	123,274 km^2
Population:	47,502
Reef Area:	231 km^2
Number of hurricanes in the past 20 years:	4

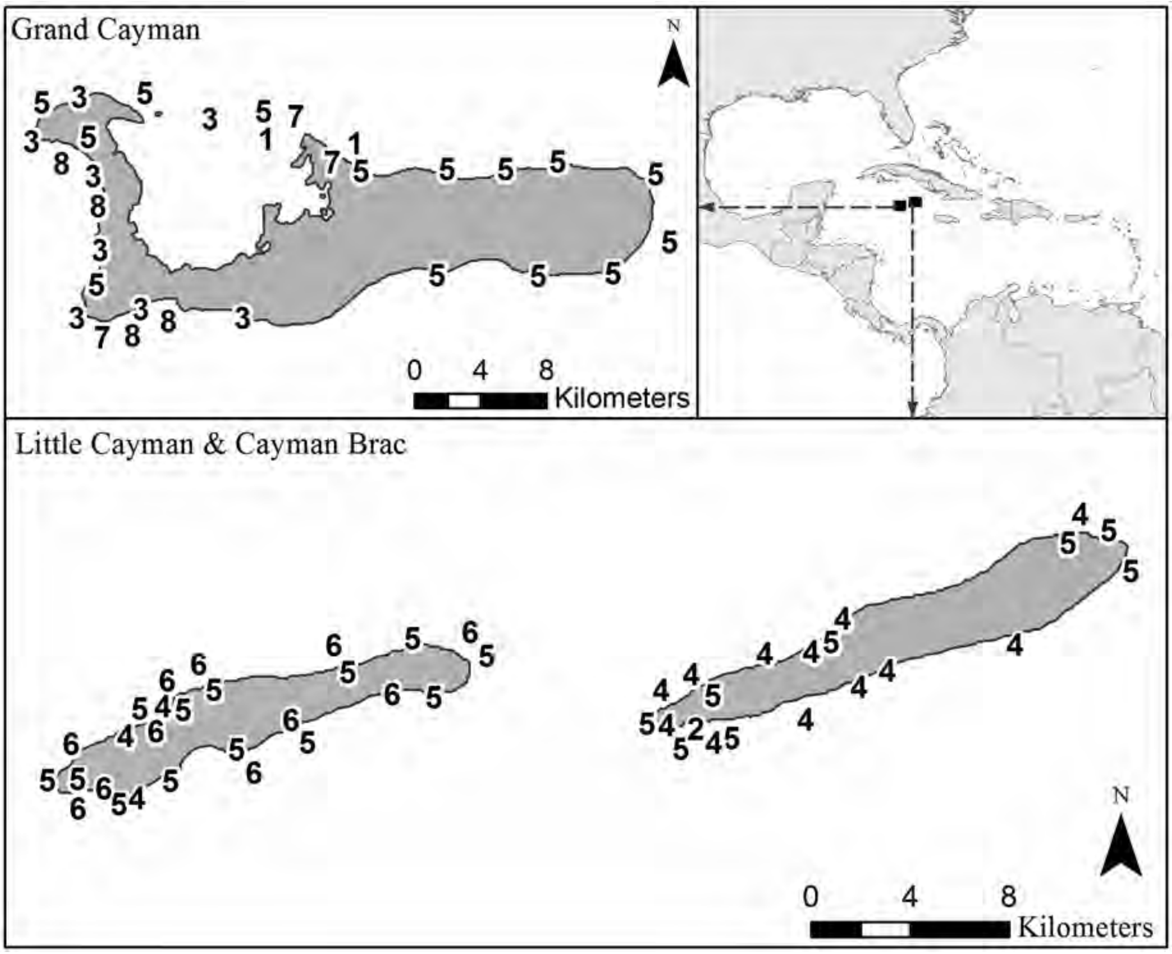

Fig. 8.1 Map of Cayman Islands, codes represent studies listed in Table 8.1. Missing map code(s) due to unavailable coordinates.

Table 8.1 Data sources from Cayman Islands used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 8.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Croy, McCoy; Bush, Philippe/ CARICOMP*[1]	Grand Cayman	1995, 1997, 2000, 2001	4	X		X	
2	Fenner, Douglas*[2]	Little Cayman; Cayman Brac	1988	1	X			
3	Nagelkerken, Ivan*[3]	Grand Cayman	2006	1	X			
4	Miller, Jeff*	Little Cayman; Cayman Brac	1992	1	X		X	
5	Manfrico, Carrie, AGRRA*[4,5]	Grand Cayman;	1999-2000	2	X	X		X
6	Manfrino, Carrie*[6]	Little Cayman; Cayman Brac	1999-2011	13	X	X		
7	Weil, Ernesto*	Grand Cayman	2005, 2006, 2009	3	X		X	
8	Williams, Ivor; Polunin, Nicholas*[7]	Grand Cayman	1997	1	X		X	X
9	Bauer 1980[8]	Grand Cayman	1977	2		X		
a	Reef Check*	Grand Cayman	1997-1998	2		X		

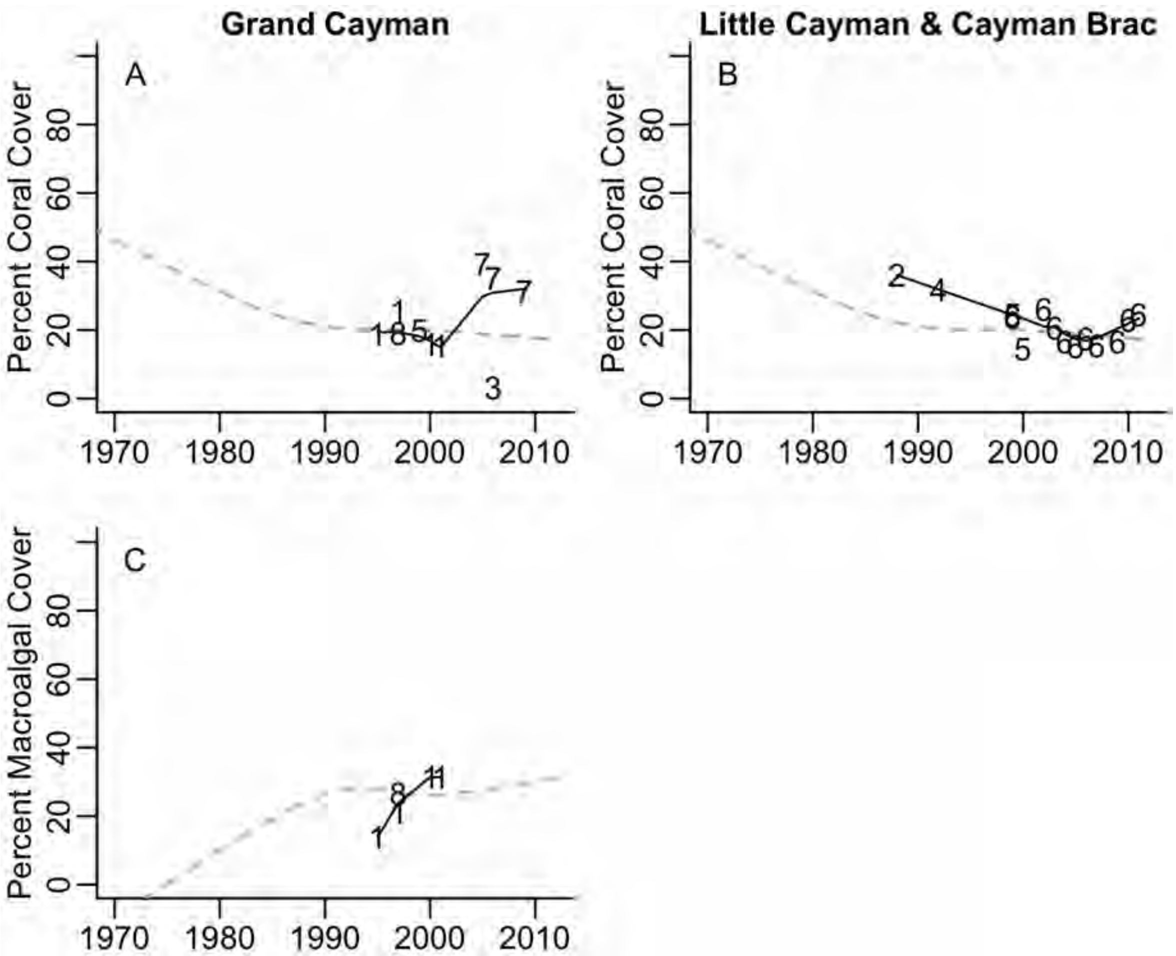

Fig. 8.2 Average percent cover of live corals and macroalgae for Grand Cayman (A & C), and Little Cayman and Cayman Brac (B). Dotted line represents the average of all Caribbean data collected for this report; solid lines are for the data plotted. (Codes same as in Table 8.1 and Figure 8.1)

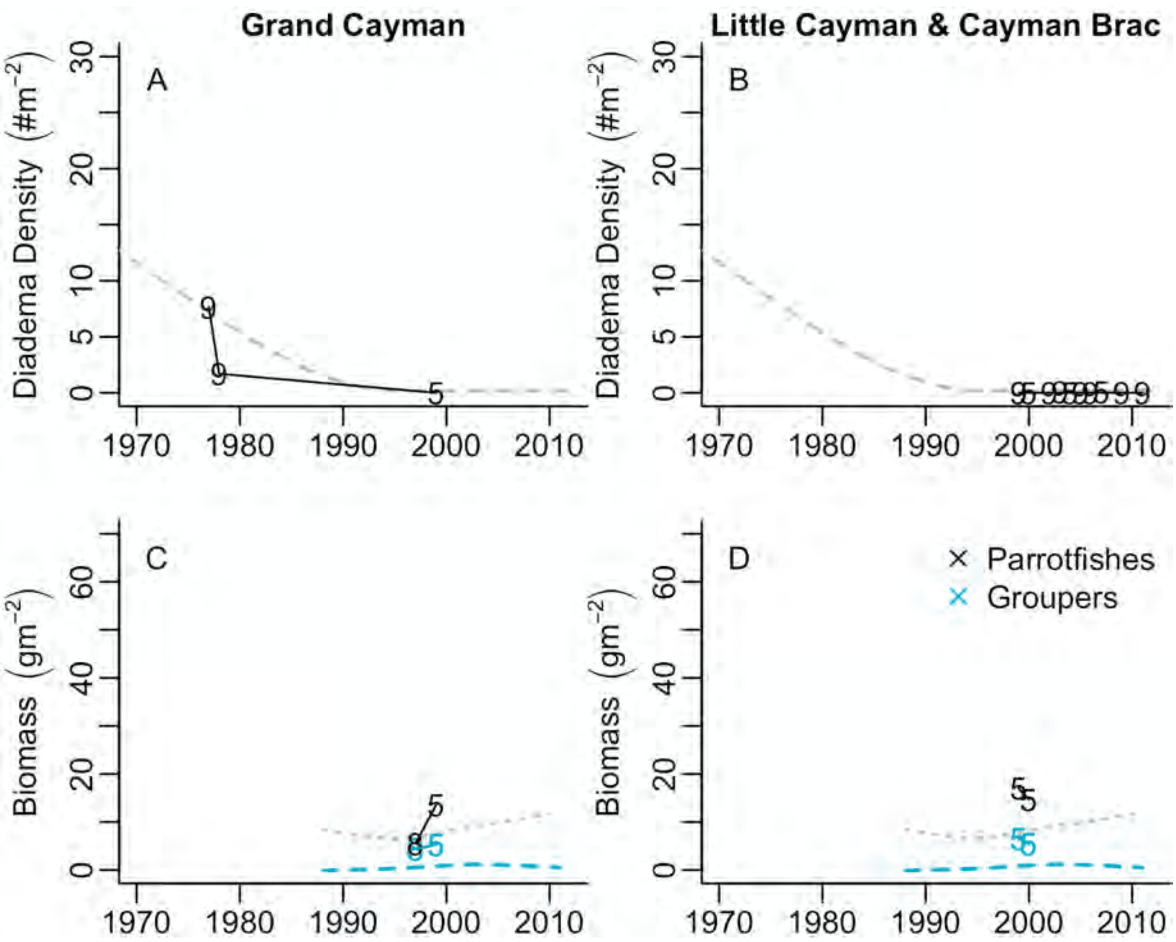

Fig. 8.3 *Diadema* density and biomass of groupers and parrotfishes for Grand Cayman (A & C), and Little Cayman and Cayman Brac (B & D). Dotted line represents the average of all Caribbean data collected for this report; solid lines are for the data plotted. (Codes same as in Table 8.1 and Figure 8.1)

Timeline

1960:	Population ~8,500
1978:	Marine Conservation Law enacted to protect all coral reefs and important fishery species; turtle protection regulations established, prohibiting the harvesting turtle eggs and nesting females during nesting period
1980:	Hurricane Allen (Category 4), heavy damage on *Acropora palmata* and *Acropora cervicornis* in the North coast of Cayman Brac and Little Cayman
1981-82:	White band disease outbreak in Acroporids
1983:	Mass mortality of *Diadema antillarum*
1985:	Turtle protection regulations amended, license system to limit turtle catch
1986:	15% of shelf area designated as marine protected area, with seasonal closures and catch limits for conch and lobster; licensing of spearfishing guns and banning the importation of spear guns
1987-88:	Major bleaching event to depth of 85m
1988:	Hurricane Gilbert (Category 4), hits all 3 islands with heavy damage on *A. palmata* and *A. cervicornis* along the south coast
1990:	2nd mass mortality of *Diadema antillarum*; yellow band disease first documented associated with bleached corals
1994:	371,847 individual scuba dives on Grand Cayman, with some dive sites receiving >16,000 dives per year
1995:	Mass coral bleaching event; tourists exceed 1 million people
1996:	Coral disease outbreaks following 1995 mass bleaching event; strong winter storm destroys substantial amounts of *A. palmata* on west side of Grand Cayman; Massdam cruise ship grounding in Grand Cayman, 1000 m^2 of coral reefs damaged
1998:	Mass coral bleaching event; severe macroalgae overgrowth on reefs; spearfishing ban at designated grouper spawning sites

1999-2000:	Disease outbreaks following 1998 mass bleaching event
2000:	Black band disease outbreak
2001:	Tropical Storm Michelle; Goliath grouper, tile fish, file fish and angelfish added to protected species list; minimum catch size of 20 cm enacted for all fish; licensing of fish traps (maximum 2 per household for Caymanian residents only) and double funnel Antillean Z-fish traps banned
2002:	All echinoderms put on protected species list
2003:	Regulations to protect grouper spawning aggregations, coral bleaching event
2005:	Hurricane Ivan (Category 4) direct hit on Grand Cayman, *Porites porites* along south coast destroyed; coral bleaching event
2008:	Hurricane Paloma (Category 4) direct hit on Cayman Brac and Little Cayman; Hurricane Gustav (Category 2) direct hit on Little Cayman; first documentation of lionfish
2009:	Coral bleaching event; lionfish removal program commenced
2010:	Population >54,000; tourists exceed 1.5 million by sea and 280,000 by air
2012:	>1,500 certified to cull lionfish; >1,000 lionfish caught in a 2 day tournament

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COLOMBIA

Coauthors: Alan Friedlander, Raul Navas, Martha Prada, Alberto Rodríguez-Ramírez, Sven Eloy Zea Sjoberg, CARICOMP, CORALINA, INVEMAR and Reef Check

Geographic Information	
Coastal Length:	5,833 km
Land Area:	1,137,484 km^2
Maritime Area:	816,334 km^2
Reef Area:	1,418 km^2
Number of hurricanes in the past 20 years:	3

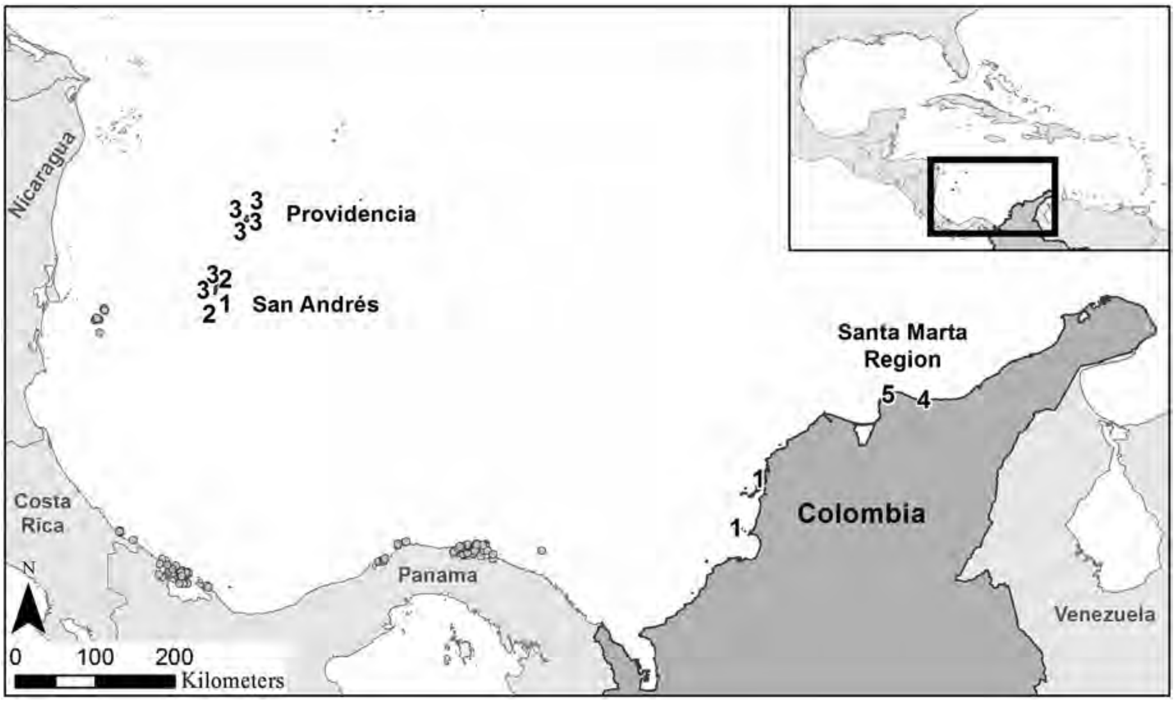

Fig. 9.1 Map of Colombia, codes represent studies listed in Table 9.1. Missing map code(s) due to unavailable coordinates.

Table 9.1 Data sources from Colombia used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 9.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Garzón-Ferreira 1994[1,2,3,4,5,6,7,8]	Santa Marta; San Andrés	1980, 1982-1983, 1987-1990, 1992	9	X		X	
2	CARICOMP*[9]	San Andrés; Providencia	1995-1998, 2000-2005	10	X	X	X	
3	Friedlander, Alan*[10]	San Andrés; Providencia	2000	1	X	X	X	X
4	Liddell & Ohlhorst 1988[11]	Santa Marta	1977	1	X		X	
5	Rodríguez-Ramírez, Alberto*[12]	Santa Marta	1993-2005	13	X		X	
6	Reef Check*	Santa Marta	1997-2006	10		X		

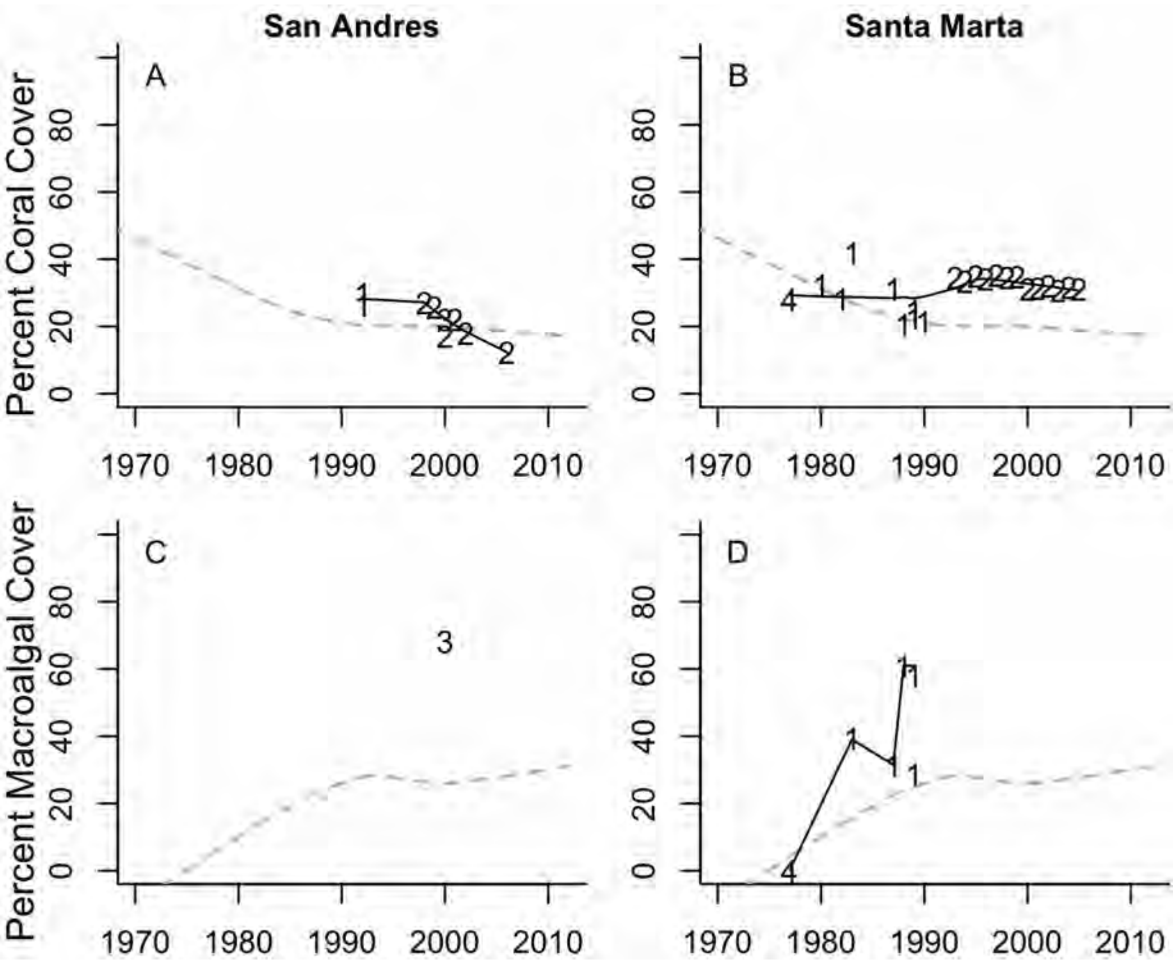

Fig. 9.2 Average percent cover of live corals and macroalgae for San Andrés (A & C), and Santa Marta (B & D). Dotted line represents the average of all Caribbean data collected for this report; solid lines are for the data plotted. (Codes same as in Table 9.1 and Figure 9.1)

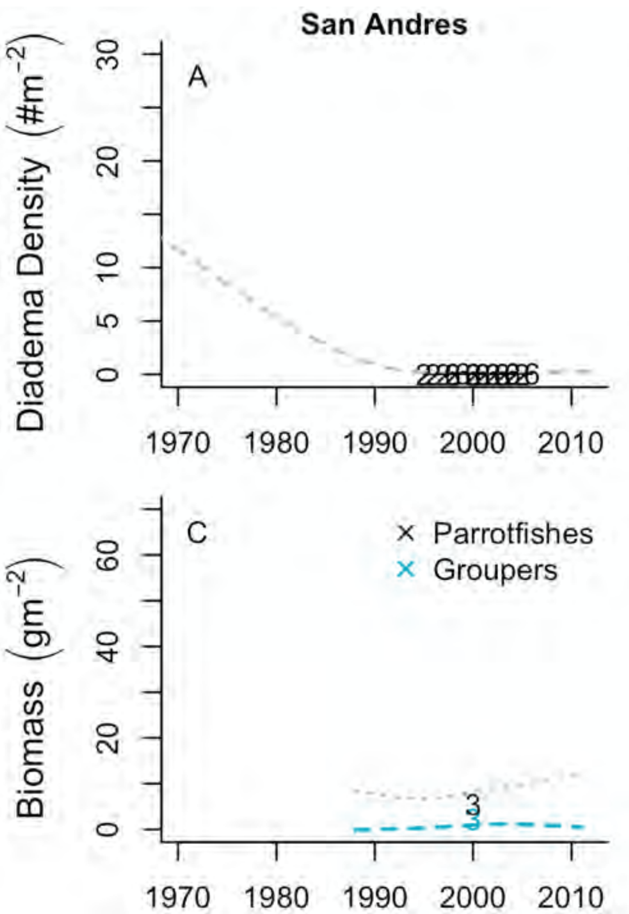

Fig. 9.3 *Diadema* density and biomass of groupers and parrotfishes for San Andrés (A & C), and Santa Marta (B). Dotted line represents the average of all Caribbean data collected for this report; solid lines are for the data plotted. (Codes same as in Table 9.1 and Figure 9.1) Note: Some data available for one year (2000) for Providencia from Friedlander et al. 2003

Timeline

1950-1970s:	No historical baseline of coral reefs condition including fisheries. Field records and underwater photographs suggest that the coral formations around the reef complex of San Andrés Island were healthy (Zea et al. 1998)
1961:	Hurricane Hattie hit Providencia Island, no quantitative data (Geister 1992)
1971:	Hurricane Irene hit San Andres Island, no quantitative data (Zea et al. 1998)
1970-1980:	Overfishing documented (Zea et al. 1998; Díaz et al. 2000)
1970s-1990s:	Extensive coral reduction of 38% in average for all Colombian Caribbean Reefs Areas due to multiples factors (Díaz et al. 2000; Garzón-Ferreira & Kielman 1994)
1980s-1990:	Macrolagal and coral diseases proliferation (Zea et al. 1998; Díaz et al. 2000; Garzón-Ferreira & Díaz 2003)
1982-1983:	Bleaching event, high mortality in *Acropora palmata* at Rosario Islands (Solano et al. 1993)
1983:	Mass mortality of *Diadema antillarum* (in Santa Marta region)
1985-1988.	Mass mortality of *Gorgonia* spp in Tayrona Park and Rosario Islands (Garzón-Ferreira & Zea 1992)
1987:	Bleaching event affecting no more 10% coral cover in Santa Marta and Tayrona Park (Zea & Duque Tobon 1989) and 25% of colonies in Portete Bay (Solano 1994)
1988:	Hurricane Joan hit San Andrés and Providencia reefs. No quantitative data (Zea et al. 1998; Geister 1992)
1990	Bleaching event affecting 10% of colonies in Rosario Islands. Low bleaching-induced mortality was observed (Solano et al. 1993)
1996-1999:	Coral reductions up to 10% in Isla Fuerte, Bajo Bushnell (Díaz et al. 2000)
1995:	Bleaching event affecting no more 5% coral cover in Tayrona Park. The proportion of bleached colonies reached up to 49%. Bleaching-induced mortality was observed up to 12% in *M. faveolata* (CARICOMP 1997; Pinzón et al. 1998)

1999:	Hurricane Lenny affected Tayrona reefs, coral reduction up to 4% (Rodríguez-Ramírez & Garzón-Ferreira 2003)
1998-2004:	No major changes with short-term stability in coral and algal covers; major threats to coral reefs are coral bleaching and overfishing (Garzón-Ferreira 2000; Rodríguez-Ramírez et al. 2010; Rodríguez-Ramírez et al. 2010)
2005:	The 2005 bleaching event was the most severe for the Colombian Caribbean in the last 25 years, affecting 0.5-80 % coral cover in 137 study sites; low bleaching-induced mortality was observed (Rodríguez-Ramírez et al. 2008)
2005:	Hurricane Beta passed very close to Providencia and Santa Catalina islands as a moderate category 1 hurricane (Rodríguez-Ramírez et al. 2008)
2008:	Lionfish *Pterois volitans* first documented at Providencia (Schofield 2009)
2009:	Lionfish *Pterois volitans* first documented at Tayrona Park (González et al. 2009)
2010:	Bleaching event; bleached colonies ranged between 5-25%; bleaching-induced mortality up to 5% at Tayrona Park (Vega-Sequeda et al. 2011)

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COSTA RICA

Coauthors: Jorge Cortés, Ana C. Fonseca-Escalante, AGRRA and CARICOMP

Geographic Information*

Coastal Length: 1,468 km
(Caribbean coast length: 212 km)
Land Area: 51,100 km^2
Maritime Area: 589,683 km^2
(Caribbean Marine Area: 2,310 km^2)
Reef Area: ~20 km^2
(Caribbean reef area: ~10 km^2)
Number of hurricanes in the past 20 years: 0

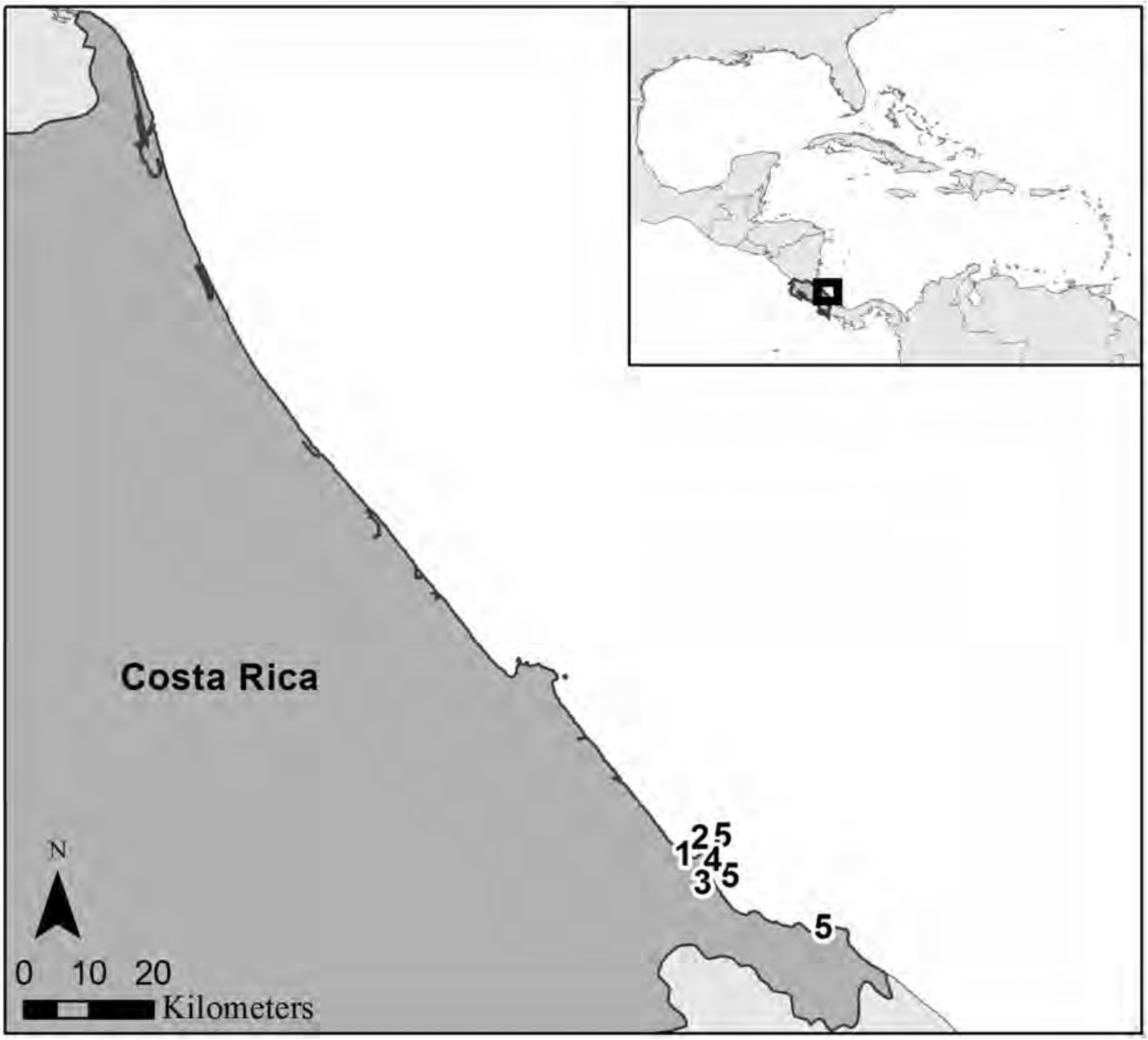

Fig. 10.1 Map of Costa Rica, codes represent studies listed in Table 10.1. Missing map code(s) due to unavailable coordinates.

Table 10.1. Data sources from Costa Rica. Map codes represent individual studies. For exact location of study, refer to Fig. 10.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Alvarado et al. 2004[1]	Cahuita National Park	1980, 1992	2		X		
2	Cortés, Jorge; Fonseca, Ana/ CARICOMP*[2]	Cahuita National Park	1999-2000, 2004-2011	10	X	X	X	
3	Cortés 1981[3], 1994[4]	Cahuita National Park	1980-1981, 1992	3	X		X	
4	Fonseca et al. 2006[5]	Cahuita National Park	2004	1	X	X	X	X
5	Fonseca, Ana/ AGRRA*[6,7]	Cahuita National Park	1999-2000	2	X		X	X
6	Myhre & Acevedo-Gutiérrez 2007[8]	Gandoca-Manzanillo National Wildlife Refuge	2000, 2004	2		X		

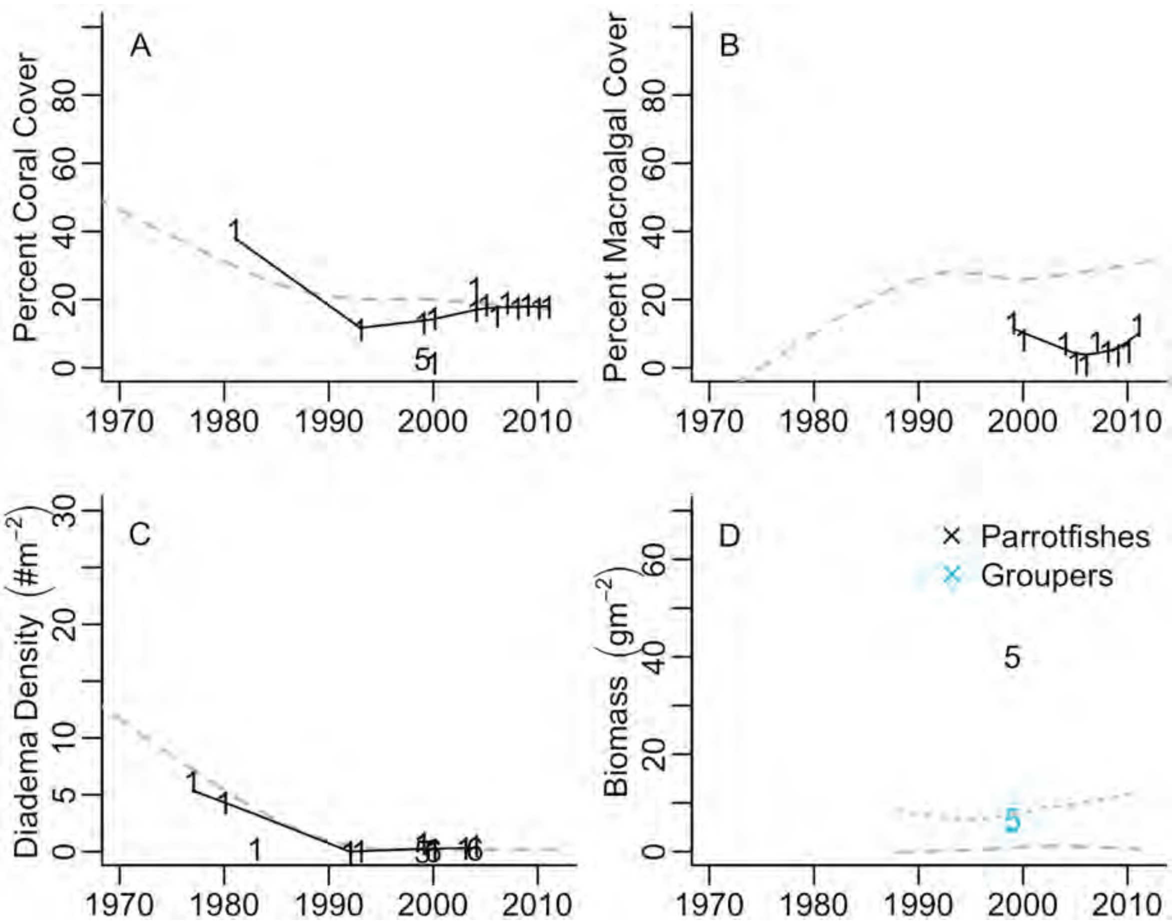

Fig. 10.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C), and biomass of parrotfishes and groupers (D) in Cahuita, Costa Rica. Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 10.1 and Figure 10.1)

Timeline

1970:	First documentation with description of coral reefs in Costa Rica (Caribbean coast), indicates presence of sediments
1970-1980:	Fish traps used on the reefs
1977:	First study of *Diadema antillarum* (3.6-8.8/m^2)
1978:	First scientific publication describing reefs at Cahuita National Park (CNP), impact of sediments is high but high diversity of coral species
1979-81:	Detailed study of reefs at CNP, sediment is the main impact on reefs
1982-83:	Bleaching event
1983:	Mass mortality of *Diadema antillarum*; coral deaths due to high temperature
1984:	Massive die off of sea fans
1991:	Limón Earthquake (7.6 magnitude), uplifted the coast and impacted on reefs and seagrass beds
1992:	2nd mass mortality of *Diadema antillarum*
1995:	Bleaching event; coral mortality due to high temperatures
2000:	Coral cover at CNP at 15%
2003:	*Diadema antillarum* density at 0.3/m^2 at CNP
2004:	Coral cover at CNP at 17%
2007:	First documentation of Lionfish *Pterois volitans*
2008:	Coral cover around 20% at CNP
2000-present:	High sediment loads, heavy fishing pressure, tourist pressure

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CUBA

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Geographic Information*

Coastal Length:	14,385 km
Land Area:	111,089 km^2
Maritime Area:	343,034 km^2
Population:	11,325,600
Reef Area:	4,919 km^2
Number of hurricanes in the past 20 years:	11

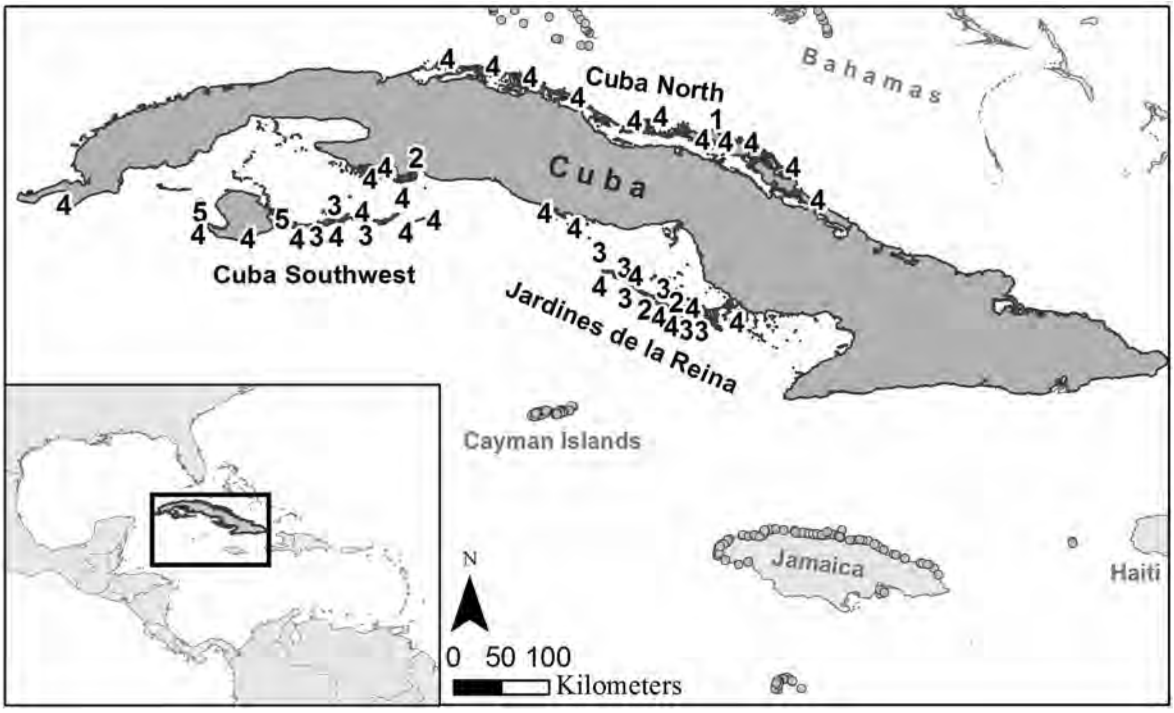

Fig. 11.1 Map of Cuba, codes represent studies listed in Table 11.1. Missing map code(s) due to unavailable coordinates. AGRRA locations are omitted for clarity.

Table 11.1 Data sources from Cuba used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 11.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Alcolado, Pedro/ CARICOMP*[1,2]	North Cuba	1994-1997, 2005	5	X	X	X	
2	Bruno, John*	South west Cuba; Jardines de la Reina	2010-2011	2	X	X	X	X
3	Hardt, Marah; Paredes, Gustavo*[3]	South west Cuba; Jardines de la Reina	2005	1	X		X	X
4	AGRRA*[4,5]	South west Cuba; Jardines de la Reina; North Cuba	1999, 2001	2	X	X		X
5	Polunin, Nicholas; Williams, Ivor*[6]	South west Cuba	1998	1				
6	Claro, Rodolfo*[7,8]	North Cuba; South west Cuba	1984, 1986, 1988, 1989-1991, 2000	7				X
7	Reef Check*		2001-2005	5		X		

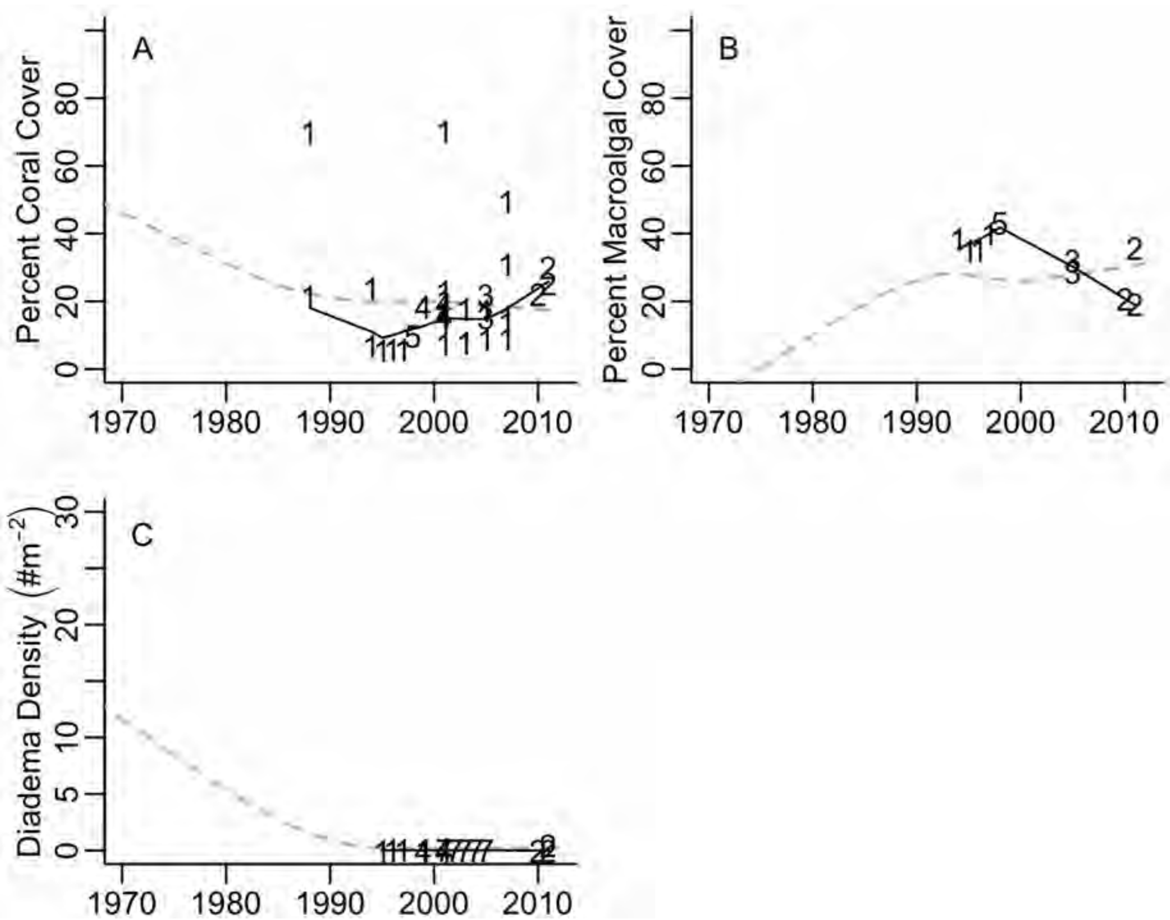

Fig. 11.2 Average percent cover of live corals (A) and macroalgae (B), and density of *Diadema antillarum* (C). Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through the data presented. (Codes same as in Table 11.1 and Figure 11.1)

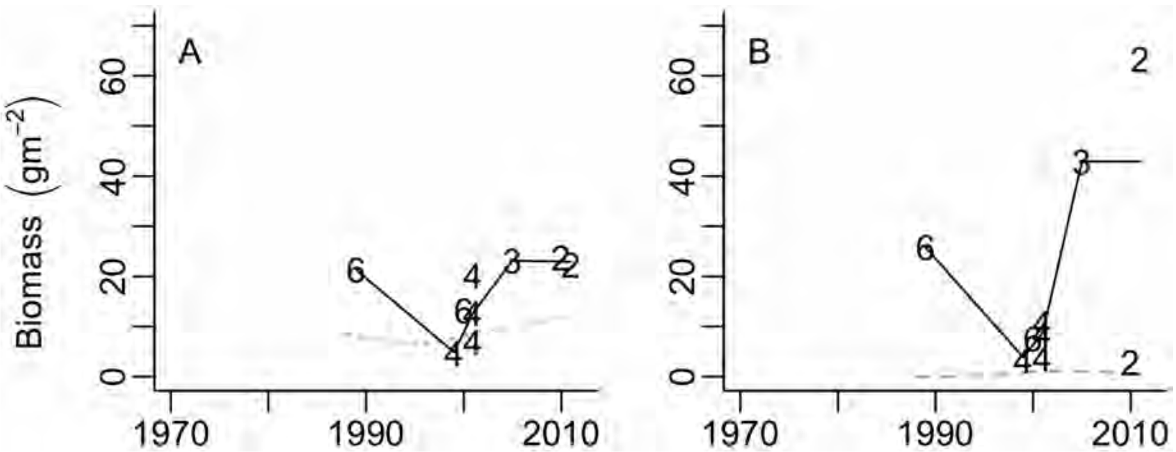

Figure 11.3. Average biomass of parrotfishes (A) and groupers (B) in Cuba. Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through the data presented. (Codes same as in Table 11.1 and Figure 11.1)

Timeline

1980:	Hurricane Allen
1983:	Mass mortality of *Diadema antillarum*; first mass coral bleaching recorded in Cuba
1985:	Hurricane Kate
1988:	Hurricane Gilbert, great damage to *Acropora palmata*
1989:	Bleaching event in northern central Cuba
1990:	Bleaching event in northwest Cuba
1993:	Annual CARICOMP surveys at Cayo Coco initiated; bleaching event in northeast Cuba
1994:	Ministry of Science, Technology and Environment was created leading to improvement of environmental legislation; fisheries law (decree law) approved
1995:	Widespread and intense coral bleaching in north Cuba; National Center for Protected areas established
1996:	Established first explicit regulations for coral reefs, among which collection and using explosives in coral reefs is banned; first marine reserves declared under Fisheries Law (named as Zone Under Special Regime of Use and Protection)
1998:	Severe bleaching event in north Cuba
1999:	Decree-Law on protected areas enacted, facilitating coral reef protection
2000:	White band disease affecting Acroporid reefs; white plague and other diseases affecting other hard corals; Sea fans affected by aspergillosis; Ministry of Fisheries declared 9 'no-take' areas mostly on coral reefs; Decree Law on coastal zone management enacted, facilitating coral reef protection
2001:	Hurricane Michelle; massive outbreak of white plague disease in south and east of Gulf of Batabanó and Jardines de la Reina; first MPAs declared under Protected Areas Law
2005:	Hurricane Dennis (Category 4); bleaching event
2007:	Lionfish first documented in north Cuba; beginning of the integrated coastal zone management process
2008:	Hurricane Ike and Gustav (Category 4); first zones under coastal management declared
2009:	Bleaching event
2010:	Bleaching event
2011:	Low coral disease incidence with the exception for local outbreaks of white plague disease mostly but not only affecting *Dichocenia stokesii*; resolution to control and protect species of special significance of Cuba biological diversity enacted
2012:	Hurricane Sandy; trawling for fish banned

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CURAÇAO

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Geographic Information

Coastal Length:	175 km
Land Area:	444 km^2
Maritime Area:	4,915 km^2
Population:	168,801
Reef Area:	103 km^2
Number of hurricanes in the past 20 years:	0

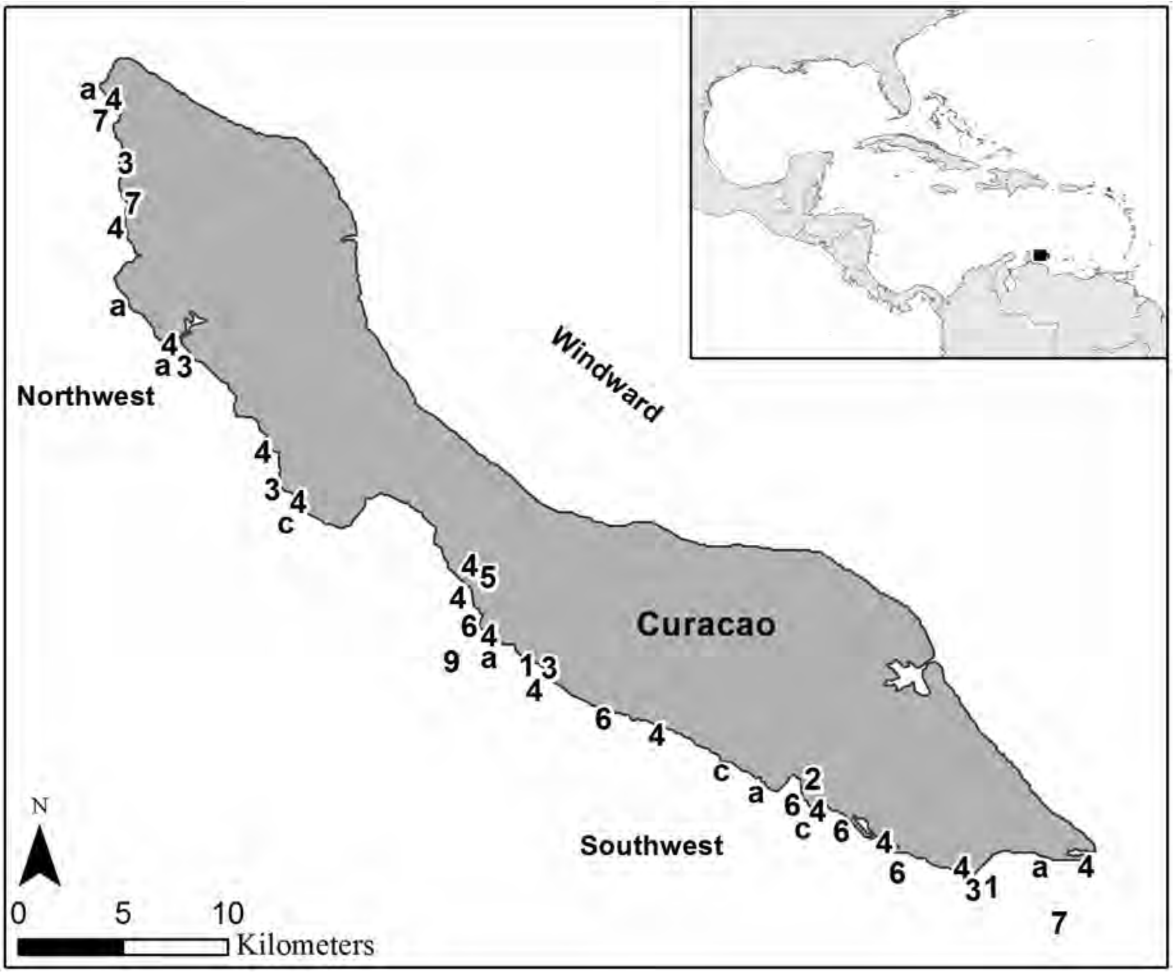

Fig. 12.1 Map of Curaçao, codes represent studies listed in Table 12.1. Missing map code(s) due to unavailable coordinates.

Table 12.1 Data sources from Curaçao. Map codes represent individual studies. For exact location of study, refer to Fig. 12.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Bak, Rolf; Nugues, Maggy; Nieuwland, Gerard; Meesters, Erik*[1,2,3,4,5,6]	SW	1973, 1979, 1983, 1989, 1991-1993, 1997-1998, 2002, 2006, 2008-2009	13	X		X	
2	Nagelkerken, Ivan; Pors, Leon/ CARICOMP*[7]	SW	1994-1995	2	X	X	X	
3	CARMABI report*[8,9,10,11,12,13,14]	SW, NW	1974, 1981, 1983, 2005	4	X	X		
4	Vermeij, Mark*[15]	SW, NW	2003, 2010	2	X	X		
5	Debrot & Nagelkerken 2006[16,17]	SW	2002	1		X		X
6	Nagelkerken, Ivan*	SW	2006	1	X			X
7	AGRRA*[18]	SW, NW	1998, 2000	2	X	X		X
8	Liddell & Ohlhorst 1988[19]	SW	1977	1	X		X	
9	Nagelkerken 2005[20]	SW	1973, 2000	2	X			X
a	Reef Care*	SW, NW	1994, 1997-2008, 2011	13	X		X	
c	Weil, Ernesto*	SW, NW	2005-2006, 2009, 2011	4	X		X	
d	Bak, Rolf 1984[21]	SW	1982-1983	2		X		
e	Bak, Rolf 1975[22]	SW	1974	1		X		
f	Bauer 1980[23]	SW	1977	1		X		
h	Reef Check*		1998, 2000, 2002-2008	9		X		
k	Sandin, Stuart*[24]	SW, NW, windward	2001	1				X

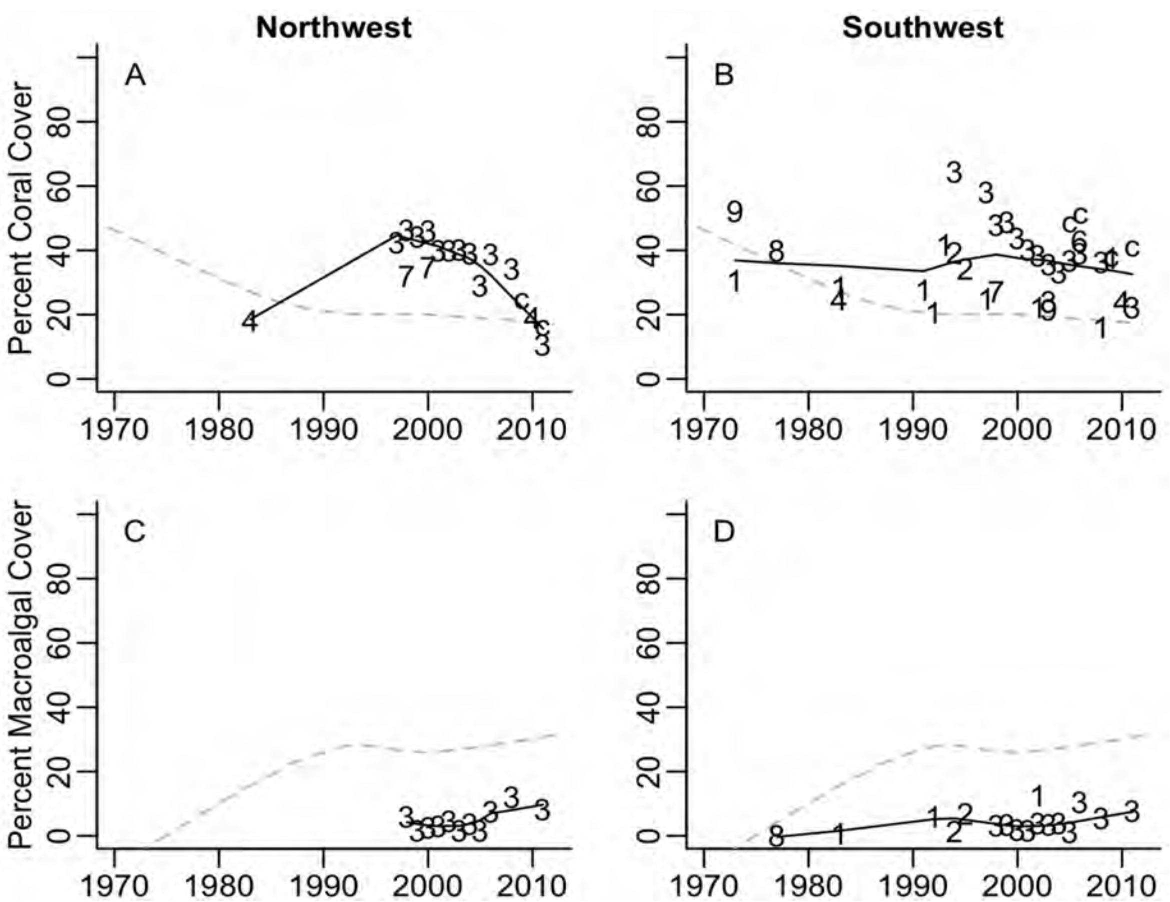

Fig. 12.2 Average percent cover of live corals and macroalgae for two locations in Curaçao: Northwest (north of Kaap St. Marie) (A & C) and Southwest (south of Lighthouse Bullenbaai) (B &D). Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 12.1 and Figure 12.1)

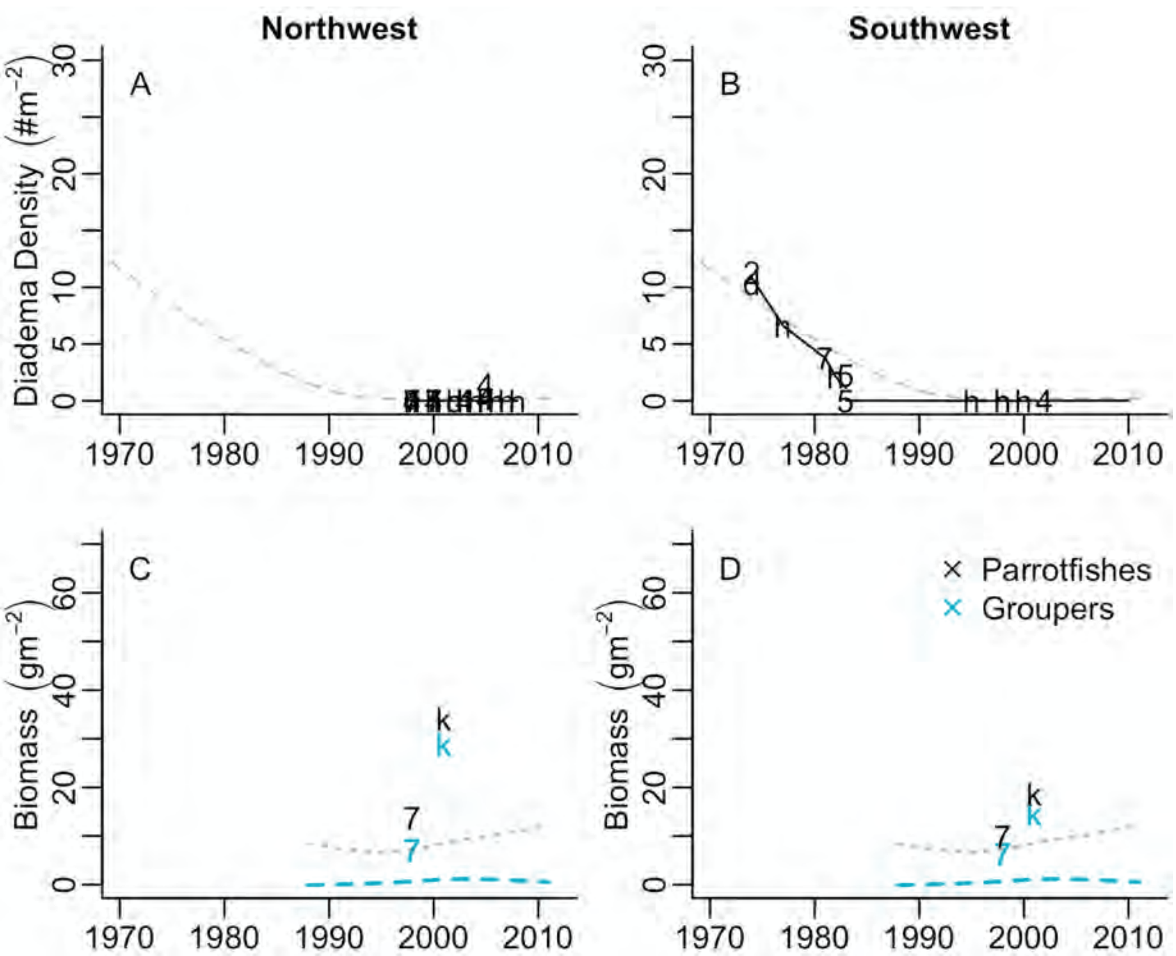

Fig. 12.3 Average density of *Diadema antillarum*, and biomass of parrotfishes and groupers for two locations in Curaçao: Northwest (north of Kaap St. Marie) (A & C) and Southwest (south of Lighthouse Bullenbaai) (B &D). Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 12.1 and Figure 12.1)

Timeline

1916:	Start of oil refinery on Curaçao
1950s:	Synthetic fishing lines introduced
1955:	CARMABI Marine Biological Institute established
1960:	Large fish are commonly seen
1971:	Use of spear guns banned
1975:	Harvesting corals banned
1976:	Spearfishing and harvesting of corals banned
1980:	Joined RAMSAR Convention on Wetlands
1983	White band disease wipes out *Acropora cervicornis*; Curaçao Marine Park established but not enforced; mass mortality of *Diadema*
1986:	Dumping of chemicals and trash in oceans banned
1988:	Tropical Storm Joan
1990s:	Coral cover at deep reefs started to decline
1996:	Catching sea turtles and disturbing nesting sites banned
1997:	30% of island legally designated as conservation habitat by the Curaçao Island Development Plan
1999:	Hurricane Lenny
2007:	Laws established to regulate marine activities, including coastal construction

2009:	Hurricane Omar; ban on gill nets without permit; restrictions on fish and lobster fisheries; first lionfish sighting; population reaches 140,000
2010	Tropical Storm Tomas; bleaching event (10-20% corals bleached); Netherlands Antilles cease to exist, Curaçao became an independent country whilst remaining within the Kingdom of the Netherlands
2012	Official lionfish removals commence
2013	Designation of four RAMSAR areas

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DOMINICAN REPUBLIC

Coauthors: Rodrigo Garza Pérez, Francisco X. Geraldes, Jake Kheel, Patricia Lancho, Yolande Leon, Rúben Torres, AGRRA, CARICOMP and Reef Check

Geographic Information

Coastal Length:	1,610 km
Land Area:	48,257 km^2
Maritime Area:	255,029 km^2
Population:	9,248,710
Reef Area:	838 km^2
Number of hurricanes in the past 20 years:	3

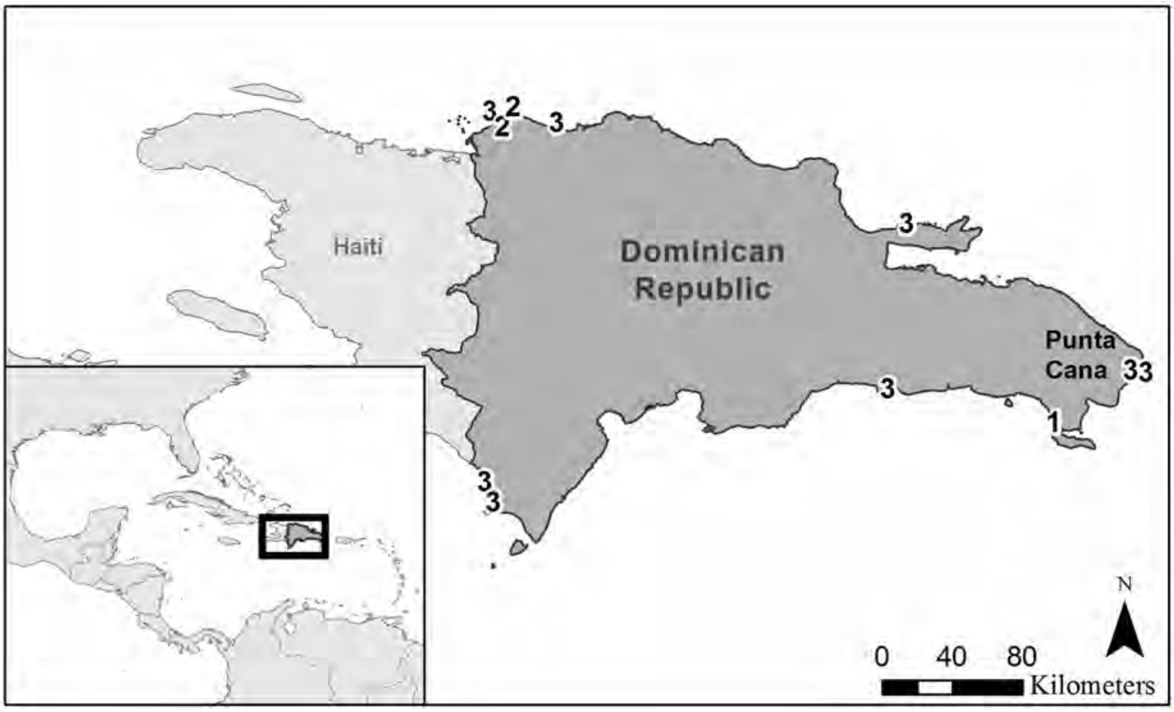

Fig. 14.1 Map of Dominican Republic, codes represent studies listed in Table 14.1. Missing map code(s) due to unavailable coordinates.

Table 14.1 Data sources from Dominican Republic. Map codes represent individual studies. For exact location of study, refer to Fig. 14.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Geraldes, Francisco/ CARICOMP*[1]	1994, 1996-1997, 2000-2001	5	X	X	X	
2	Garza Pérez, Rodrigo*[2,3]	2006	1	X		X	X
3	AGRRA*[4,5]	2003-2004	2	X	X		X
4	Reef Check*	2004-2007	4		X		

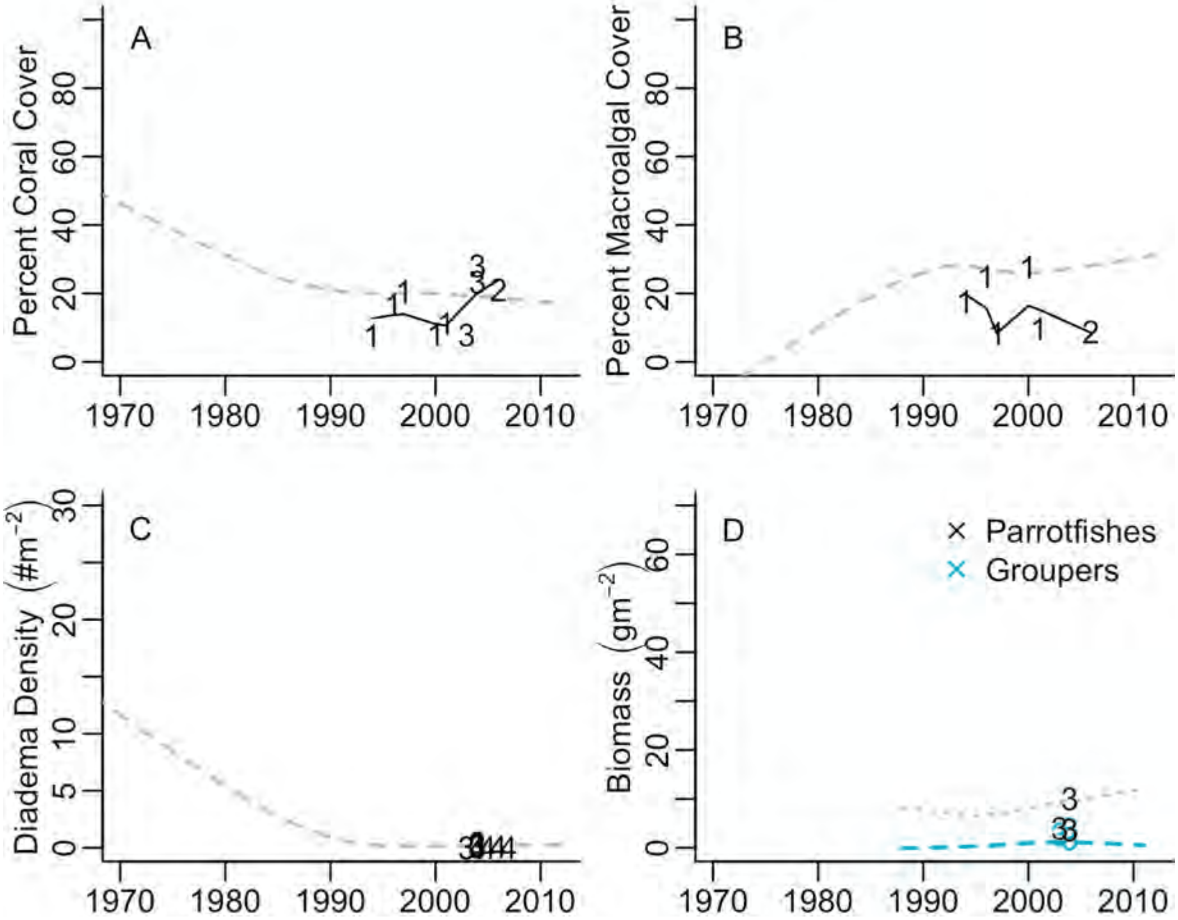

Fig. 14.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C), and biomass of parrotfishes and groupers (D) in Dominican Republic. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through the data presented. (Codes same as in Table 14.1 and Figure 14.1)

Timeline

1950s-1970s:	Reefs appeared healthy, coral cover and benthic density high, relatively few macroalgae (Francisco Geraldes, pers. comm.)
1970-1974:	Artisanal fisheries increased from few fishing boats to ~400 operating along the coast with main fishing grounds associated with reefs
1974:	Reefs showed reduction of large fishes, coral colonies damaged by divers and anchors; presidential decree banned the collection of corals
1974-1980s:	Development of tourism, mainly in Puerto Plata in the north coast and Boca Chica to Guayacanes in the south coast; high fishing pressure, targeting grouper, lobsters, conch and occasionally turtles and sharks
1979:	Hurricane David (Category 5) affected south coast, no bleaching detected
1980:	Hurricane Allen (Category 5), produced large waves on the south coast (website)
1980-1983:	Increase in fishing activities, landings exceeded 10,000 metric tons/year in fish and shellfish products for local and export
1981:	Tropical storm Gert passed through the northwest
1982:	Tropical storm Derby
1983:	Mass mortality of *Diadema* antillarum; Montecristi National Park established
1986:	Banco de la Plata marine sanctuary; La Caleta Underwater Marine Park created; inclusion of marine and coastal areas up to 30m depth in Parque Nacional del Este and Jaragua; Montecristi National Park boundaries defined; regulations on fishing gear; establishment of no fishing areas but lack of funding prevented full implementation; National Aquarium built
1987:	Hurricane Emily
1987-1988:	Major coral bleaching event, affecting reefs up to 85m in depth

1988: Hurricane Gilbert (Category 5); large grouper and parrotfish are rare

1989: Regulations passed to protect grouper spawning aggregations and restrict conch harvest

1990: Mass mortality of *Diadema antillarum*; yellow band disease first documented associated with bleached corals

1995: Number of tourists exceed 1 million

1996: Hurricane Hortense

1993: Boundaries of Montecristi National Park significantly increased

1998: Hurricane George (Category 3); severe macroalgae overgrowth on coral reefs

2000: Tropical storm Derby affects north coast; black band disease epizootics; Dominican Republic Environmental Law issued by Congress; Montecristi National Park ratified

2001: Increasing pressure on reefs from diving tourism

2003: Tropical storm Odette

2004: Hurricane Jeanne (Category 1); AGRRA conducts surveys on 5 reefs

2005: Montecristi National Park reef assessment and characterization (Garza Pérez & Ginsburg 2009)

2007: Hurricane Dean (Category 4)

2008: Tropical storms Noel and Olga

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DOMINICA

Coauthor: MACC

Geographic Information

Coastal Length:	149 km
Land Area:	765 km^2
Maritime Area:	28,593 km^2
Population:	76,017
Reef Area:	49 km^2
Number of hurricanes in the past 20 years:	1

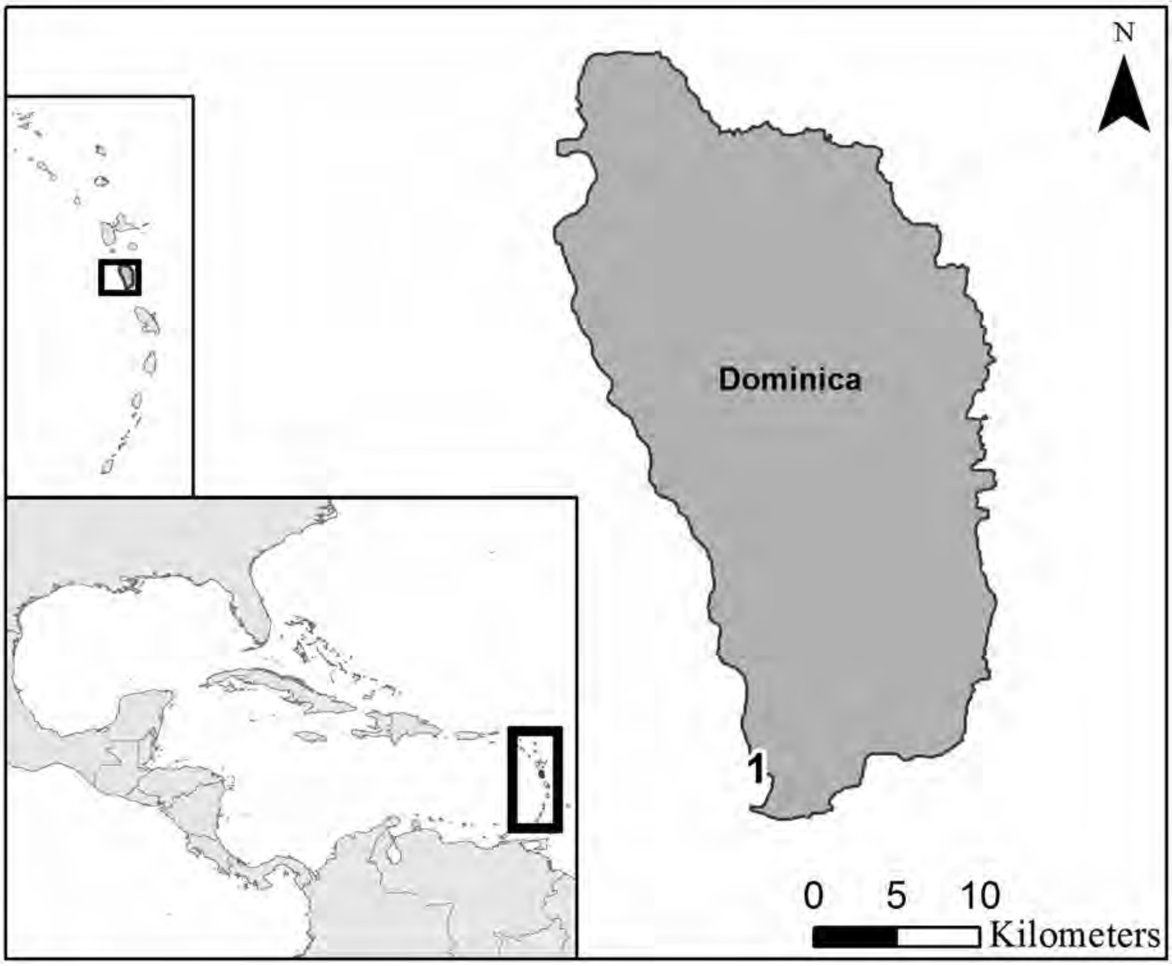

Fig. 13.1 Map of Dominica, codes represent studies listed in Table 13.1. Missing map code(s) due to unavailable coordinates.

Table 13.1 Data sources from Dominica. Map codes represent individual studies. For exact location of study, refer to Fig. 13.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	MACC*[1,2]	2007, 2009	2	X		X	
2	Steiner & Kerr 2008[3]	2005-2006	2	X			
3	Steiner & Williams 2006[4,5]	2001-2005	5		X		

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FLORIDA REEF TRACT

Coauthors: Andrea Atkinson, Bill Alevizon, Jerry Ault, Chris Caldow, Billy Causey, Mark Chiappone, Mike Colella, Phil Dustan, David Gilliam, Ben Greenstein, Marah Hardt, Walter Jaap, Karen Lukas, Steven Miller, John Pandolfi, Gustavo Paredes, Ben Ruttenberg, Rob Ruzicka, Ernesto Weil, AGRRA, CARICOMP, FWC, NOAA SEFSC, National Park Service South Florida/ Caribbean Network (NPS/SFCN) and USEPA

Geographic Information

Coastal Length:	11,347 km
Land Area:	147,906 km^2
Maritime Area:	460,994 km^2
Reef Area:	1,179 km^2
Number of hurricanes in the past 20 years:	7

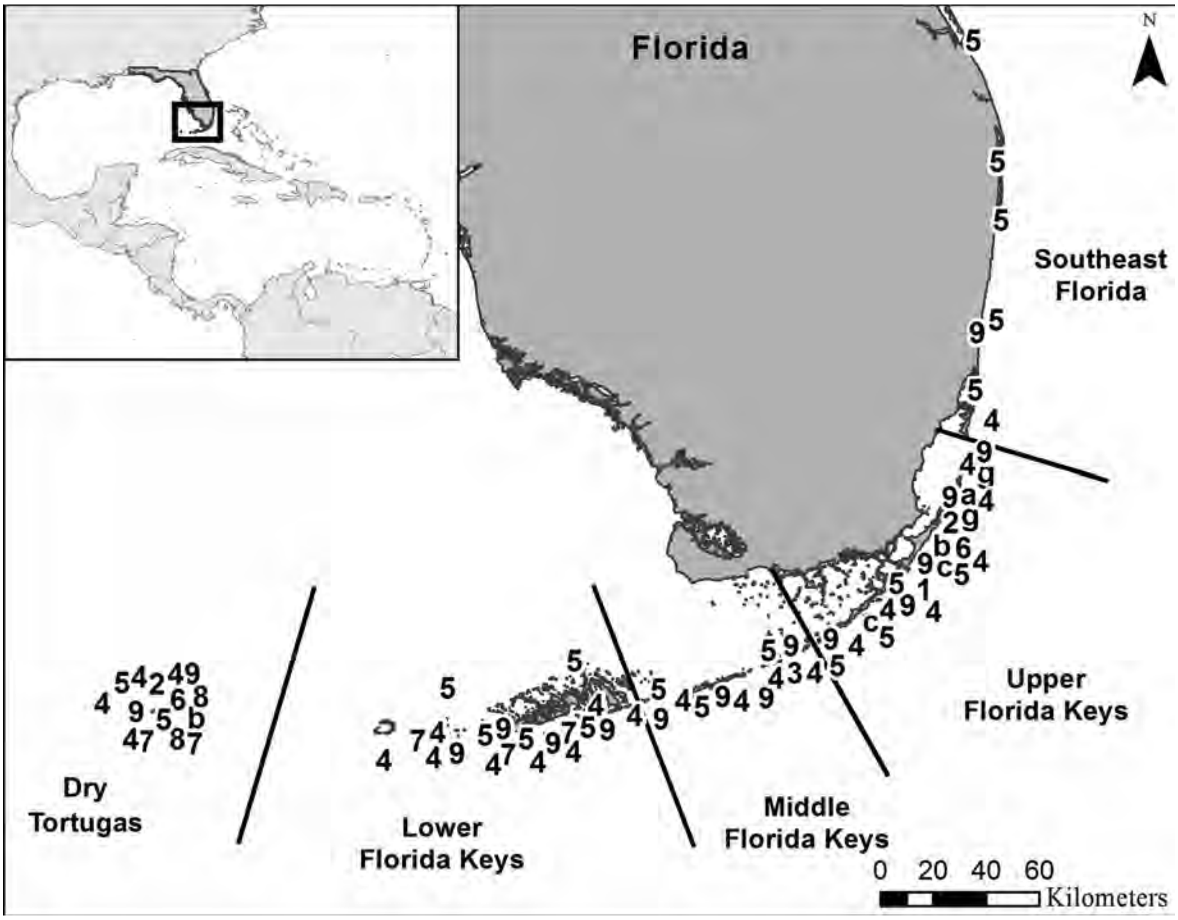

Fig. 15.1 Map of Florida, codes represent studies listed in Table 15.1. Missing map code(s) due to unavailable coordinates.

Table 15.1 Data sources from Florida. Map codes represent individual studies. For exact location of study, refer to Fig. 15.1; * denotes original data; for full references, refer to published literature sources in the last section. UK = Upper Keys, MK = Middle Keys, LK = Lower Keys, DT = Dry Tortugas

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Alevizon, Bill*[1,2]	UK	1974, 2000	2	X		X	X
2	National Park Service/SFCN*	UK, DT	2004-2011	8	X	X	X	
3	CARICOMP*	MK	2001-2002, 2004	3	X		X	
4	Chiappone, Mark; Miller, Steven*[3,4,5,6,7,8]	All	1999-2002, 2005-2011	11	X	X	X	
5	SECREMP & CREMP; Florida Fish & Wildlife Research Institute (FWC FWRI), NOAA CRCP, FLDEP CRCP, NSUOC*[9,26]	All	1996-2011	16	X	X	X	
6	Dustan, Phil*[10,11,12]	UK, DT	1975, 1982-1983	3	X			
7	Hardt, Marah; Paredes, Gustavo*[13]	LK, DT	2005	1	X	X		X
8	Jaap, Walter*[14,15,16,17]	DT	1975-1976, 1989-1991	5	X			
9	AGRRA*	All	1999, 2003-2004, 2006	4	X	X		X
a	Lirman & Fong 1997[18]	UK	1993-1994, 1996	3	X			
b	Lukas, Karen*	UK, DT	1975	1			X	
c	Greenstein, Ben; Pandolfi, John*[19,20]	UK	1994, 1996	2	X			
d	Porter & Meier 1992[21]	UK, SE Florida	1984-1986, 1988-1991	7	X			
e	Porter et al. 1982[22]	DT	1976-1977	2	X			
f	Weil, Ernesto*	UK, SE Florida	1994	1	X		X	
g	NOAA SEFSC*	All	1980-2011	31				X
h	Kissling 1977[23]	LK	1972-1973	2		X		
i	Bauer 1980[24]	UK	1977-1978	2		X		
j	Forcucci 1994[25]	UK, MK, LK	1991	1		X		

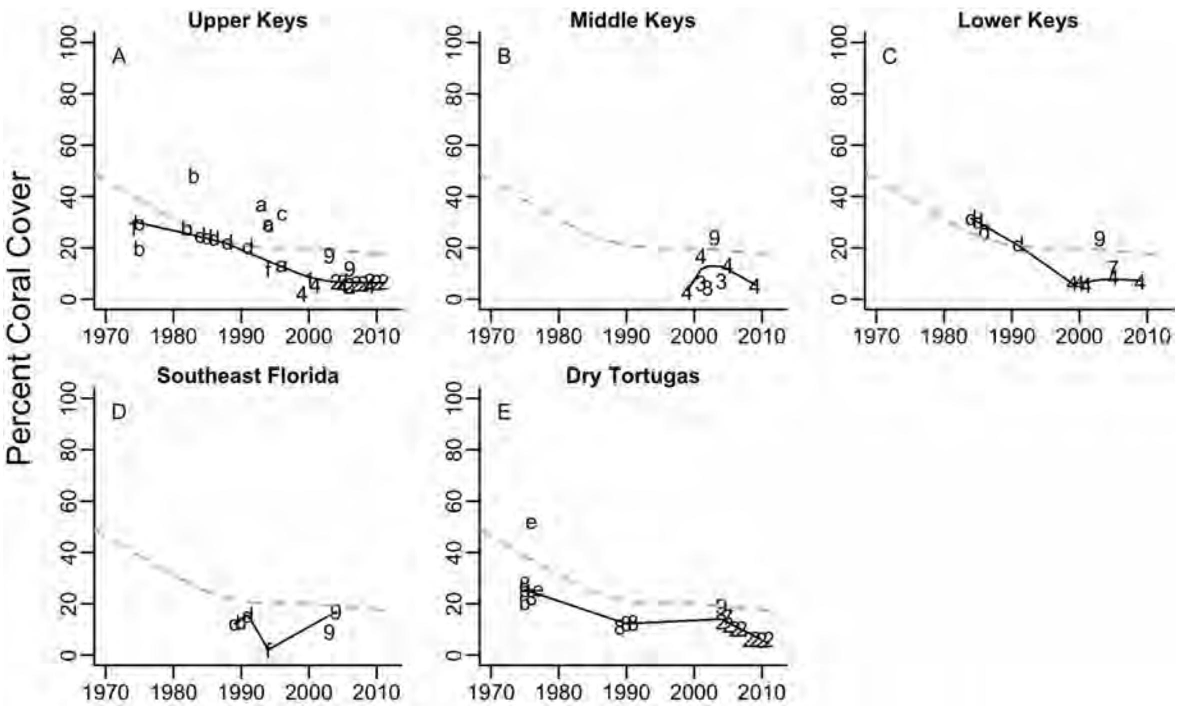

Fig. 15.2 Average percent cover of live coral for 5 regions of the Florida reef track including Upper Keys (A), Middle Keys (B), Lower Keys (C), Southeast Florida (north of Biscayne Bay; D) and Dry Tortugas (E). Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 15.1 and Figure 15.1)

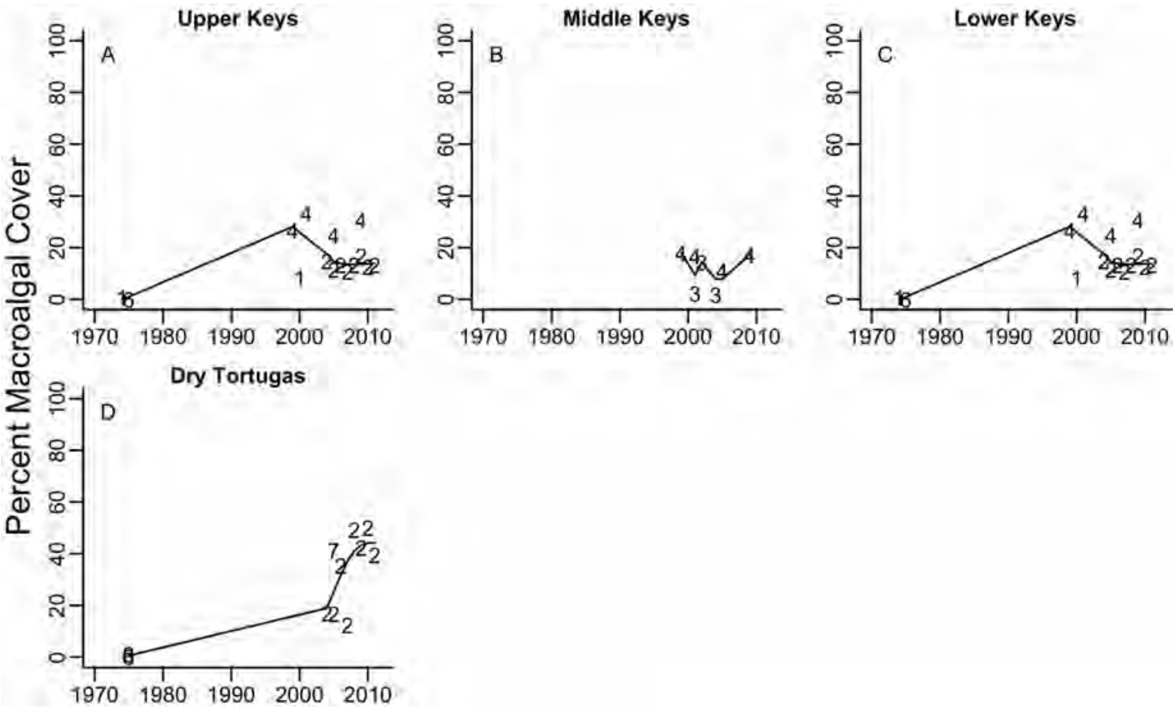

Fig. 15.3 Average percent cover of macroalgae for 4 regions of the Florida reef track including Upper Keys (A), Middle Keys (B), Lower Keys (C), Dry Tortugas (D). Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 15.1 and Figure 15.1)

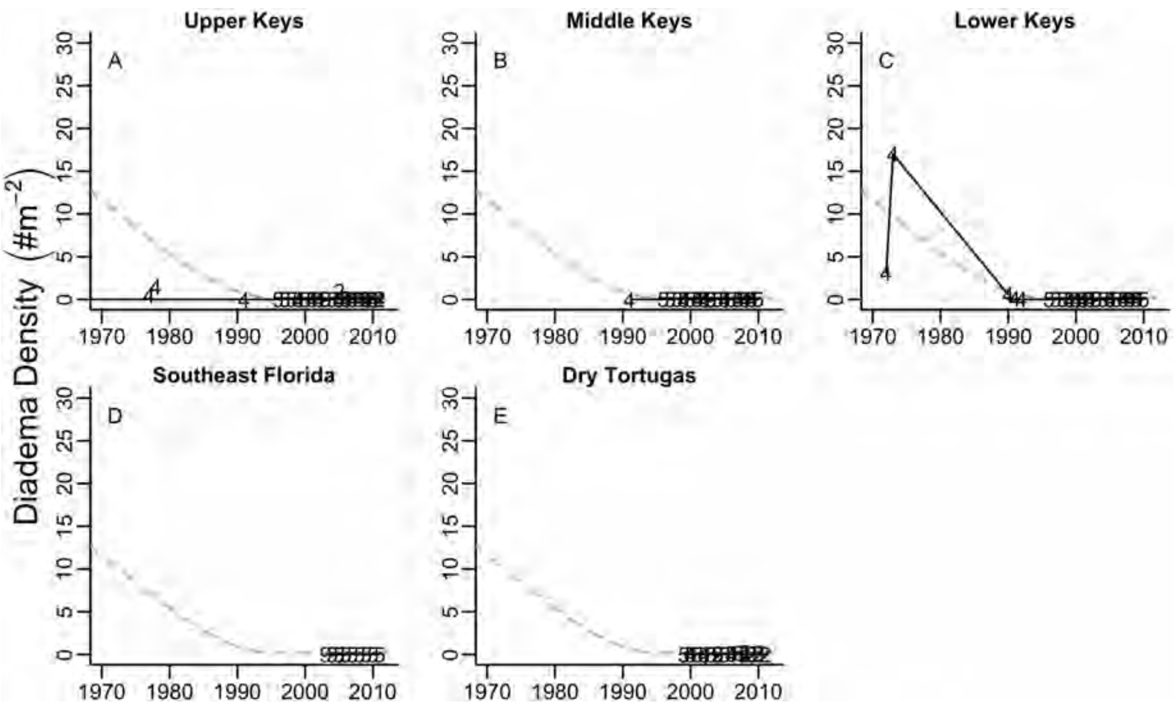

Fig. 15.4 Average density of *Diadema antillarum* for 5 regions of the Florida reef track including Upper Keys (A), Middle Keys (B), Lower Keys (C), Southeast Florida (north of Biscayne Bay; D) and Dry Tortugas (E). Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 1 and Figure 1)

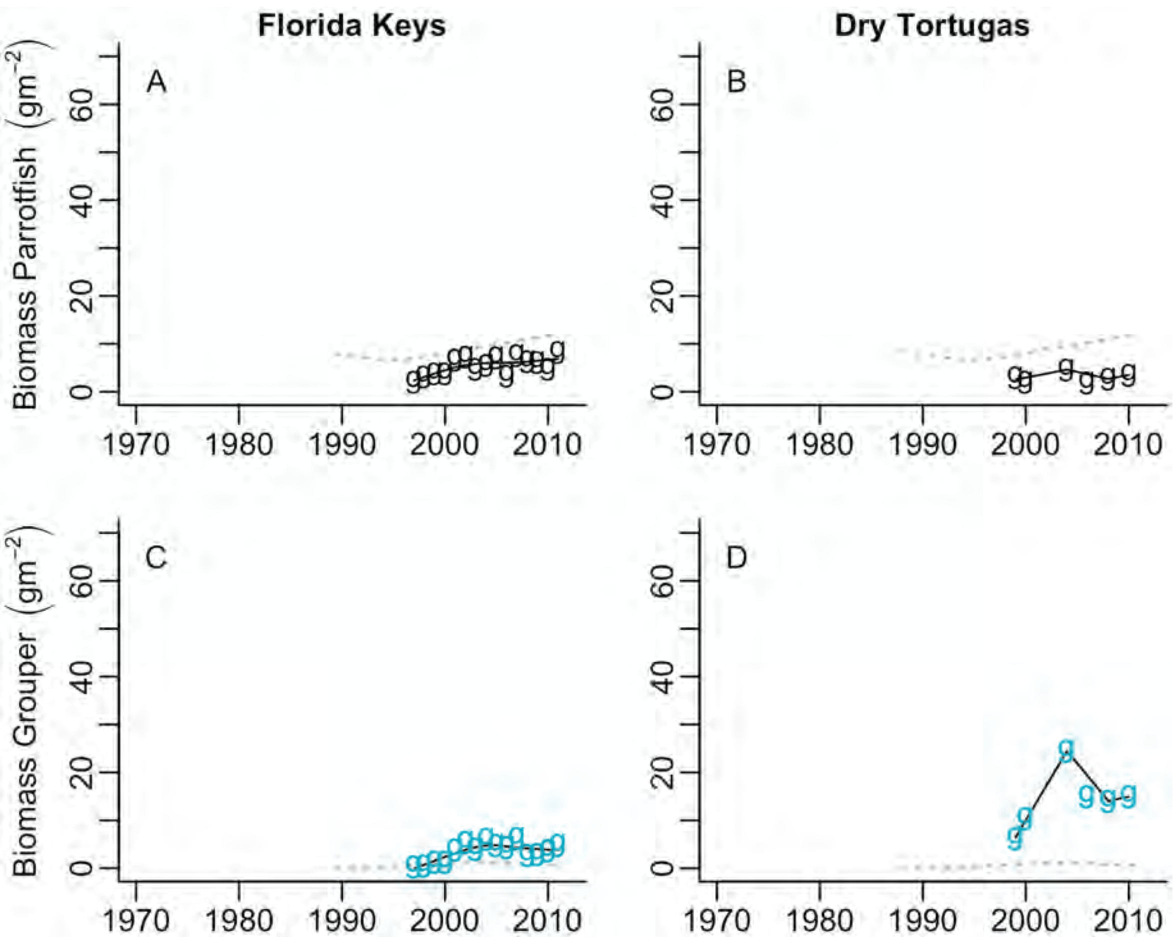

Fig. 15.5 Average biomass of parrotfishes and groupers for 2 regions of the Florida reef track including (A,C) Florida Keys domain wide; and (B,D) Dry Tortugas. Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 1 and Figure 1)

Timeline

1878:	Black water event kills *Acropora* in Dry Tortugas
1902:	Miami harbor channel dredging
1928:	Port Everglades (Broward County) opens
1935:	Dry Tortugas National Monument established by President Roosevelt
1950s:	Sportfishing becomes popular
1960:	Spearfishing banned in Pennekamp Park; Hurricane Donna (Category 4); recovery study by Shinn shows recovery in 5 years
1965:	Hurricane Betsy (Category 3)
1968:	Biscayne National Monument established
1972:	Black Band Disease described/discovered
1974:	Bleaching event at Middle Sambo Reef; Maya trimiran wreck on Key Largo Dry Rocks
1975:	Key Largo National Marine Sanctuary established; mild bleaching event; white plague (Type I) first identified killing *Colpophyllia, Helioseris, Mycetophyllia* and *Montastraea annularis* (Published in 1978 but observed in 1974-1975)
1977:	Cold water-event extirpates nearly all *Acropora* at Dry Tortugas; White Plague (Type I) first identified killing *Colpophyllia, Leptoseris, Mycetophyllia* and *Montastraea annularis*
1980s:	White band disease breakout
1980:	Fort Jefferson National Monument and Biscayne National Park established; fish traps banned in state waters
1981:	Looe Key National Marine Sanctuary established (no spearfishing allowed)

1983:	Mass mortality of *Diadema*; epidemic bleaching in Lower Keys
1984:	Black band disease; mooring buoy installation begins in Key Largo by John Halas
1985:	Bleaching event; total state ban on conch harvest
1987:	Widespread bleaching event; Gulf Council bans new fish traps and 10 year phase out of existing traps
1988:	South Atlantic Fishery Council outlawed fish traps
1990:	Drought; bleaching event with little mortality; Florida Keys National Marine Sanctuary (FKNMS) established; Goliath grouper fishing banned
1991:	Second *Diadema antillarum* dieoff; Establishment of Marine Life Rule (1991): prohibiting the take of parrotfishes >12cm for food consumption
1992:	Dry Tortugas National Park established; replaces Fort Jefferson National Monument; Hurricane Andrew (Category 5, 17-foot storm surge); water quality protection program for Florida Keys National Marine Sanctuary Program mandates monitoring of water quality, seagrasses and coral reefs
1994:	Yellow band disease first identified in the Florida Keys; massive release of freshwater by the Army Corps of Engineers containing increased nitrogen levels; commercial entanglements net larger than 500 square feet outlawed
1995:	White Plague (Type II) identified
1996:	Water Quality Protection Plan (WQPP) initiated; bleaching event
1997:	White band disease 2 outbreak, pathogen linked to sewage identified; bleaching event
1998:	Major bleaching event and major increase in geographic distribution of diseases (CRMP)
1999:	Law that requires all sewage facilities in Florida Keys upgraded to conform BAT and AWT (stringent) standards by 2010, later extended to 2015
2000:	White plague (Type III) identified
2001:	Ecological Reserves (no take areas) in FKNMS established
2002:	Harmful algal bloom/blackwater event along SW Florida shelf
2004:	Hurricanes Charley (Category 4), Frances (Category 3) and Jeanne (Category 2-3)
2005:	Hurricanes Dennis, Katrina, Rita and Wilma
2007:	Dry Tortugas National Park Research Natural Area established (no take and no anchoring)
2009:	Florida Coral Reef Protection Act
2010:	Cold water bleaching and mortality event; Deepwater Horizon oil spill
2012:	Hurricane Sandy

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FLOWER GARDEN BANKS

Coauthors: Robert Edward Abbott, Thomas Bright, Chris Caldow, Randy Clark, Steve Gittings, Emma Hickerson, AGRRA, Flower Garden Banks National Marine Sanctuary (FGBNMS), Office of National Marine Sanctuaries (ONMS) and NOAA Biogeography Branch

Geographic Information

Number of hurricanes in the past 20 years:	1

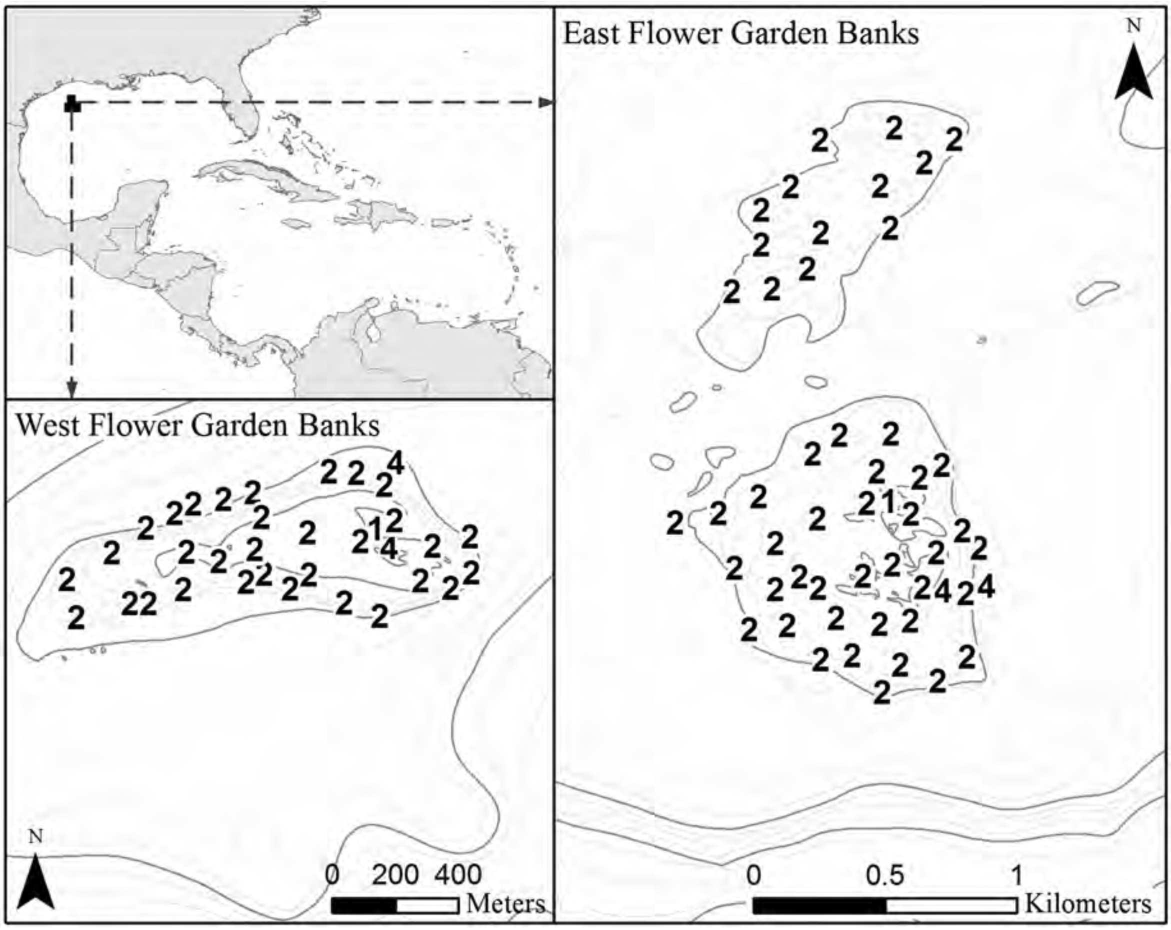

Fig. 16.1 Map of Flower Garden Banks, codes represent studies listed in Table 16.1. Grey lines represent bathymetric contours. Missing map code(s) due to unavailable coordinates.

Table 16.1 Data sources from Flower Garden Banks. Map codes represent individual studies. For exact location of study, refer to Fig. 16.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroal-gae	Fishes
1	Bright et al. 1984[1]	1974-1980 average		X			
2	NOAA Biogeography Branch*[2,3]	2006-2007, 2010-2011	4	X	X	X	
3	AGRRA*[4]	1999	1	X	X		X
4	Gittings, Steve; Hickerson, Emma/ FGBNMS/MMS-BOEM*	1978-1983, 1988-1991, 1994-2010	17	X	X	X	

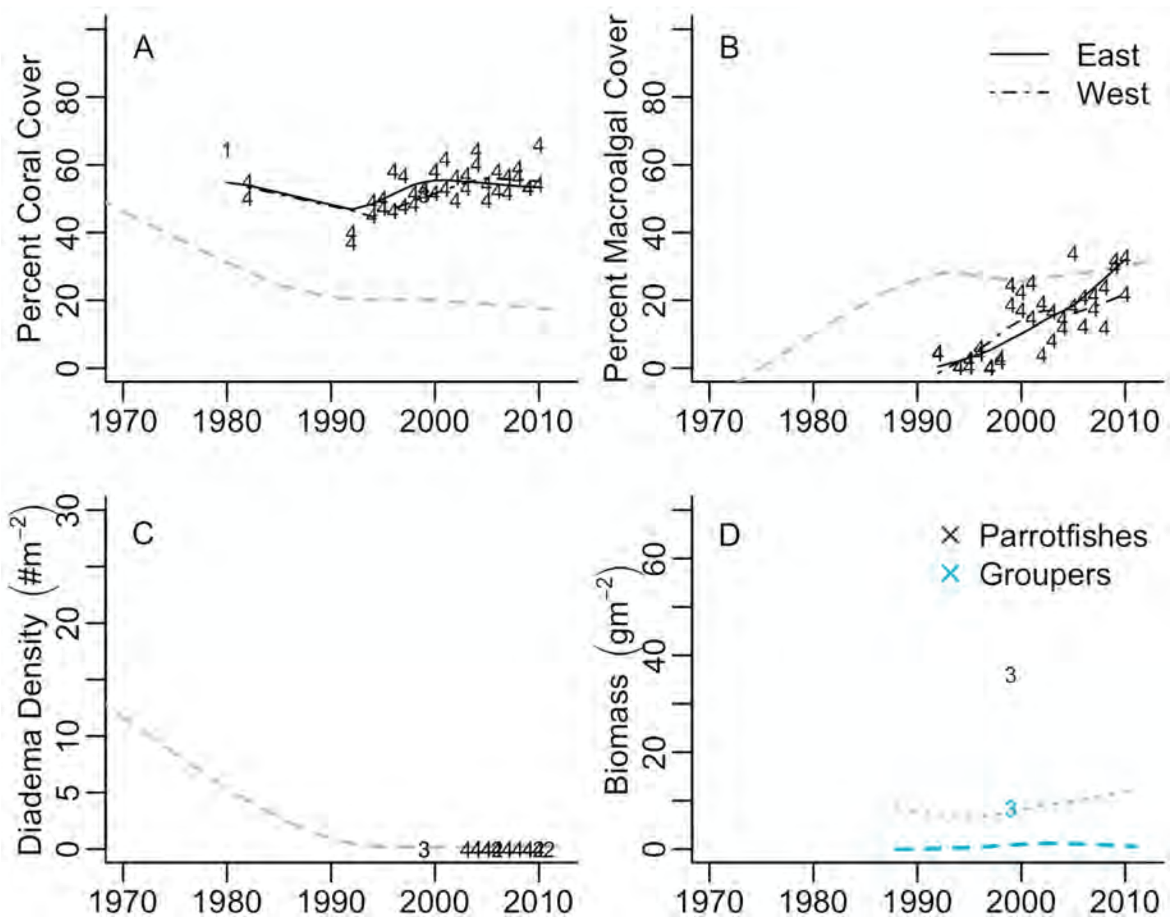

Fig. 16.2 Average percent cover of live corals (A) and macroalgae (B) for the East and West Flower Garden Banks, and density of *Diadema antillarum* (C), and biomass of groupers and parrotfishes for the Flower Garden Banks combined (D). Dotted grey line represents the average of Caribbean data collected for this report; black lines are drawn through data presented. (Codes same as in Table 16.1 and Figure 16.1)

Timeline

~1900:	Snapper fishermen discover Flower Garden Banks
1936:	First recorded discovery of banks
1960:	First diving exploration of FGB
1972:	Earliest quantitative benthic assessment, indicating live hard coral cover of nearly 50%
1972-1982:	Extensive diving and submersible surveys on banks
1976:	East Flower Garden Bank Brine Seep discovered
1977:	Hurricanes Anita and Babe, heavy surge impacted reefs in East Flower Garden; bleaching event followed (Abbott 1979)
1980:	Hurricane Allen (Category 5)

1983-1984:	Mass mortality of *Diadema antillarum*
1985:	Recreational dive charters begin
1988:	Current long term monitoring protocols initiated
1990:	Mass spawning of Atlantic corals first observed on banks
1992:	Flower Garden Banks National Marine Sanctuary (FGBNMS) designated, banning fishing techniques that damage benthic resources
1997:	New species discovered - the Mardi Gras Wrasse (*Halichoeres burekae*)
1998:	Coral bleaching event
2001:	Sanctuary designated as first International No-Anchor Zone by International Maritime Organization; ROV and sub surveys expand studies in deep portions of sanctuary first explored in the 1970s
2002:	Invasive species of orange cup coral, *Tubastraea* sp., discovered
2003:	First discovery of living *Acropora palmata* on banks
2005:	Coral bleaching event; coral disease event; and Hurricanes Katrina (Category 5) and Rita (Category 2)
2006:	Acoustically tagged manta rays found to travel between banks of the sanctuary; mass spawning of multiple sponge species first observed; and discovery of fossil *Acropora* reefs
2007:	Case of ciguatera poisoning leads to 2008 FDA Advisory on fish consumption
2008:	Coral bleaching event, Hurricane Ike (Category 2); Sanctuary research vessel MANTA begins service
2009:	Whale sharks found to travel between Mesoamerican reef and NW Gulf of Mexico
2010:	Deepwater Horizon oil spill response found no impacts to reefs
2011:	First lionfish sighting in sanctuary
2012:	FGBNMS listed under the Special Protected Areas and Wildlife (SPAW) Protocol of the Cartegena Convention

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FRENCH ANTILLES

Coauthors: Claude Bouchon, Yolande Bouchon-Navaro, Max Louis, Franck Mazeas, Jean-Philippe Maréchal, Pedro Portillo, Ewan Tregarot and Reef Check

Geographic Information

	Guadeloupe	Martinique	St. Barthélemy
Coastal Length (km):	576	365	32
Land Area (km^2):	1,746	1,151	22
Maritime Area (km^2):	28,764	18,673	4,000
Population:	436,366	408,147	9,072
Reef Area (km^2):	275	155	10

Number of hurricanes in the past 20 years: 8

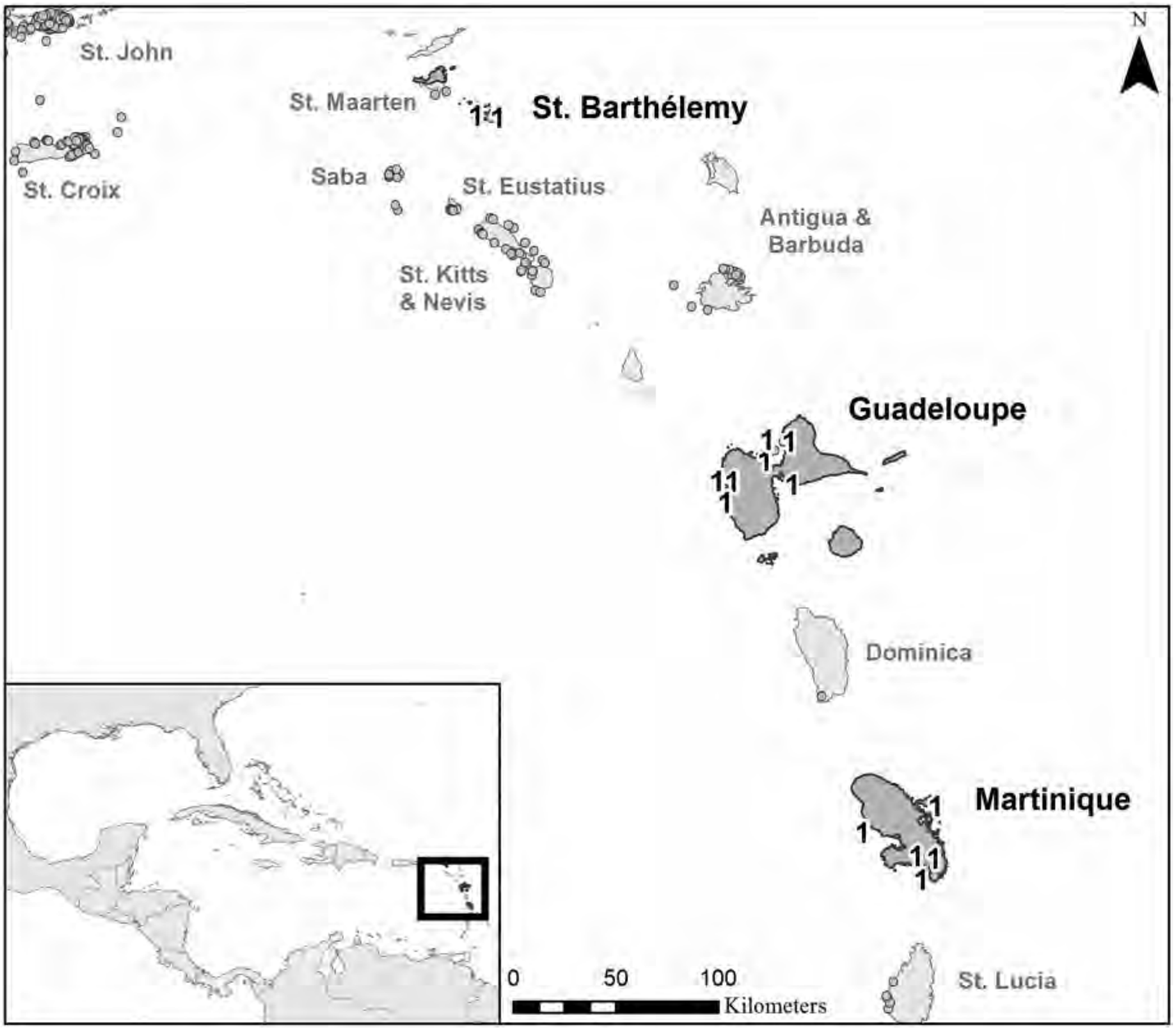

Fig. 17.1 Map of French Antilles, codes represent studies listed in Table 17.1. Missing map code(s) due to unavailable coordinates.

Table 17.1 Data sources from French Antilles used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 17.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Bouchon, Claude; Bouchon, Yolande*[1,2,3,4,5,6,7]	Martinique	2001-2007	7	X		X	X
1	Bouchon, Claude; Bouchon, Yolande*[1,2,3,4,5,6,7]	St. Barthélemy	2002-2011	10	X		X	X
1	Bouchon, Claude; Bouchon, Yolande*[1,2,3,4,5,6,7,8,9,10]	Guadeloupe	2002-2011	10	X		X	X
1	Maréchal, Jean-Philippe; Tregarot, Ewan, Burgneau, Sophie; Pérez, Cecile; Mahieu, Josianne; Renaudie, Bernard; Dupont, Priscilla*[11]	Martinique	2001-2009	2	X	X	X	
3	Reef Check*	Martinique; Guadeloupe	2003, 2007, 2008	3		X		

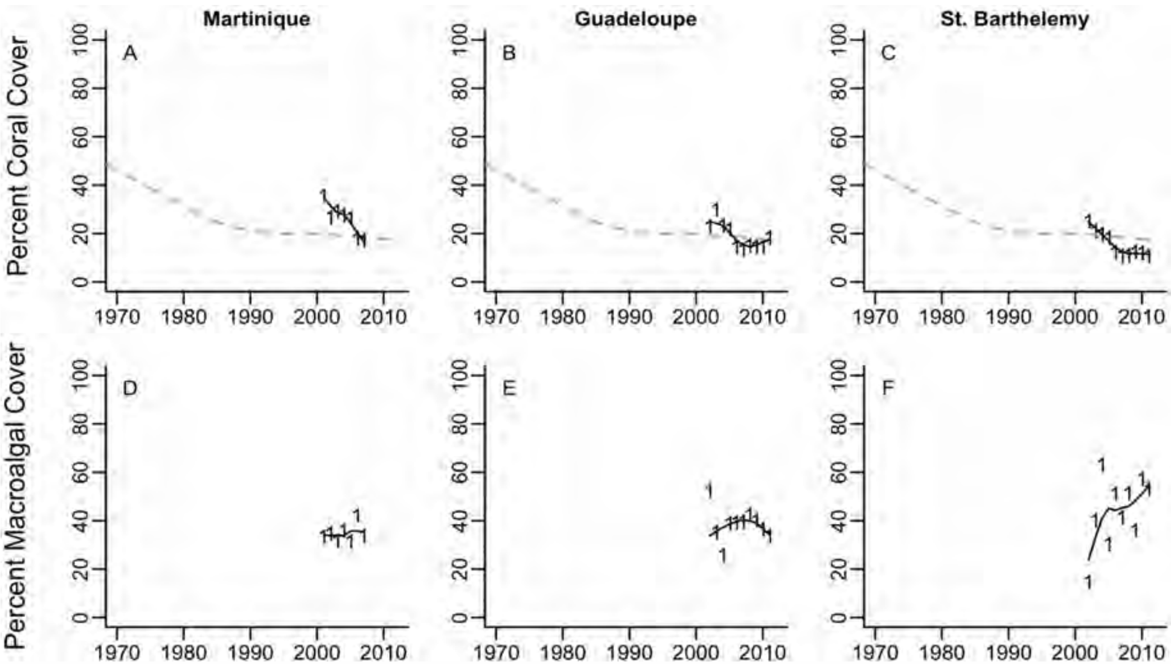

Fig. 17.2 Average percent cover of live corals and macroalgae for 3 islands in French Antilles. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 17.1 and Figure 17.1)

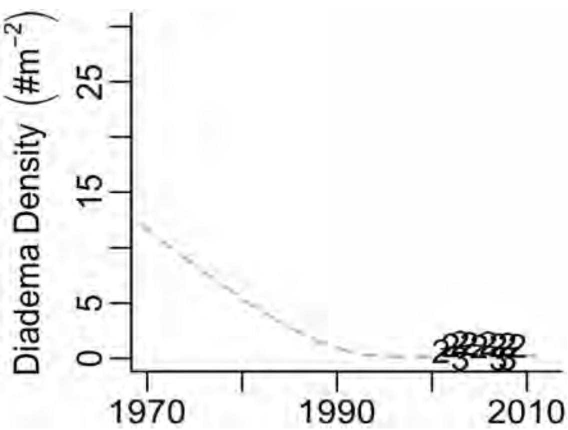

Fig. 17.3 Average density of *Diadema antillarum* for all French Antilles locations combined. Dotted line represents the average of Caribbean data collected for this report solid line is drawn through data presented. (Codes same as in Table 17.1 and Figure 17.1)

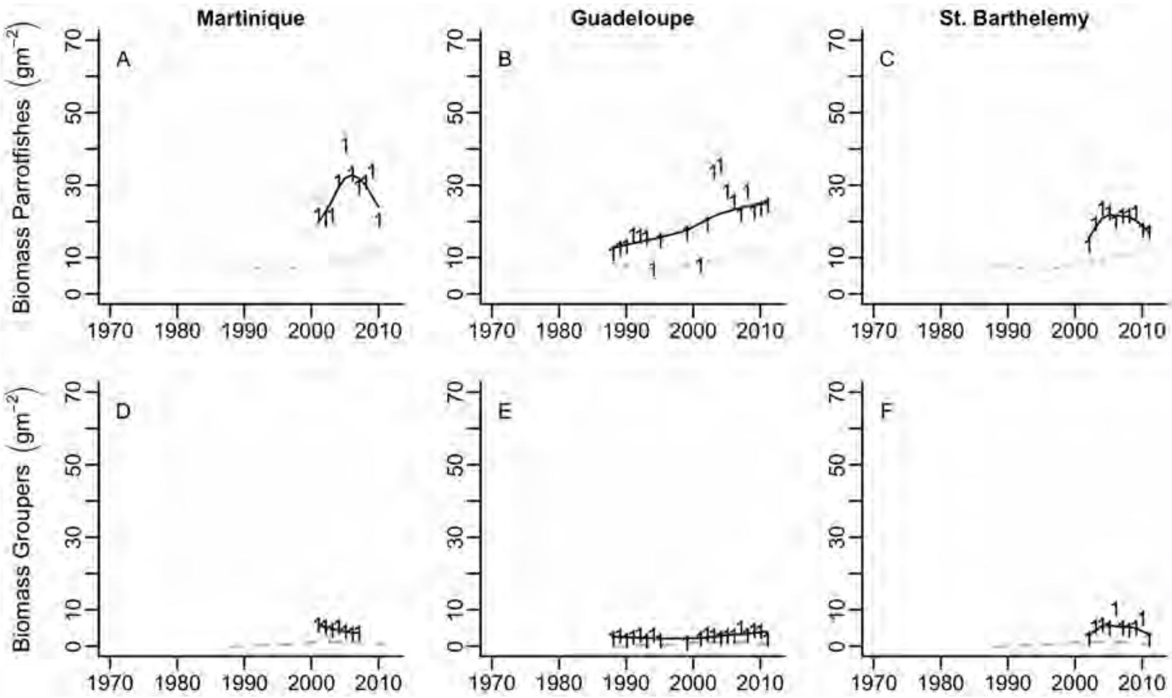

Fig. 17.4 Average biomass of parrotfishes (A-C) and groupers (D-F) for 3 islands in French Antilles. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 17.1 and Figure 17.1)

Timeline

1960s-1970s:	Mechanization of the coastal fishing boats
1970s-1979s:	Reefs appeared healthy, coral cover and benthic density high, few macroalgae
1979:	Hurricane David (Category 5) struck Dominique Island, relatively low damages on Martinique and Guadeloupe coral reefs
1980:	Gale associated to hurricane Allen (Category 5) damaged the coral reefs of Martinique and Guadeloupe
1983:	Mass mortality of *Diadema antillarum* occurred chronologically in Martinique, Guadeloupe, Saint-Barthélemy and Saint-Martin/St-Marteen
1984s-1985s:	White band disease massively destroyed the *Acropora palmata* and *A. cervicornis* communities
1986:	Coral macro-algae phase shift evident on the reefs of the French Antilles
1989:	Hurricane Hugo (Category 5), important damage on the Guadeloupe's reefs to 20m deep. Remaining *Acropora palmata* and *A. cervicornis* assemblages were further destroyed
1995:	Hurricanes Luis (Category 4) and Marilyn (Category 3) damaged the coral reefs of Saint-Barthélemy and Saint-Martin/St-Marteen
1996:	Bleaching event in the French Antilles, with moderate consequences (most of the bleached colonies recovered)
1998:	Bleaching event in the French Antilles, with moderate consequences (most of the bleached colonies recovered) Unexplained massive coral reef fishes mortality in Martinique and Guadeloupe
1999:	Hurricane Luis and Lenny (category 4) damaged the coral reefs of the leeward coast of Guadeloupe
2005:	Massive bleaching event of the corals of the French Antilles, ~40 % of the coral cover of the reefs disappeared, average loss in Martinique was 15%
2006:	Another 15% loss of coral from a disease outbreak following bleaching in Martinique; Invading seagrass species *Halophila stipulacea* reached Martinique
2005-2010s:	No sign of evident heath recovery of the coral communities. Increasing organic pollution continues to favor algal coral phase shift
2010-2011:	Lionfish *Pterois volitans* reached Martinique, Guadeloupe, Saint-Barthélemy and Saint-Martin/St-Marteen
2010:	Invading seagrass species *Halophila stipulacea* reached Guadeloupe, Saint-Barthélemy and Saint-Martin/St-Marteen
2011:	Massive *Sargassum* landing of the beaches of the French Antilles

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GRENADA

Coauthors: Jerry Mitchell, Clare Morrall, Steve Nimrod, Ernesto Weil, AGRRA, MACC and Reef Check

Geographic Information	
Coastal Length:	245 km
Land Area:	370 km^2
Maritime Area:	27,309 km^2
Population:	83,803
Reef Area:	118 km^2
Number of hurricanes in the past 20 years:	2

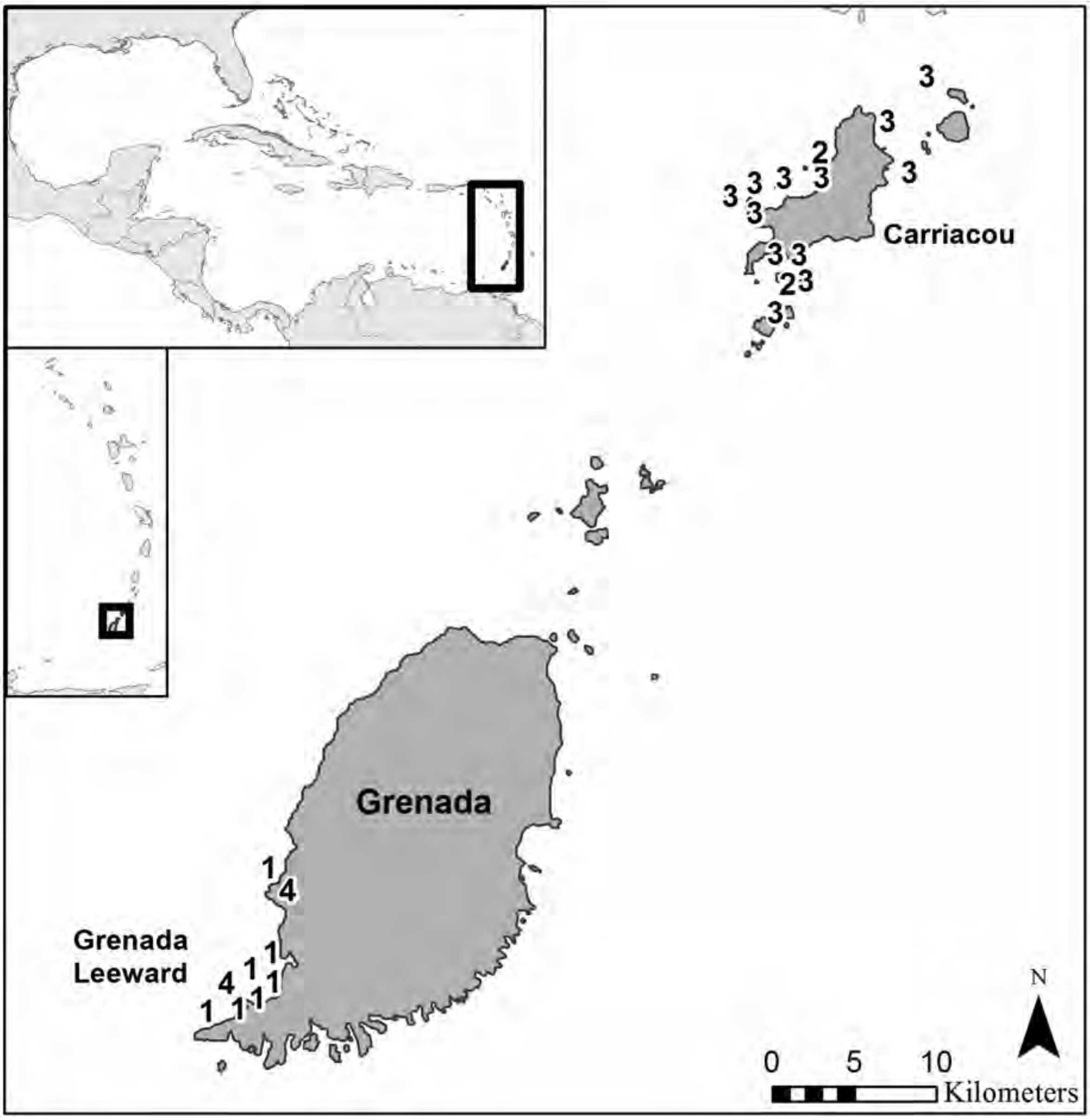

Fig. 18.1 Map of Grenada, codes represent studies listed in Table 18.1. Missing map code(s) due to unavailable coordinates.

Table. 18.1 Data sources from Grenada. Map codes represent individual studies. For exact location of study, refer to Fig. 18.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Mitchell, Jerry/MACC*[1,2]	Grand Anse Reef	2007, 2009	2	X		X	
2	Goodwin et al. 1976[3]	Carriacou	1976	1	X			
3	AGRRA*	Carriacou	2005	1	X	X		
4	Weil, Ernesto*	West coast	2005-2006, 2009	3	X		X	
5	Reef Check*		2004	1		X		

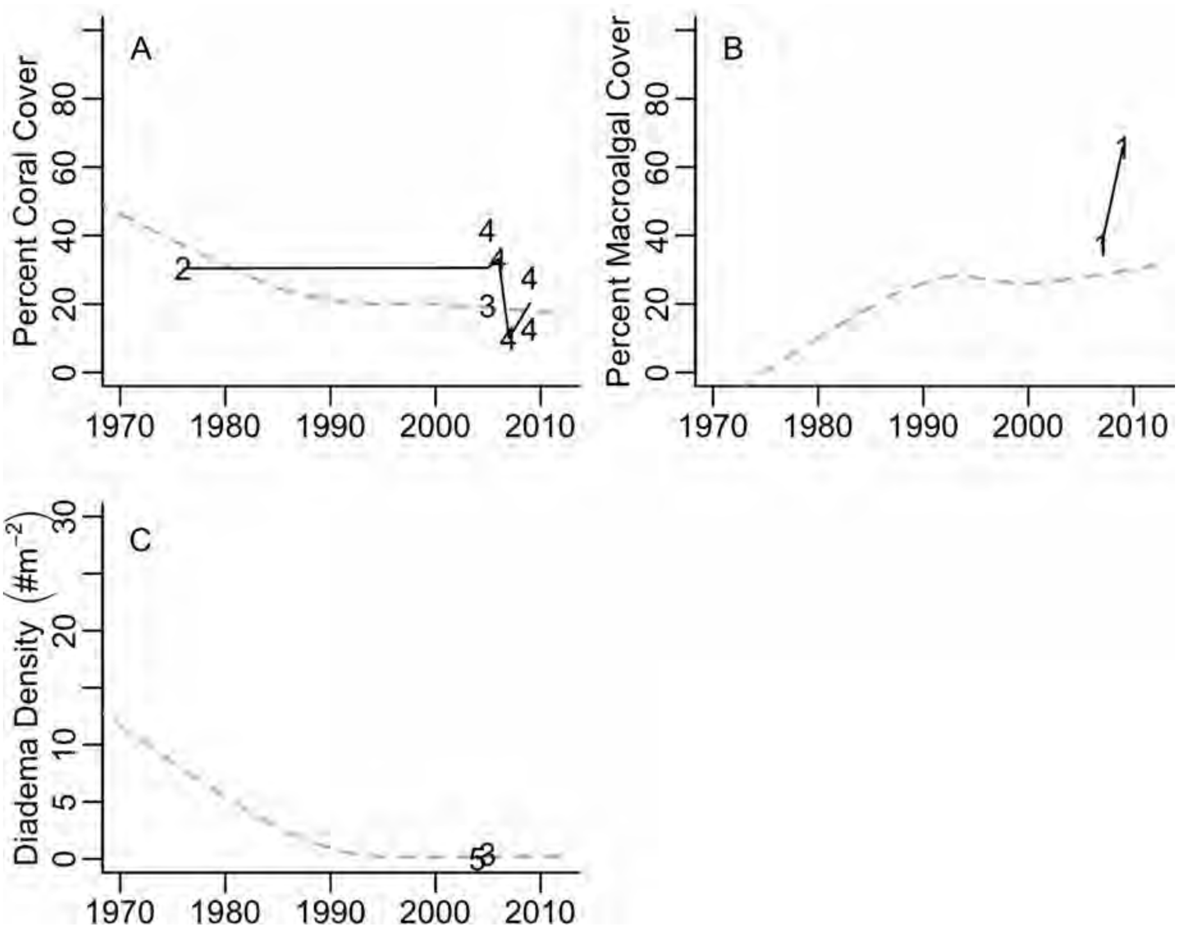

Fig. 18.2 Average percent cover of live corals (A) and macroalgae (B), and density of *Diadema antillarum* (C) in Grenada. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 18.1 and Figure 18.1)

Timeline

1995:	Sea egg (*Tripneustes ventricosus*) fishery moratorium (continued to present).
1999:	Severe storm surge associated with the passage of Hurricane Lenny (widespread physical damage to coral reefs on west coast of Grenada)
1999:	Mass mortality of reef fish in Grenada and some other Caribbean islands
2004:	Hurricane Ivan (Category 3)
2005:	Hurricane Emily (Category 1); coral bleaching event
2008:	Coral bleaching event
2010:	Mass mortality of reefs fish and eels on west coast of Grenada

2010: Coral bleaching event

2010: Sandy Island Oyster Bed MPA and Moliniere/Beausejour MPA management plan implemented and warden patrols commence

2011: Lionfish *Pterois volitans* first documented

General Literature

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GUATEMALA

Coauthors: Ana Giró, Melanie Mcfield, Robert Steneck, AGRRA, Healthy Reefs Initiative, The Nature Conservancy and Reef Check

Geographic Information
Number of hurricanes in the past 20 years: 1

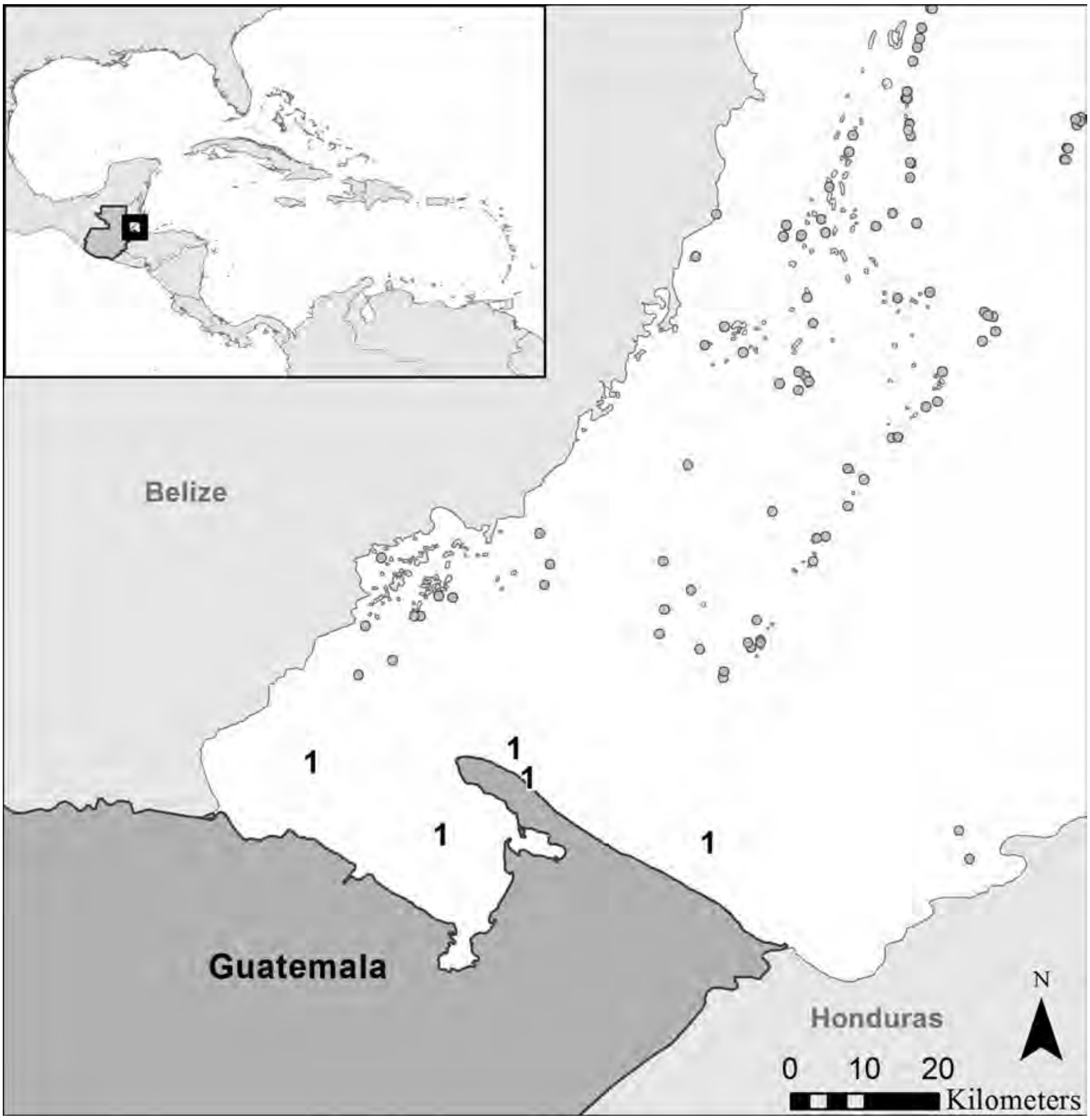

Fig. 19.1 Map of Guatemala, codes represent studies listed in Table 19.1. Missing map code(s) due to unavailable coordinates.

Table 19.1 Data sources from Guatemala collected in current synthesis. Map codes represent individual studies. For exact location of study, refer to Fig. 19.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroal-gae	Fishes
1	HRI/TNC/AGRRA*[1,2]	2006	1	X	X	X	X
2	Reef Check*	2006	1		X		
3	Steneck, Bob*	2007-2008	2	X		X	

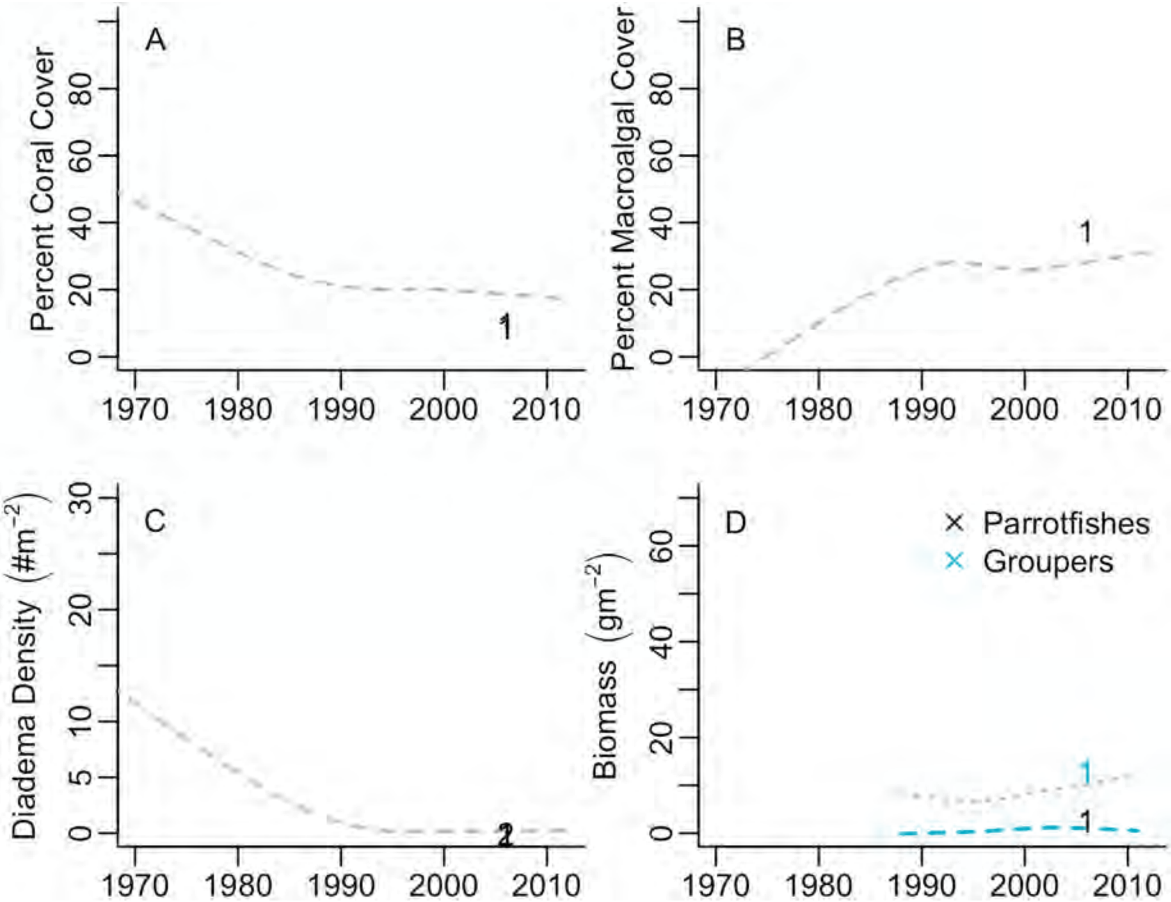

Fig. 19.2 Average percent cover of live corals (A), average percent cover of macroalgae (B), density of *Diadema antillarum* (C), and biomass of parrotfishes and groupers (D) in Guatemala. Dotted line represents the average of Caribbean data collected for this report. (Codes same as in Table 19.1 and Figure 19.1)

Timeline

1945:	Hurricane
1949:	Hurricane
1972:	Hurricane Fifi
1976:	Earthquake (Young *et al.* 1989)
1983:	Bleaching event; Mass mortality of *Diadema antillarum*
1998:	Hurricane Mitch; bleaching event
2009:	Ban on trawling inside the Wildlife refuge Punta de Manabique

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[2] McField M (2012) Report card for the mesoamerican reef, an evaluation of ecosystem health 2012. Healthy Reefs Initiative. 25 + 11 p.

HONDURAS

Coauthors: Andrés Alegría, Claude Bouchon, Steve Box, Ian Drysdale, Douglas Fenner, Melanie McField, Andrzej Narozanski, Pedro Portillo, Bob Steneck, AGRRA, Healthy Reefs Initiative and Reef Check

Geographic Information

Coastal Length:	1,879 km
Land Area:	112,583 km^2
Maritime Area:	219,772 km^2
Reef Area:	1,061 km^2
Number of hurricanes in the past 20 years:	5

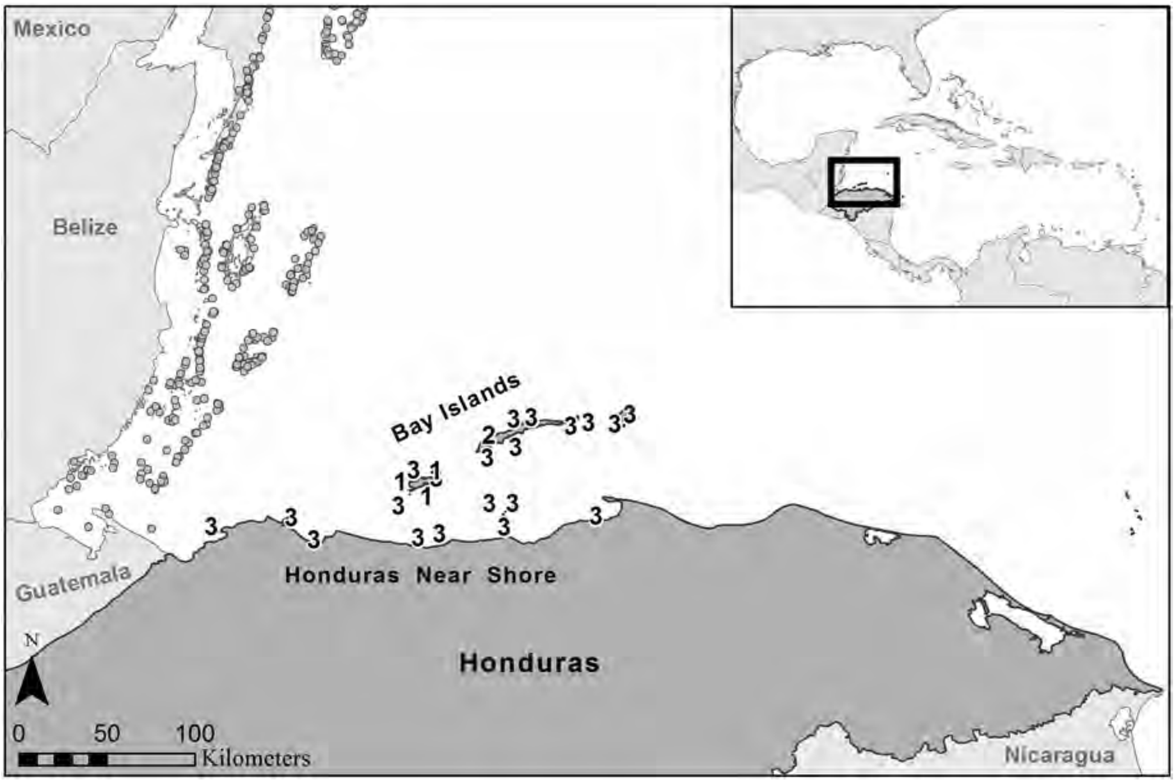

Fig. 20.1 Map of Honduras, codes represent studies listed in Table 20.1. Missing map code(s) due to unavailable coordinates.

Table 20.1 Data sources from Honduras collected in current synthesis. Map codes represent individual studies. For exact location of study, refer to Fig. 20.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Box, Steve; Narozanski, Andrzej*	Utila	2009-2010	2	X		X	X
2	Fenner, Douglas*[1]	Roatan	1987	1	X			
3	Mcfield, Melanie/HRI/ AGGRA*[2,3,4]	Bay Islands	2006, 2009	2	X	X	X	X
4	Hay 1984[5]	Media Luna	1981	1		X		
5	Reef Check*		1997-2007	11		X		
6	Bouchon, Claude; Portillo, Pedro*[6]	Bay Islands	1999-2002, 2007-2008	6	X		X	
7	Alegria, Andrés/ HRI/ AGRRA*	Tela, Roatan and Swan Islands	2011	1	X	X		X
8	Steneck, Bob*		2003, 2006 -2008	4	X		X	X

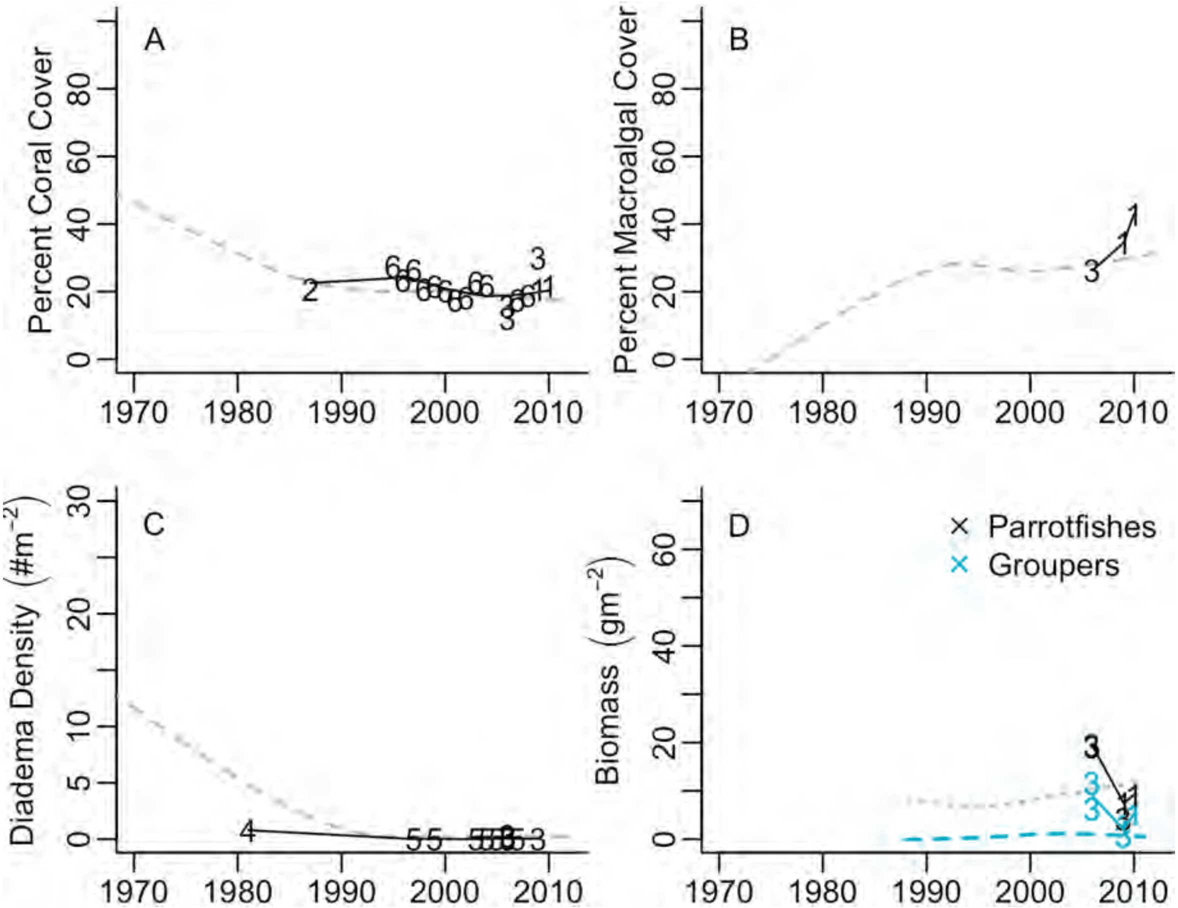

Fig. 20.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C), and biomass parrotfishes and groupers (D) in Honduras. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 20.1 and Figure 20.1)

Timeline

1950-1960s:	Can see many sharks from the dock in the Utila Cays and Utila (The Force Project, 2011 Utila Cays and Utila)
1959:	Creation of fishery law (The Force Project, 2011 – Honduras)
1960s:	Nets and pots were used to catch fish in the Utila Cays (The Force Project, 2011 Utily Cays)
1974:	Hurricane Fifi (category 2) damaged the coral reefs of the Bay Islands
Mid 1970s:	Spear fishing activities in Utila (The Force Project, 2011 Utila)
1978:	Hurricane Greta
1980s:	Big green parrotfish start to decline in Utila; red tide event (The Force Project, 2011 Utila)
1980:	End of dynamite fishing in West End (The Force Project, 2011 West End)
1983:	Mass mortality of *Diadema antillarum*
1989:	Fish pots and drag nets banned in West End; Sandy Bay/West End Marine Reserve declared (The Force Project, 2011 West End)
1990s:	Beautiful coral, no fish in Utila; cruise ships start arriving at the West End (The Force Project, 2011 Utila; West End)
1993:	Closed seasons established (The Force Project, 2011 Honduras)
1996:	Bleaching event
1997:	Declaration of Tulum, the starting point for initiatives in reef conservation; red tide in Utila (The Force Project, 2011 Utila)
1998:	Hurricane Mitch (Category 5); major damages on the coral reefs of the Bay Islands, major bleaching event which caused an important coral mortality on the reefs of the Bay islands, Utila and West End (The Force Project, 2011) Major impact to mangrove forests on Guanaja.
1999:	Coral Cay Conservation project initiated

2000:	Conch moratorium in West End (The Force Project, 2011 West End)
2005:	Roatan Marine Park established
2009:	First documentation of the lionfish *Pterois volitans*; earthquake with some damage to corals in Utila
2010:	Some coral coming back for the first time in Utila (The Force Project, 2011 Utila); Bay Islands Marine National Park established (The Force Project, 2011 West End)
2011:	Socialization of new Fisheries Law and shark sanctuary established (The Force Project, 2011 Honduras)

General Literature

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Berthou P, Lespagnol P, Andreakis V, López E, Oquelí MD, et al. (2000) El censo de los pescadores artesanales y de los botes de pesca de las Islas de la Bahía. Proyecto de Manejo Ambiental de las Islas de la Bahía. Informe técnico PES 01, 68.

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JAMAICA

Coauthors: Karl Aiken, Robert Carpenter, Bernadette Charpentier, Marcia Creary, Rachel D'Silva, Phil Dustan, Peter Edmunds, Peter Gayle, Nakhle Hado, Marah Hardt, Terry Hughes, Joshua Idjadi, Jeremy Jackson, Stephen Jameson, Nancy Knowlton, Judy Lang, Yossi Loya, Gustavo Paredes, Nicholas Polunin, Bob Steneck, Ivor Williams, AGRRA, CPACC, CARICOMP and Reef Check

Geographic Information

Coastal Length:	894 km
Land Area:	11,000 km^2
Maritime Area:	242,311 km^2
Population:	2,798,580
Reef Area:	763 km^2
Number of hurricanes in the past 20 years:	3

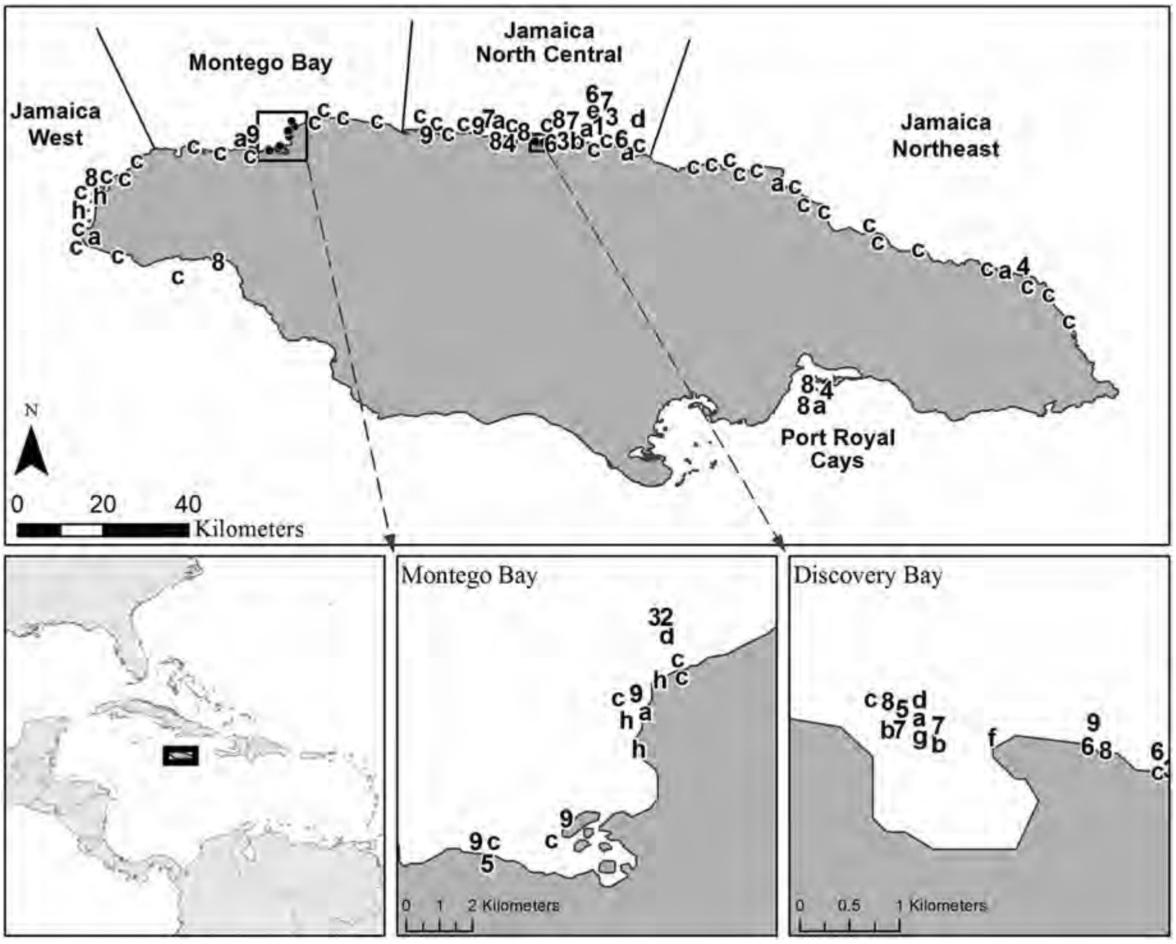

Fig. 21.1 Jamaica, codes represent studies listed in Table 21.1. Missing map code(s) due to unavailable coordinates. AGRRA locations are omitted for clarity.

Table 21.1 Data sources for Jamaica used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 21.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Andres & Witman 1995[1]	North Central Jamaica	1992	1	X		X	
2	Aronson & Precht 2000[2]	Montego Bay	1993-1996, 1998-1999	6	X	X	X	
3	Cho & Woodley 2000[3]	Montego Bay, North Central Jamaica	1994, 1997	2	X		X	
4	CPACC*	Islandwide	2001-2003	3	X		X	
5	Dustan, Phil*	Montego Bay, North Central Jamaica	1972-1973	2	X			
6	Edmunds & Bruno 1996*[4]	North Central Jamaica	1984	1	X	X	X	
7	Edmunds, Peter; Carpenter, Robert*	North Central Jamaica	2000	1	X		X	
8	Gayle, Peter; Charpentier, Bernadette/CARICOMP*[5]	Islandwide	1994-2007, 2009-2012	18	X	X	X	
9	Hardt, Marah; Paredes, Gustavo*[6]	Montego Bay, North Central Jamaica	2006	1	X		X	X
a	Hughes, Terry*[7,8,9,10,11,12,13]	9 sites islandwide	1977-1990, 1993	15	X	X	X	
b	Idjadi, Joshua*[14]	North Central Jamaica	2000	1	X	X	X	
c	AGRRA*[15]	66 sites islandwide	2000, 2005	2	X	X		X
d	Knowlton et al. 1990[16]	Montego Bay, North Central Jamaica	1982, 1984-1987	6	X	X	X	
e	Liddell & Ohlhorst 1986[17], 1987[18], 1988[19], 1992[20]	North Central Jamaica	1977, 1980-1984, 1987, 1989	8	X	X	X	
f	Loya, Yossi*	North Central Jamaica	1969	1	X			
g	Steneck, Bob*[21]	North Central Jamaica	1978, 1982, 1988	3	X	X	X	
h	Williams, Ivor; Polunin, Nicholas*[22]	Montego Bay, North Central Jamaica	1997	1	X		X	X
i	Morrison 1988[23]	North Central Jamaica	1982	1		X		
k	Sammarco 1980[24], 1982[25]	North Central Jamaica	1973, 1976	2		X		
m	Moses & Bonem 2001[26]	North Central Jamaica	1998	1		X		
n	Reef Check*	42 sites islandwide	1998, 2000-2008	10		X		
o	Knowlton 1981[27]	North Central Jamaica	1997, 1981	2		X		
p	Haley & Solandt 2001[28]	North Central Jamaica	1996-2000	6		X		
q	Woodley 1999[29], 1981[30]	North Central Jamaica	1976-1977, 1980	3		X		
r	Karlson 1983[31]	North Central Jamaica	1976-1978	3		X		
s	Jackson 1987[32]	North Central Jamaica	1982-1983, 1986	3		X		

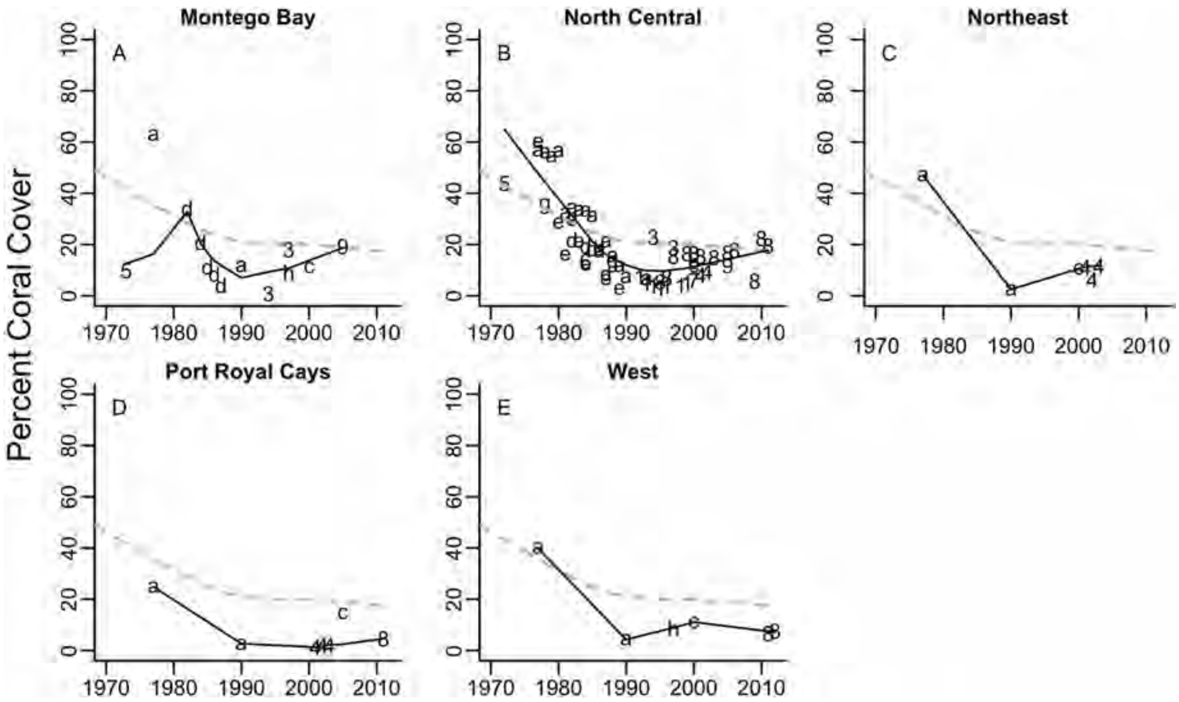

Fig. 21.2 Average percent cover of live corals for 5 locations in Jamaica: Montego Bay (A), North Central coast (B), Northeast coast (C), Port Royal Cays (D) and West Jamaica (E). Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 21.1 and Figure 21.1)

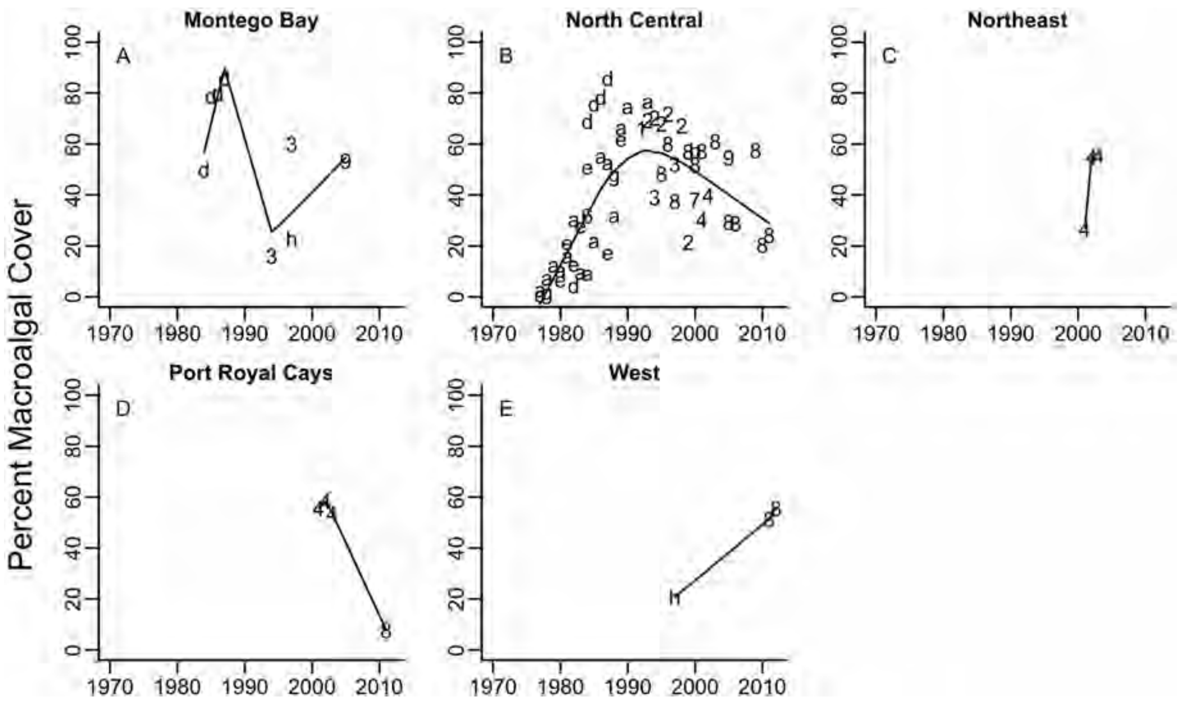

Fig. 21.3 Average percent cover of macroalgae for 5 locations in Jamaica: Montego Bay (A), North Central coast (B), Northeast coast (C), Port Royal Cays (D) and West Jamaica (E). Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 21.1 and Figure 21.1)

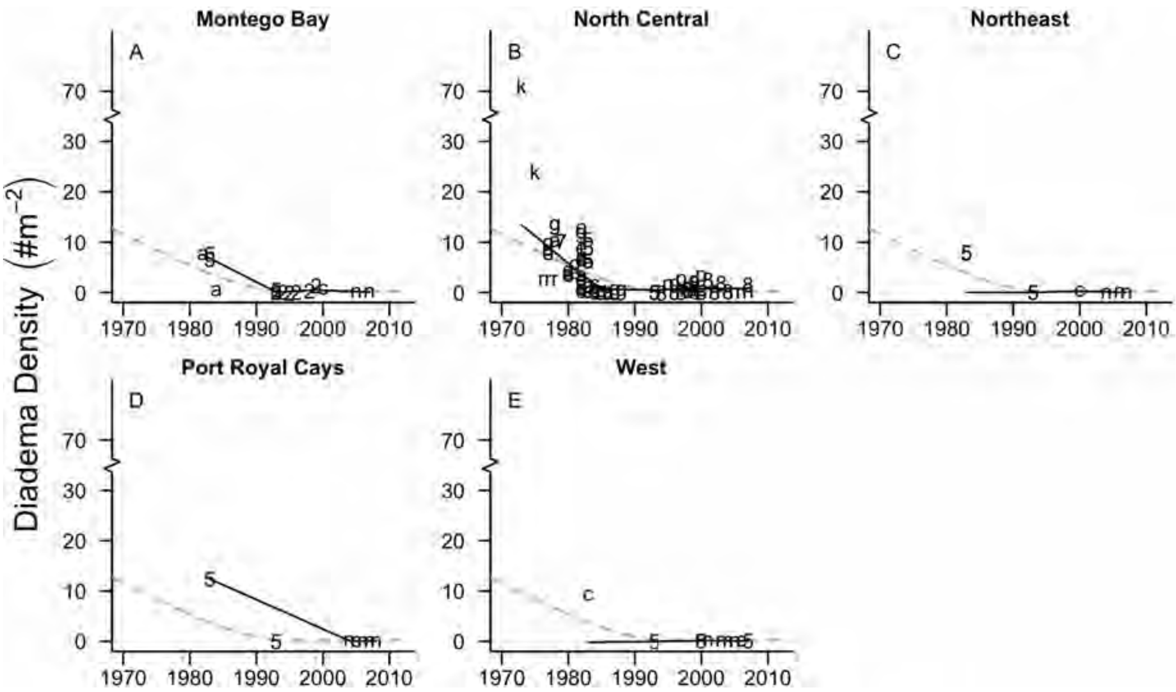

Fig. 21.4 Average density of *Diadema antillarum* for 5 locations in Jamaica: Montego Bay (A), North Central coast (B), Northeast coast (C), Port Royal Cays (D) and West Jamaica (E). Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 21.1 and Figure 21.1)

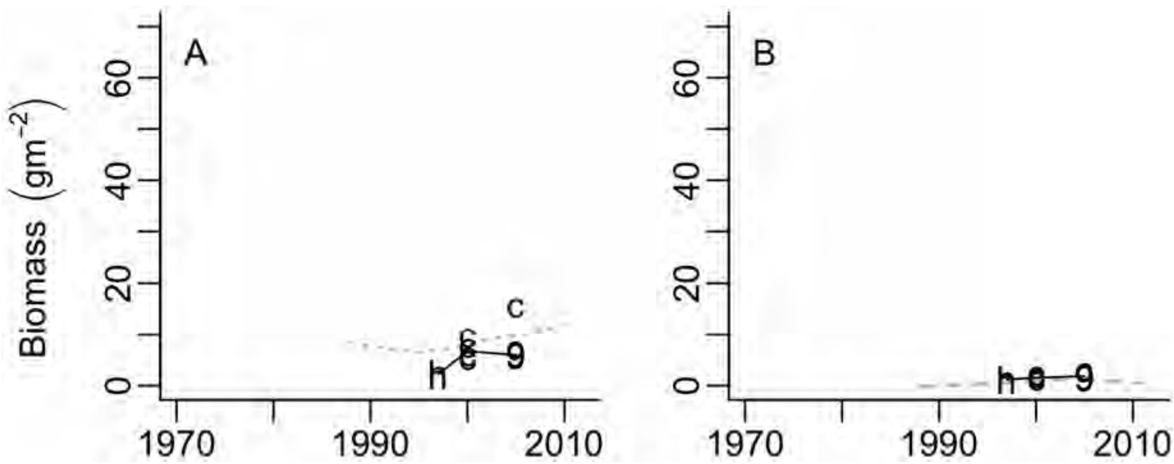

Fig. 21.5 Average biomass of parrotfishes (A) and groupers (B) for all locations in Jamaica combined. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 21.1 and Figure 21.1)

Timeline

1950s-1970s:	Reefs appeared healthy, coral cover and benthic density high, relatively few macroalgae
1963:	Hurricane Flora; followed by massive bleaching event in the south coast (Goreau 1964)
Mid-1960s:	Grouper fishery collapsed (Munro's work)
1973:	~1800 fishing canoes deploying traps on the north coast
1970s-1980s:	Mechanization of fishing fleets
1980:	Hurricane Allen (Category 5)
1983:	Mass mortality of *Diadema antillarum*

1987:	Minor bleaching event
1988:	Hurricane Gilbert
1990:	Record growth in mass tourism and agriculture
1990s:	Minimum 1.25-inch mesh size for pots
1995:	Minor bleaching event
2004:	Hurricane Ivan
2005:	Bleaching, affecting 45-75% coral cover
2005:	Lionfish *Pterois volitans* first documented
2007:	Hurricane Dean
2010:	Bleaching, affecting 18-40% coral cover
2011:	Inshore waters heavily overfished, fishermen forced to fish in open waters

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MEXICO

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Geographic Information

Coastal Length:	23,516 km
Land Area:	1,956,366 km^2
Maritime Area:	3,149,371 km^2
Reef Area:	1,481 km^2
Number of hurricanes in the past 20 years:	14

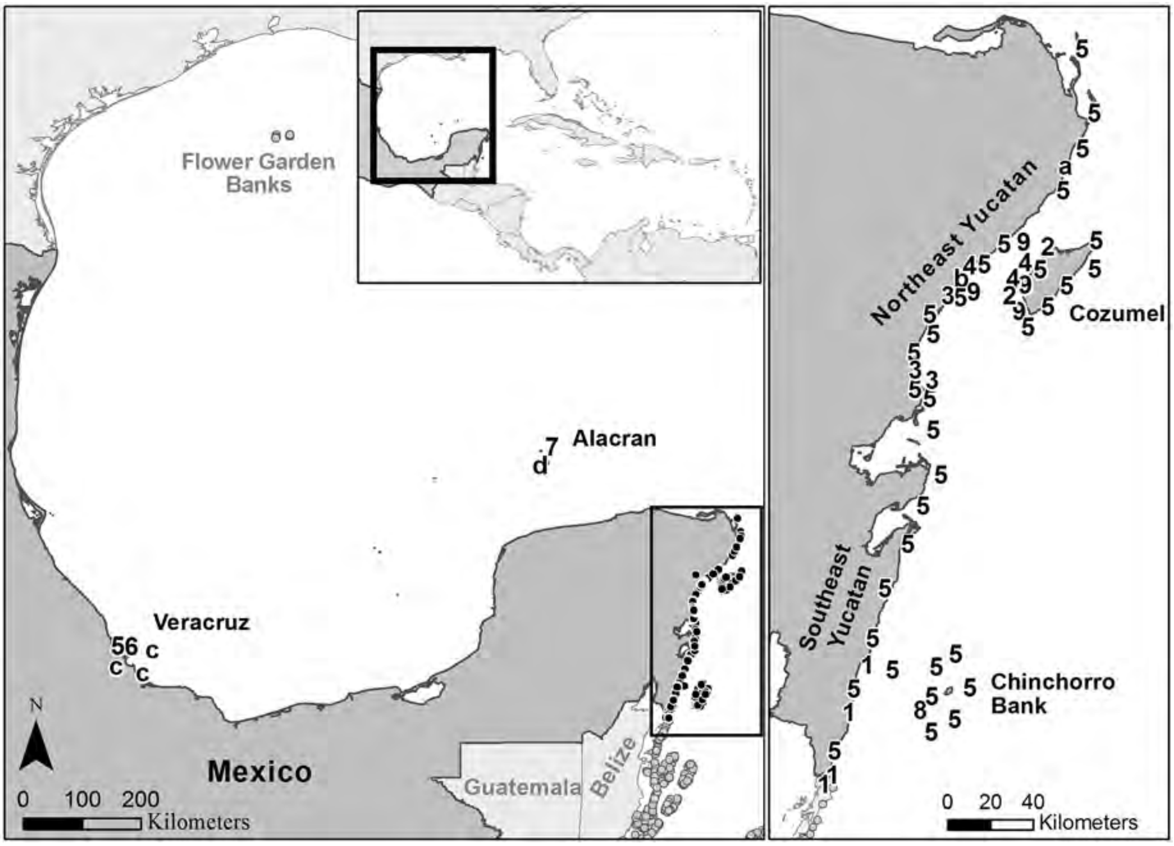

Fig. 22.1 Map of Mexico, codes represent studies listed in Table 22.1. Missing map code(s) due to unavailable coordinates.

Table 22.1 Data sources from Mexico used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 22.1; * denotes original data; for full references, refer to published literature sources in the last section. SE = Southeast; NE = Northeast

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Arias, Ernesto*[1]	SE Yucatán	1999, 2000, 2005-2008,	6	X		X	X
2	Fenner, Douglas*[2]	Cozumel (Leeward)	1984, 1986, 1988	3	X			
3	Garza, Rodrigo*[3,4]	NE Yucatán	2001, 2002, 2009-2010	4	X		X	X
4	Hardt, Marah; Paredes, Gustavo*[5]	Cozumel (Leeward); NE Yucatán	2004	1	X		X	X

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
5	AGRRA*[6,7,8,9] / HRI / TNC	SE Yucatán; NE Yucatán; Chinchorro Bank; Cozumel (Leeward & Windward); Veracruz	1999-2000, 2005-2006, 2009	5	X	X	X	X
6	Kühlmann 1975[10]	Veracruz	1965	1	X			
7	Liddell & Ohlhorst 1988[11]	Alacran	1985	1	X		X	
8	Mumby, Peter*[12]	Banco Chinchorro	2002	1	X		X	
9	Bonilla Reyes, Héctor*[13,14]	Cozumel (Leeward)	2005-2011	1	X		X	
a	Jordán Dahlgren, Eric; Rodríguez-Martínez Rosa/ CARICOMP*[15,16]	NE Yucatán	1979, 1985, 1989,1993-1999, 2001, 2003, 2004-2009	14	X	X	X	
b	Roy 2004[17]	NE Yucatán	1998-2000	3	X			
c	Secretaria de Marina 1987[18]	Veracruz	1985-1986	2	X			
d	Chávez et al. 2007[19,20]	Alacran	1985	1	X			
e	Bauer 1980[21]	Cozumel	1979	1		X		
f	Reef Check*	Akumal; Xcalak	1997, 2000, 2002-2007	8		X		
g	Steneck, Bob*		2003	1	X		X	

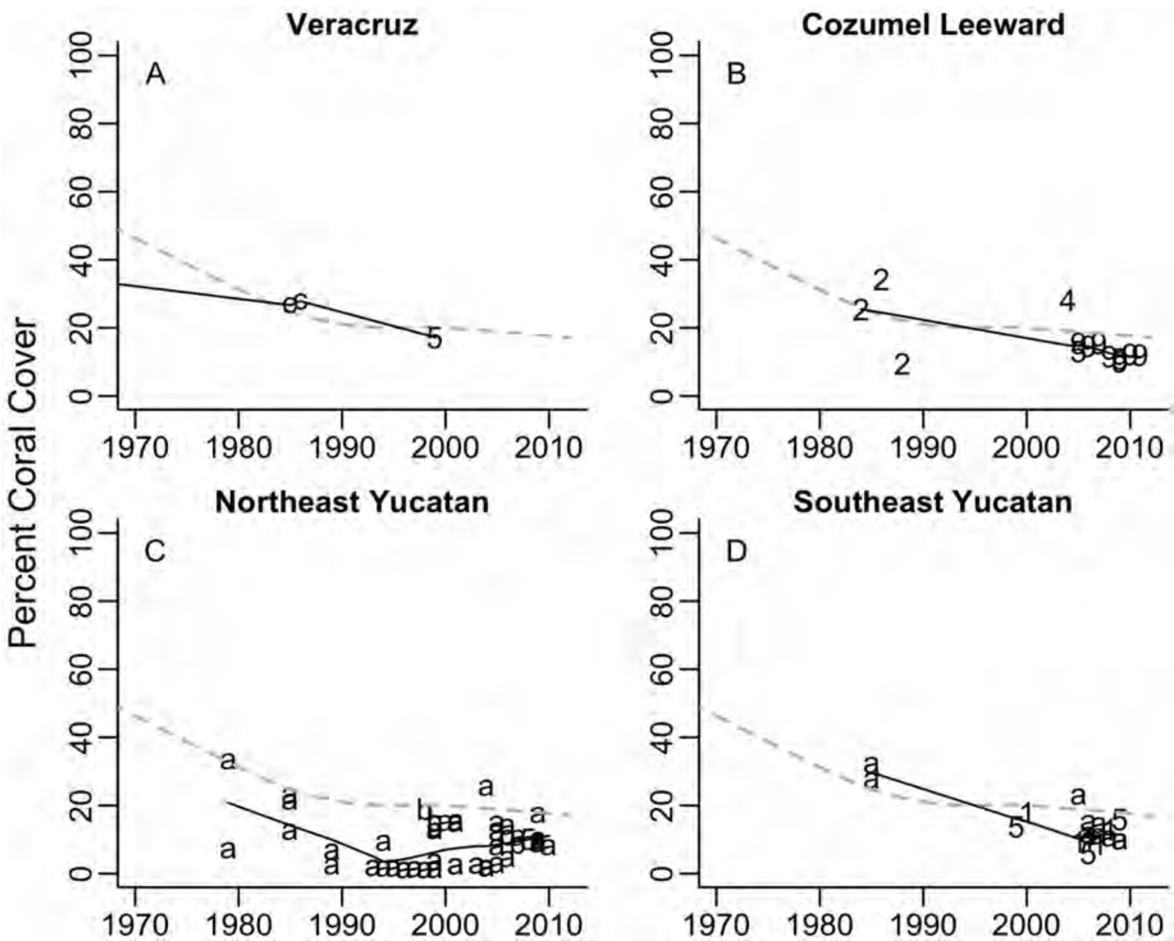

Fig. 22.2 Average percent cover of live corals for 4 locations in Mexico: Veracruz (A), Leeward Cozumel (B), Northeast Yucatán (C) and Southeast Yucatán (D). Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 22.1 and Figure 22.1)

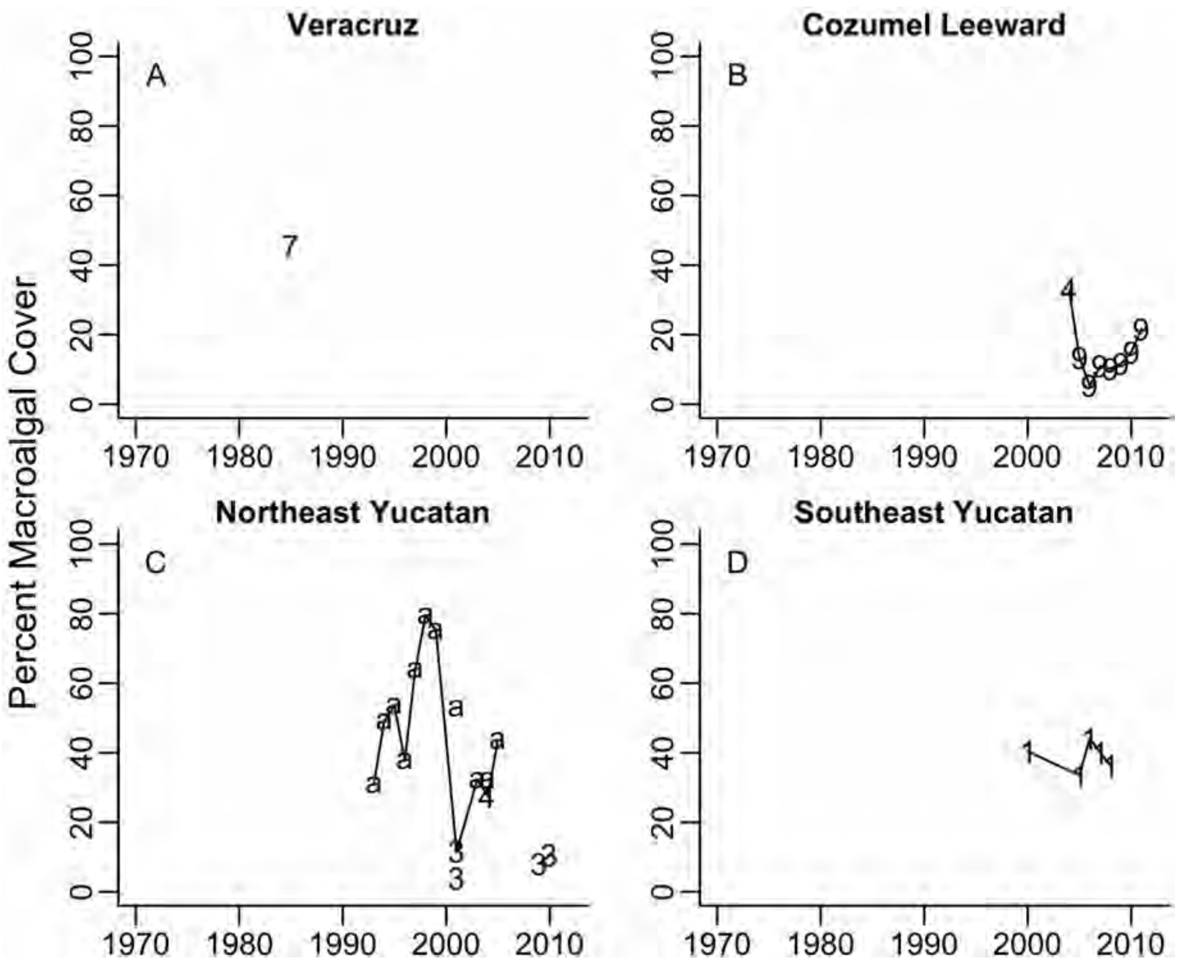

Fig. 22.3 Average percent cover of macroalgae for 4 locations in Mexico: Veracruz (A), Leeward Cozumel (B), Northeast Yucatán (C) and Southeast Yucatán (D). Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 22.1 and Figure 22.1)

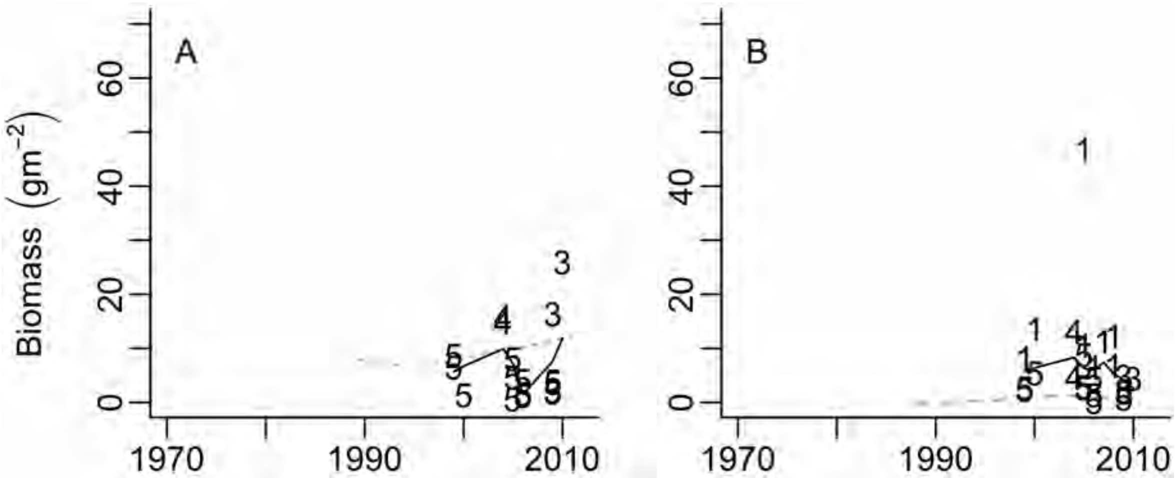

Fig. 22.4 Average biomass of parrotfishes (A) and groupers (B). Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 22.1 and Figure 22.1)

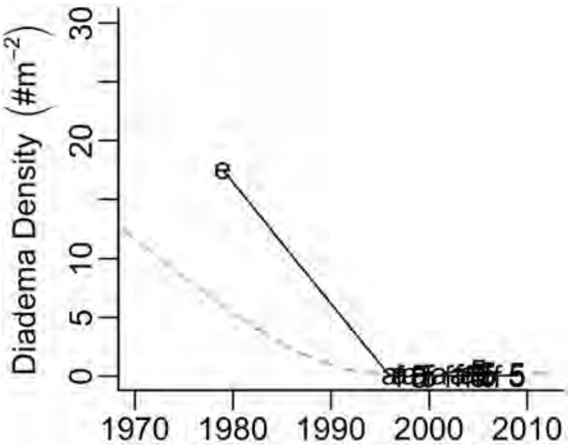

Fig. 5 Average biomass of parrotfishes (A) and groupers (B) for all Mexico locations combined. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 1 and Figure 1)

Timeline

11,000 CE:	Human remains on coastal caves of the Caribbean coast
200 BC to 1541 CE:	Mayan colonization of the Yucatán, large and abundant settlements along the Caribbean coast
1540-1950:	Period of low-density population along the mainland coast.
1843:	Fishing on turtle: loggerhead for its eggs and oil; green turtle for its meat, eggs and shell; hawksbill for its shell and meat
1898-1950:	Shipment activities of agricultural products (wood, chewing gum) that favors colonization of the Puerto Morelos area.
1900:	Green turtle fishing increased in Quintana Roo as fisherman of British Honduras worked out stock in their area
1906:	Documented fishing on *Strombus*, barracuda, *Chelonia mydas* and *Eretmochelys imbricate*, sponge
1928:	Henequen, coconut and turtle fishing were the chief industries of Cozumel
1940:	The waters off the northern end of Quintana Roo were exploited heavily by handliners and shark fishermen from Cuba
1950:	Puerto Morelos has a population of 80 habitants; turtles continue to be a main export product
1951:	Hurricane Charlie (Category 3)
1952:	Deployment of “Atajo” traps in southern coast (Costa Maya)
1961:	Hurricane Carla (Category 1)
1967:	Hurricane Beulah (Category 2)
1970:	Luxury species as conch, shrimp and lobster are fished in the northern, central and southern zones
1975:	Creation of Cancún
1980:	Cancún and nearby areas has a population of 226,000; 99,500 tourist visits; establishment of ecological zone in SW Cozumel
1978:	Beginning of commercial lobster fishing on Puerto Morelos reefs (spearfishing)
1978-1980:	First Puerto Morelos reefs survey. Reefs naturally well developed on the crest and back reef and coral-gorgonian grounds dominate the low profile fore reef. Healthy and pristine in some sites. Back reef coral cover (31 ± 26%) and reef-crest (33 ±21%); fore-reef coral cover (7 ± 8%)
1979:	Concerns about the rapid rate of conch exploitation produce regulations that limit its capture to six tons per month in Xcalak and two tons per month in Cozumel, Vigía Chico and Cozumel; fishing gears include Australian-style lobster traps, lobster nets, artificial habitats to attract lobsters, turtle nets, shrimp nets, snapper reels, shark longlines, nets-including seines and gill nets, lobster gaffs, spearguns, handlines
1980:	Outbreak of White Band Disease in *Acropora cervicornis* and *A. palmata;* Hurricane Allen (Category 5)
1982:	Mass mortality of *Diadema antillarum*
1988:	Hurricane Gilbert (Category 5); Hurricane Keith (Category 1); beginning of trap lobster fishing on the deep fore reef /shelf edge and net trapping of migrating snappers in the reef lagoon.
1989:	Second Puerto Morelos reefs survey: severe drop in coral cover (loss of 68-85%, mostly due to the acroporids demise).
1990:	Cancún and nearby areas population 176,765; 1.5 million tourist visits

1992:	CARICOMP surveys begin
1995:	First massive bleaching event recorded affected > 50 % coral colonies; Hurricane Roxanne (Category 3); Cancún and Isla Mujeres Reef National Park created
1997:	Mild bleaching event – subjective estimation < 20 % coral colonies.
1998:	Mild bleaching event – subjective estimation affecting 20-50% coral colonies; outbreak of white-pox disease in *A. palmata* (prevalence = 9%).
1998:	Creation of the Puerto Morelos Reef National Park, fishing is banned; beginning of coral bleaching; Hurricane Mitch (Category 5) struck Quintana Roo Coast
1999:	Fisherman at Puerto Morelos change from fishing to snorkeling and dive operators
2000:	Cancún and nearby areas population 419,815; 3 million tourist visits; Mahahual cruise pier construction begins; first observation of yellow-band disease in Mexico from Quintana Roo; Hurricane Keith
2001:	Epizootic of yellow-band disease in *Montastraea annularis* species complex (prevalence = 22%)
2002:	Permanent ban for fishing queen conch; Hurricane Isidore (Category 3) over northern Quintana Roo
2003:	Bleaching event, affecting 20-50% coral colonies.
2004:	Bleaching event, affecting 20-50% coral colonies; high prevalence of yellow-band disease (52%) and white-pox diseases (11%); Hurricane Ivan (Category 5)
2005:	Hurricanes Emily (Category 3); Wilma (Category 4) devastates Cozumel and northern Quintana Roo reefs; bleaching event affecting >50% coral colonies.
2006-2008:	Mild bleaching events, affecting <20% coral colonies.
2007:	Hurricane Dean (Category 5) strikes Quintana Roo, levels Mahahual village and devastates Chinchorro Bank
2009:	Lionfish *Pterois volitans* first documented in Quintana Roo;
2010:	Cancún and nearby areas population 661,176; 4 million tourist visits
2009-2011:	Bleaching events, affecting 20-50% coral colonies.
2012:	Pilot project to control lionfish through fishing cooperatives; Hurricane Ernesto makes landfall over Mahahual
Present:	During tourist high season, Cozumel has 20-30 cruise ship visits bringing 70,000-80,000 weekly visitors

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NAVASSA ISLAND

Coauthors: Margaret Miller, David McClellan and Mandy Karnauskas

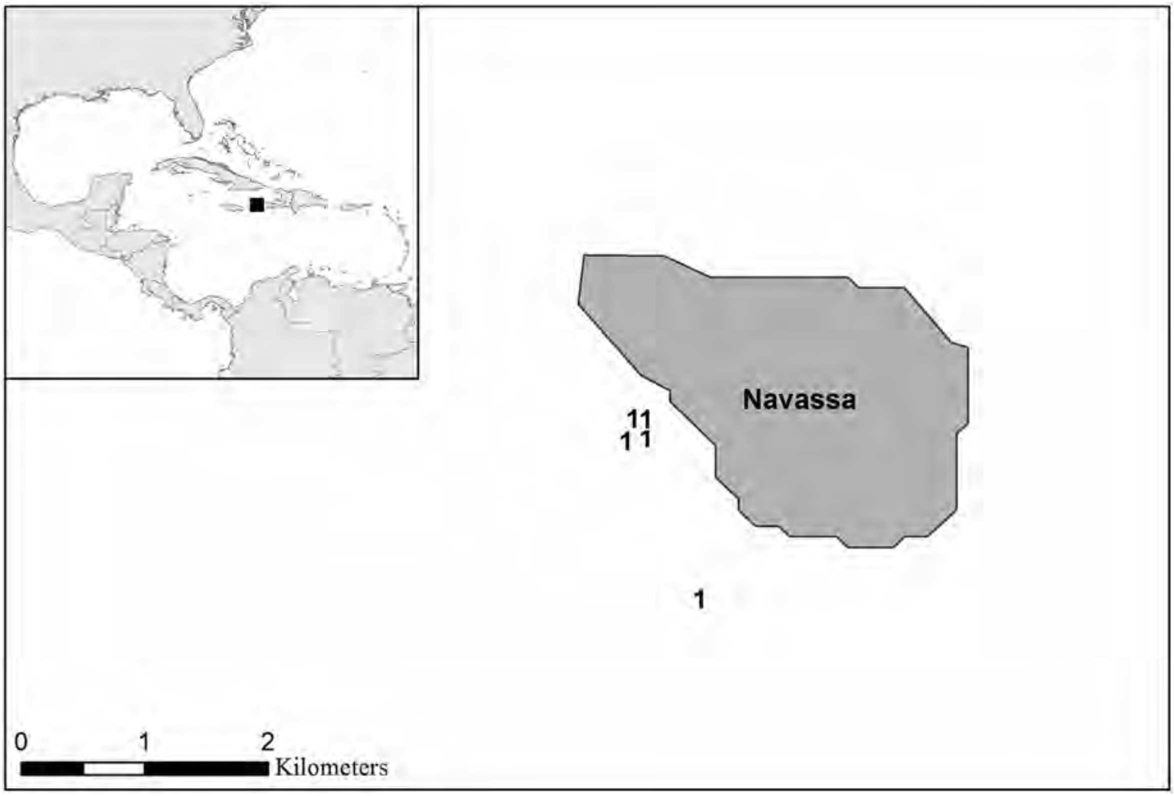

Fig. 23.1 Map of Navassa Island, codes represent studies listed in Table 23.1. Missing map code(s) due to unavailable coordinates.

Table 23.1 Data sources from Navassa Island. Map codes represent individual studies. For exact location of study, refer to Fig. 23.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Miller, Margaret; McClellan, David*[1,2]		2002, 2004, 2006, 2009, 2012	5	X		X	
2	Miller, 2002[3]		2000	1		X		

Timeline

July – August 1998:	Initial ecological characterization of Navassa Reefs by the Center for Marine Conservation (CMC)
1999:	Comprehensive fish inventory and terrestrial inventory including geological assessments by CMC
2000:	Surveys on echinoderms, molluscs, crustaceans and general reef status in shallow shelf habitats; initial documentation of artisanal fishing via hook-and-line and Antillean Z-traps.
2002:	Comprehensive reef assessment including fish counts, benthic community description and fishery description; first observation of net fishing
2004:	Hurricanes Charley and Ivan (Category 4)
November 2004:	Severe disease outbreak; elevated fishing activity observed including extensive use of triple mesh entangling nets
April 2006:	Multibeam mapping of island completed; temperature loggers installed; minimal bleaching and fishing activity observed
November 2006:	Severe coral bleaching underway; apparent cessation of net fishing
2009:	First sighting of lionfish; evidence of relaxation of fishing impact; CPUE data collected for trap fishery
2012:	Lionfish commonly sighted; CPUE data collected for trap fishery

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NICARAGUA

Coauthors: Stephen Jameson, AGRRA
and CARICOMP

Geographic Information

Coastal Length:	1,887 km
Land Area:	128,469 km^2
Maritime Area:	151,434 km^2
Reef Area:	757 km^2
Number of hurricanes in the past 20 years:	4

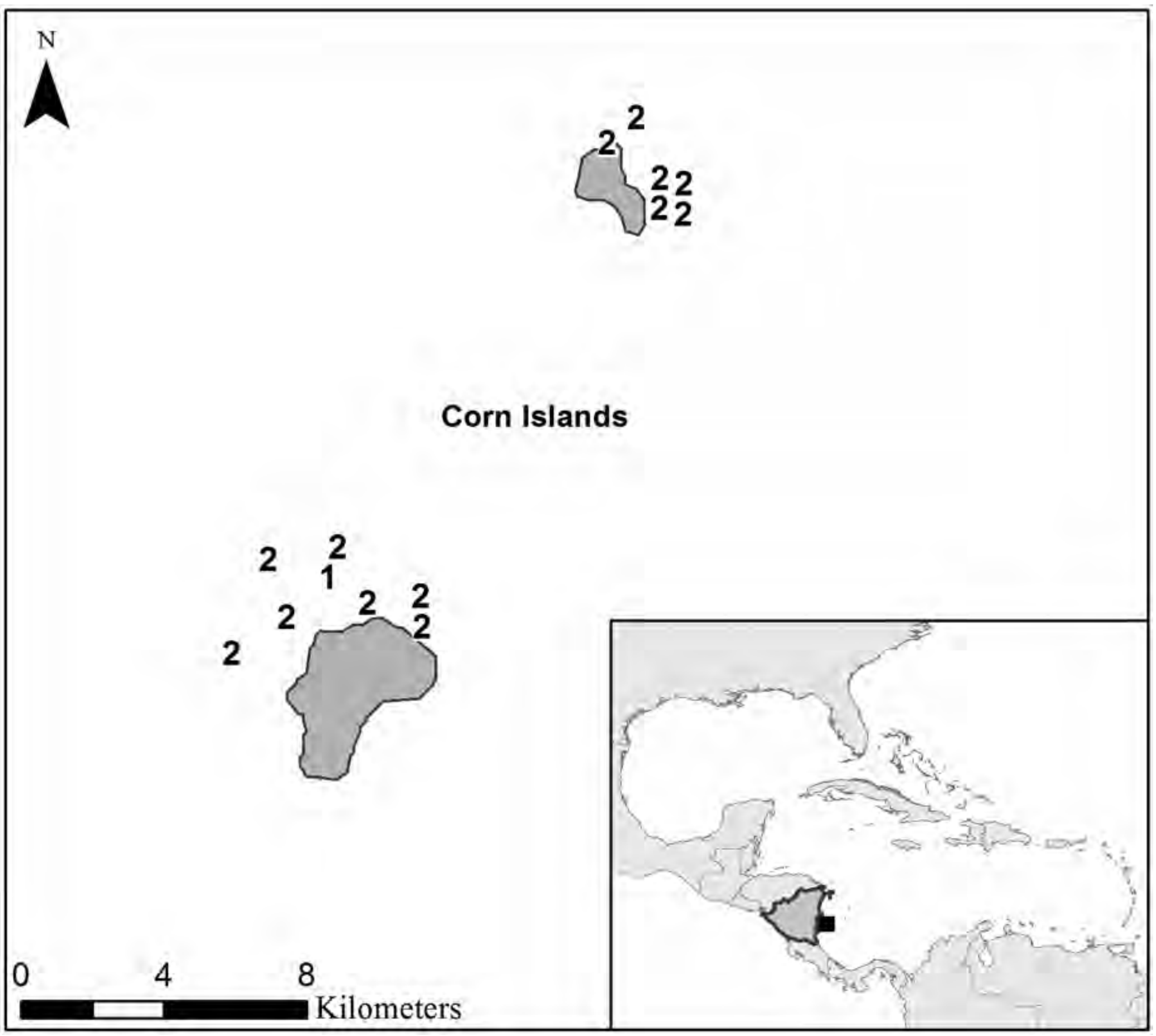

Fig. 24.1 Map of Nicaragua, codes represent studies listed in Table 24.1. Missing map code(s) due to unavailable coordinates.

Table 24.1 Data sources from Nicaragua used in current analysis. Map codes represent individual studies. For exact location of study, refer to Fig. 24.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	CARICOMP*[1]	1993, 1995, 1997-1998	4	X		X	
2	AGRRA*[2]	2003	1	X	X		X

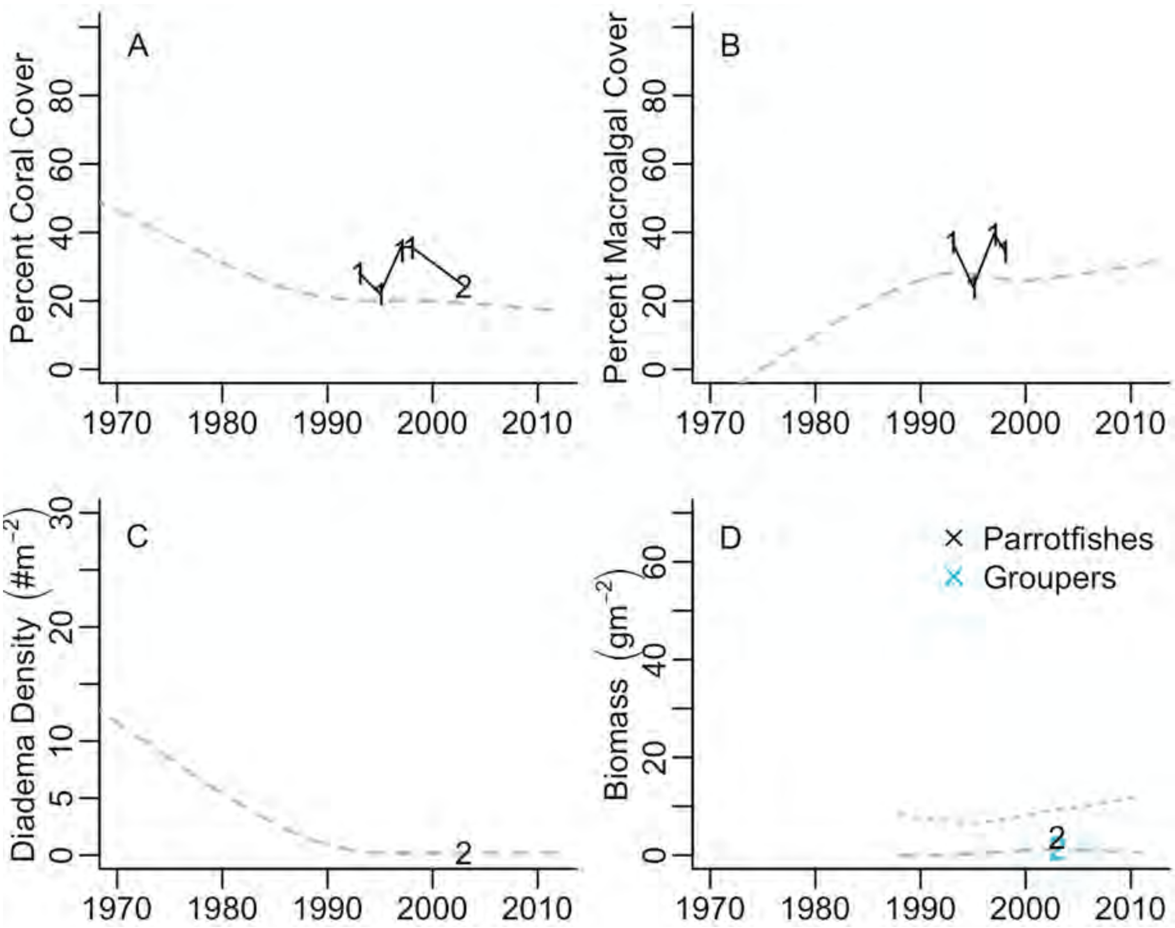

Fig. 24.2 Average percent cover of live corals and macroalgae, density of *Diadema antillarum*, and total fish biomass in Nicaragua. Dotted line represents the average of Caribbean data collected for this report. (Codes same as in Table 24.1 and Figure 24.1)

General Literature

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PANAMA

Coauthors: Katie Cramer, Héctor Guzmán, Juan Maté, Myra Shulman, Ernesto Weil, AGRRA, CARICOMP, IUCN Climate Change and Coral Reefs Marine Working Group (IUCN-CCCR) and Reef Check

Geographic Information

Coastal Length:	5,567 km
Land Area:	75,435 km^2
Maritime Area:	330,627 km^2
Reef Area:	1,521 km^2
Number of hurricanes in the past 20 years:	0

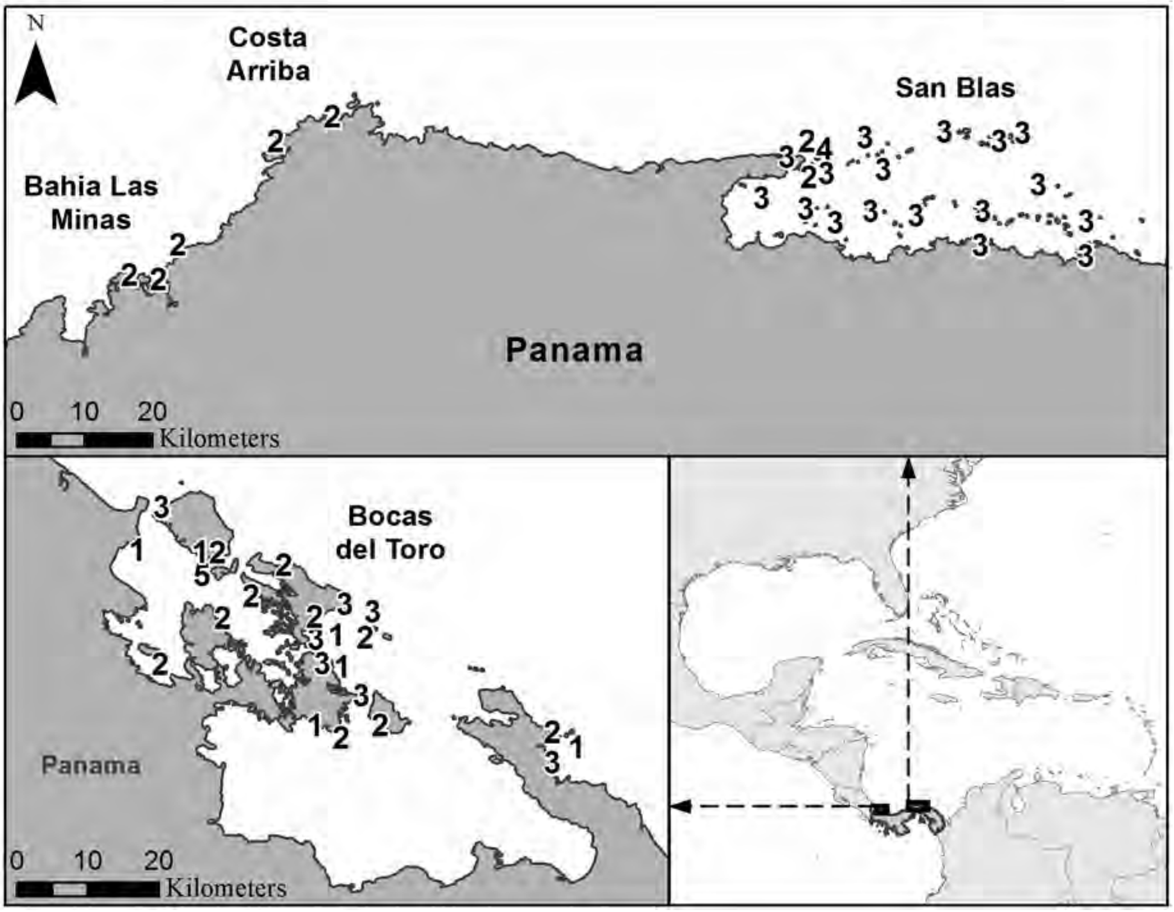

Fig. 25.1 Map of Panama, codes represent studies listed in Table 25.1. Top map includes Costa Arriba and San Blas, bottom left map includes Bocas del Toro. Missing map code(s) due to unavailable coordinates.

Table 25.1 Data sources from Panama used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 25.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Cramer, Katie*[1]	Bocas Del Toro	2008	1	X			
2	Guzmán, Héctor/CARICOMP*[2,3,4,5,6]	Bocas Del Toro; Costa Arriba; San Blas			X		X	
3	AGRRA*	Bocas Del Toro; San Blas	2002	1	X	X		X
4	Shulman, Myra*[7]	San Blas	1983-1990	8	X	X	X	
5	Weil, Ernesto*	Bocas Del Toro	2005-2006	2	X			
6	Cubit et al. 1986[8]	Punta Galeta	1973-1976	3		X		
7	Lessios 1988[9]	Galeta, San Blas	1982-1984	3		X		
8	Lessios et al. 1984[10]	Galeta, San Blas	1980-1984	4		X		
9	Lessios 2005[11]	San Blas	1980, 1982-1997, 1999-2003	20		X		
a	Lessios 1995[12]	San Blas	1982-1993	12		X		
b	Reef Check*		1997-1998, 2005	3		X		
c	Maté, Juan/ IUCN-CCCR*	Bocas Del Toro	2009	1	X			X

Fig. 25.2 Average percent cover of live corals and macroalgae, density of *Diadema antillarum*, and total fish biomass in Panama. Dotted line represents the average of Caribbean data collected for this report. (Codes same as in Table 25.1 and Figure 25.1)

Timeline

Late-1500s: Large-scale extraction of coral colonies in Costa Arriba for construction of Spanish fortresses and settlements

1870s: Excavation and dredging of reefs in Bahia Las Minas, Costa Arriba for construction of Panama Canal

Early 1800s: Coral mining and land filling in San Blas by Kuna Indians to expand island areas

Early-1900s: Dredging of coral reefs and filling in of mangrove swamps in Bahia Las Minas for construction of US military facilities

1958-1974: Dredging of coral reefs in Bahia Las Minas for construction of oil refinery

1968: Oil spill in Bahia Las Minas from grounding of tanker *Witwater*

1980s: Mass mortality of *Acropora* species across Panama

1982/1983: Coral bleaching on San Blas reefs

1983: Mass mortality of *Diadema antillarum* across Panama

1986: Oil spill in Bahia Las Minas, Costa Arriba

1987: Coral mortality event (mostly affecting *Acropora* species) in Costa Arriba

Mid-1980s: Octocoral mortality event from Aspergillosis (fungal disease)

Late-1980s: White-band, black-band, and yellow-blotch/yellow-band coral disease outbreaks

2010: Anoxic event causing mortality of reef corals and other sessile benthic organisms within Bahia Almirante, Bocas del Toro

General Literature

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PUERTO RICO

Coauthors: Richard Appeldoorn, Roy Armstrong, David Ballantine, Chris Caldow, Sara Rivero Calle, Christopher F.G. Jeffrey, Héctor Ruiz, Ernesto Weil, AGRRA, CARICOMP and NOAA Biogeography Branch

Geographic Information

Coastal Length:	1,087 km
Land Area:	9,000 km^2
Maritime Area:	204,942 km^2
Population:	3,940,410
Reef Area:	471 km^2
Number of hurricanes in the past 20 years:	4

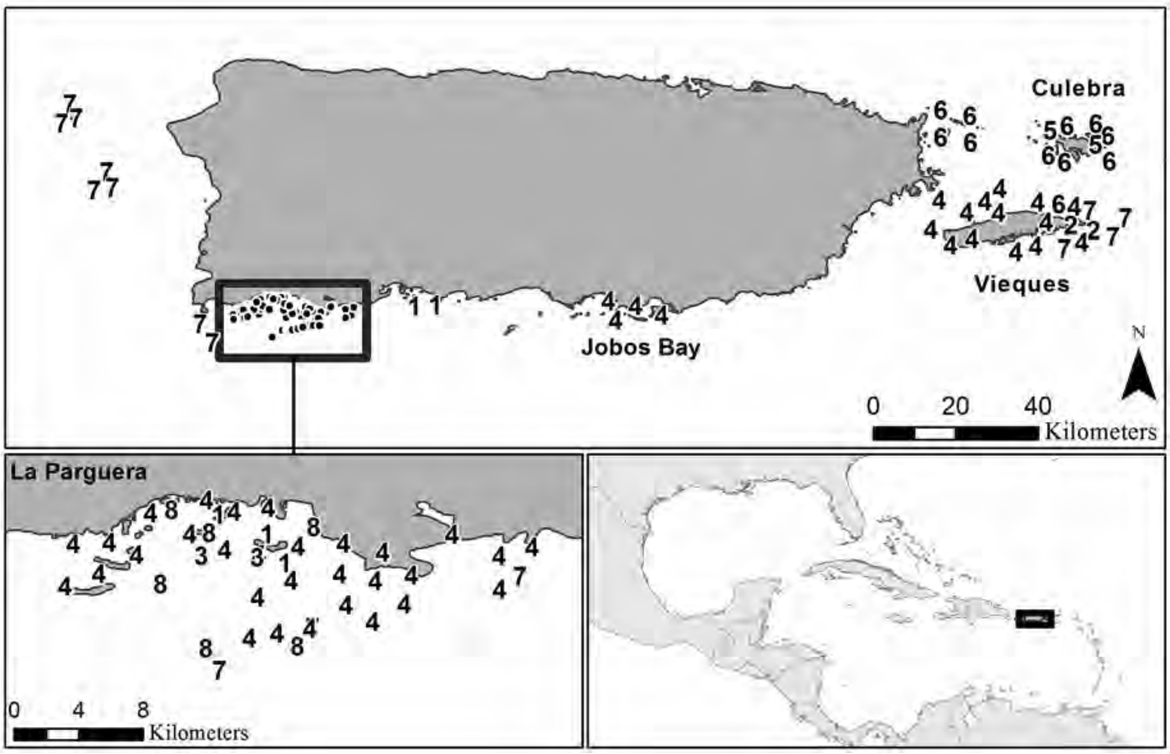

Fig. 26.1 Map of Puerto Rico, codes represent studies listed in Table 26.1. Missing map code(s) due to unavailable coordinates.

Table 26.1 Data sources from Puerto Rico used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 26.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Acevedo 1989[1]	La Parguera	1989	1	X			
2	Antonius & Weiner 1982[2]	Vieques	1978	1	X		X	
3	Weil, Ernesto; Garcia, Jorge/ CARICOMP*[3,4]	La Parguera	1994-2012	19	X	X	X	
4	NOAA Biogeography Branch*	La Parguera, Vieques, Jobos Bay	2001-2011	11	X	X	X	X
5	Garrison et al 2005[5]	Vieques	1991, 1994, 1998	3	X		X	
6	AGRRA*	Vieques & Culebra	2003	1	X	X		X
7	Armstrong, Roy; Rivero Calle, Sara*[6]	Vieques, Mona Island, La Parguera	2004, 2008	2	X		X	
8	Weil, Ernesto*[7]	La Parguera, Guanica, Turrumote	2003, 2005-2007	4	X		X	
9	Bauer 1980[8]	San Juan	1977	11		X		
a	McGehee 2008[9]	La Parguera	1995, 2005	2		X		
b	Miller et al. 2009[10]	La Parguera	2006	1		X		
c	Ruiz, Héctor; Ballantine, David*[11]	La Parguera	2003-2007	5			X	

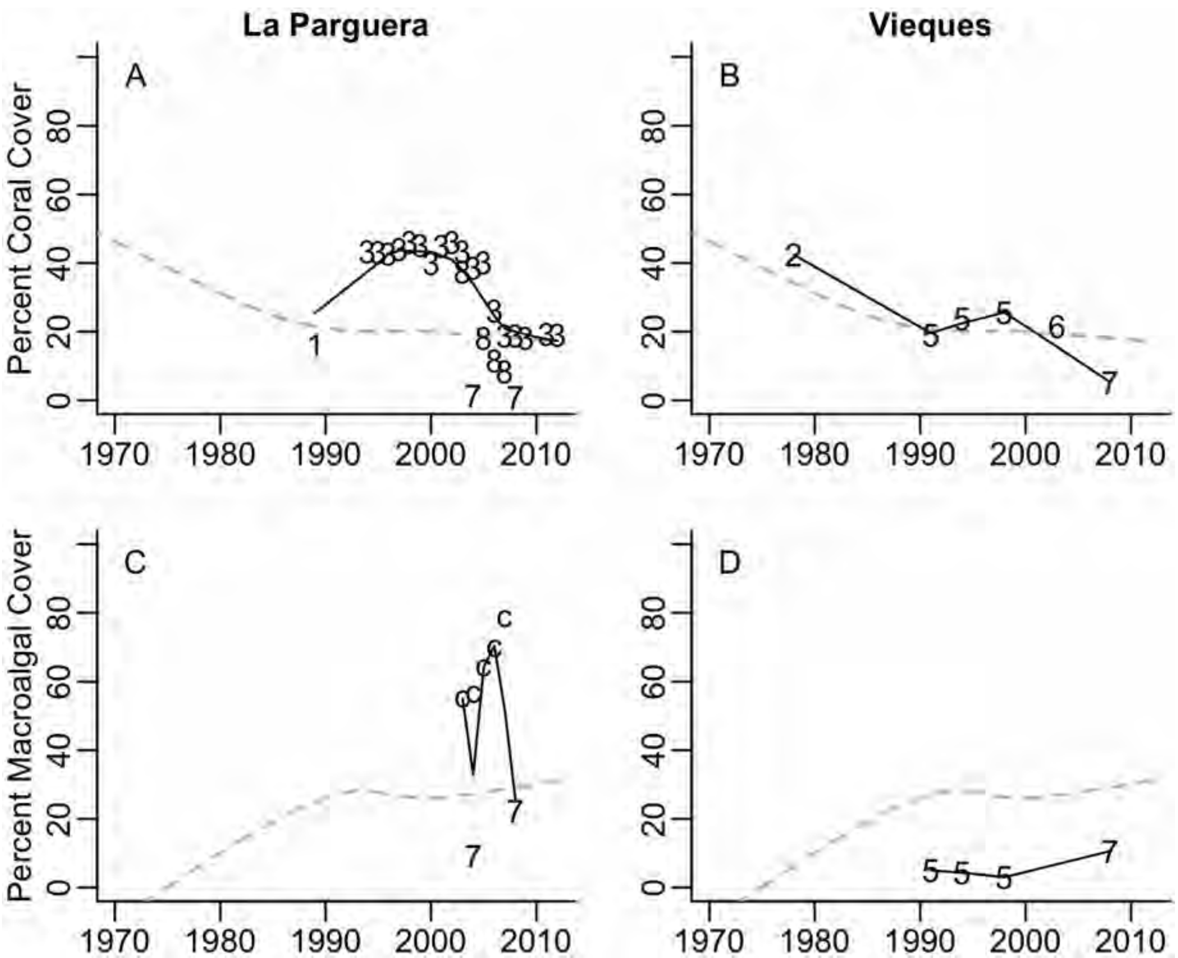

Fig. 26.2 Average percent cover of live corals and macroalgae for La Parguera (A, B) and Vieques (B, D) Puerto Rico. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 26.1 and Figure 26.1)

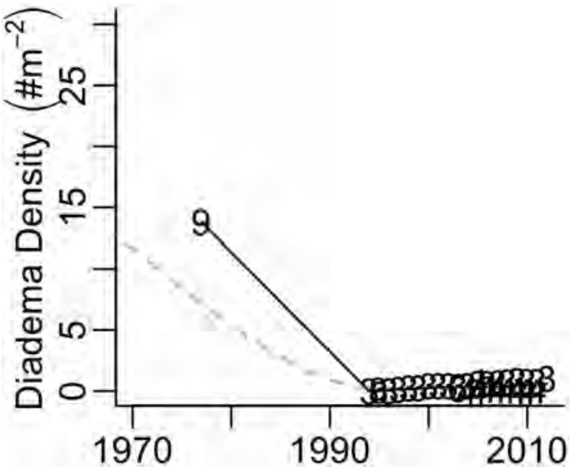

Fig. 26.3 Average density of *Diadema antillarum* for all Puerto Rico. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 26.1 and Figure 26.1)

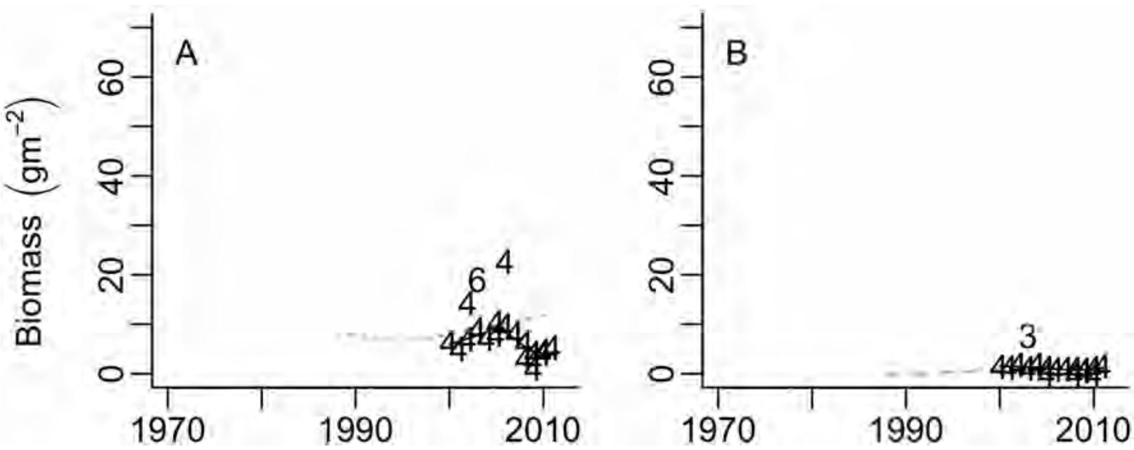

Fig. 26.4 Average biomass of parrotfishes (A) and groupers (B) for all Puerto Rico. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 26.1 and Figure 26.1)

Timeline

1969:	First documentation of bleaching event
1979:	Hurricane David; tropical storm Frederic
1980-1983:	White band disease in *A. cervicornis* and *palmata*; Hurricane Allen (Category 5); minor bleaching event
1981:	Tropical storm Gert; minor bleaching event
1983-84:	Mass mortality of *Diadema antillarum*
1984:	Tropical storm Klaus
1987-88:	Mild bleaching event, associated mortality occurred
1989:	Hurricane Hugo, caused extensive damage on eastern and northern coasts
1990:	Coral bleaching event
1995:	Hurricane Luis, affected north coast; Hurricane Marilyn, affected east and north coast; coral bleaching event
1996:	Hurricane Bertha, affected eastern and northern coasts; Hurricane Hortense, worst hurricane to hit since Hurricane Hugo, south coast great damage due to heavy rains and sediment outflow to reefs
1998-1999:	Bleaching event, high numbers but no mortality
1998:	Hurricane Georges, high *Acropora* mortalities

1999:	Tropical storm Jose; hurricane Lenny; minor bleaching; Yellow Band Disease (YBD), White Plague (WP) and Black Band Disease (BBD)
2000:	Hurricane Debby
2001:	Minor bleaching event; YBD and WP
2003:	Bleaching event affected colonies >10m
2004:	Tropical storm Jeanne; localized disease outbreaks of WBD and YBD
2005:	Major bleaching event (affecting 50-80% corals) and associated mortalities of acroporids, agariciids and *Mycetophyllia*; first report of Crustose Coralline White Syndrome (CCWS) affecting crustose coralline algae; major outbreak of WPD in the southwest coast
2006:	WPD expansion but stopped by end of winter; outbreak of YBD killed *Montastrea* colonies affected by bleaching and WPD
2010:	Caribbean wide bleaching event, no mortalities or disease outbreak

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SABA, ST. EUSTATIUS & ST. MAARTEN

Coauthors: Kenny Buchan, Adolphe Debrot, Paul Hoetjes, AGRRA and CARICOMP

Geographic Information

	Saba	St. Eustatius	St. Maarten
Coastal Length (km):	16	21	27
Land Area (km^2):	13	23	37
Maritime Area (km^2):	10,367	1,591	434
Population:	1,484	3,384	37,539
Reef Area (km^2):	214	12	5

Number of hurricanes In the past 20 years: 7 (Saba and St. Eustatius)

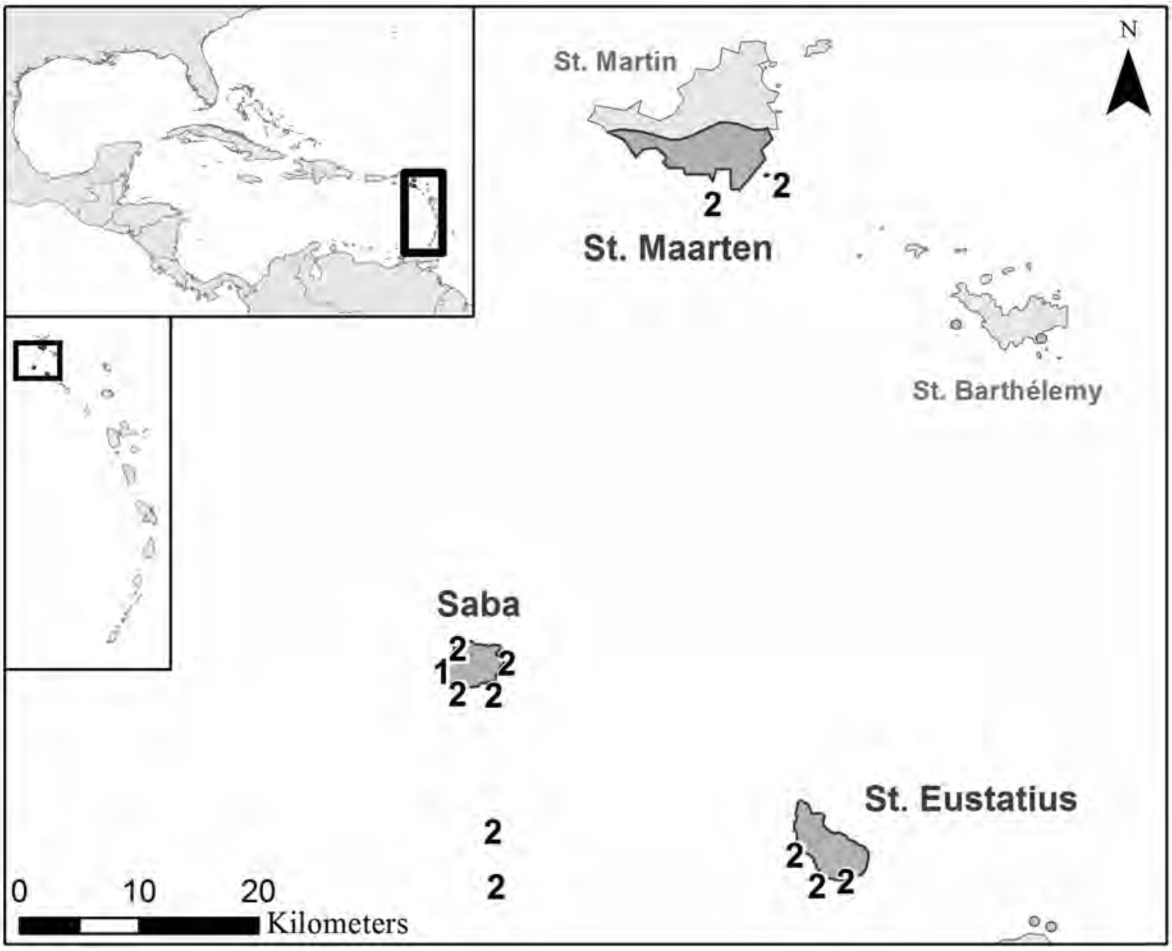

Fig. 27.1 Map of Saba, codes represent studies listed in Table 27.1. Missing map code(s) due to unavailable coordinates.

Table 27.1 Data sources from Saba. Map codes represent individual studies. For exact location of study, refer to Fig. 27.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Kenny, Buchan/ CARICOMP*	Saba	1993-1998, 2003	6	X	X	X	
2	AGRRA*[2]	Saba, St. Eustatius, St. Maarten	1999	1	X			

Timeline:

1987: Saba Marine Environmental Ordinance establishes the Saba Marine Park

1996: St. Eustatius Marine Environmental Ordinance establishes the Statia Marine Park

2003: Nature Conservation Ordinance St. Maarten

2005: Extensive bleaching event

2008: Saba Bank Management Plan presented

2010: Man-O-War Shoals Marine Park, St. Maarten established

2010: National Decree establishes the Saba Bank as SPAW protected area

2010: Saba Bank biological surveys led by Conservation International is published as special edition in Plos ONE (Hoetjes & Carpenter 2010).

2010: Dutch Ministry of Economic Affairs, Agriculture and Innovations drafts a marine biological resource management plan for the Dutch Caribbean EEZ

2011: St. Maarten declares indefinite temporary moratorium on the taking of sharks to protect its shark dive tourism

2012: The IMO declares Saba Bank as a Particularly Sensitive Sea Area (PSSA)

2013: The Dutch Ministry of Economic Affairs presents plans to establish a Dutch Caribbean marine mammal sanctuary and to develop and implement a protection plan for elasmobranchs

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ST. KITTS & NEVIS

Coauthors: Althea Athurton, Graeme Browne, AGRRA, Khaled bin Sultan Living Oceans Foundation and MACC

Geographic Information

Coastal Length:	122 km
Land Area:	266 km^2
Maritime Area:	9,818 km^2
Population:	37,477
Reef Area:	82 km^2
Number of hurricanes in the past 20 years:	2

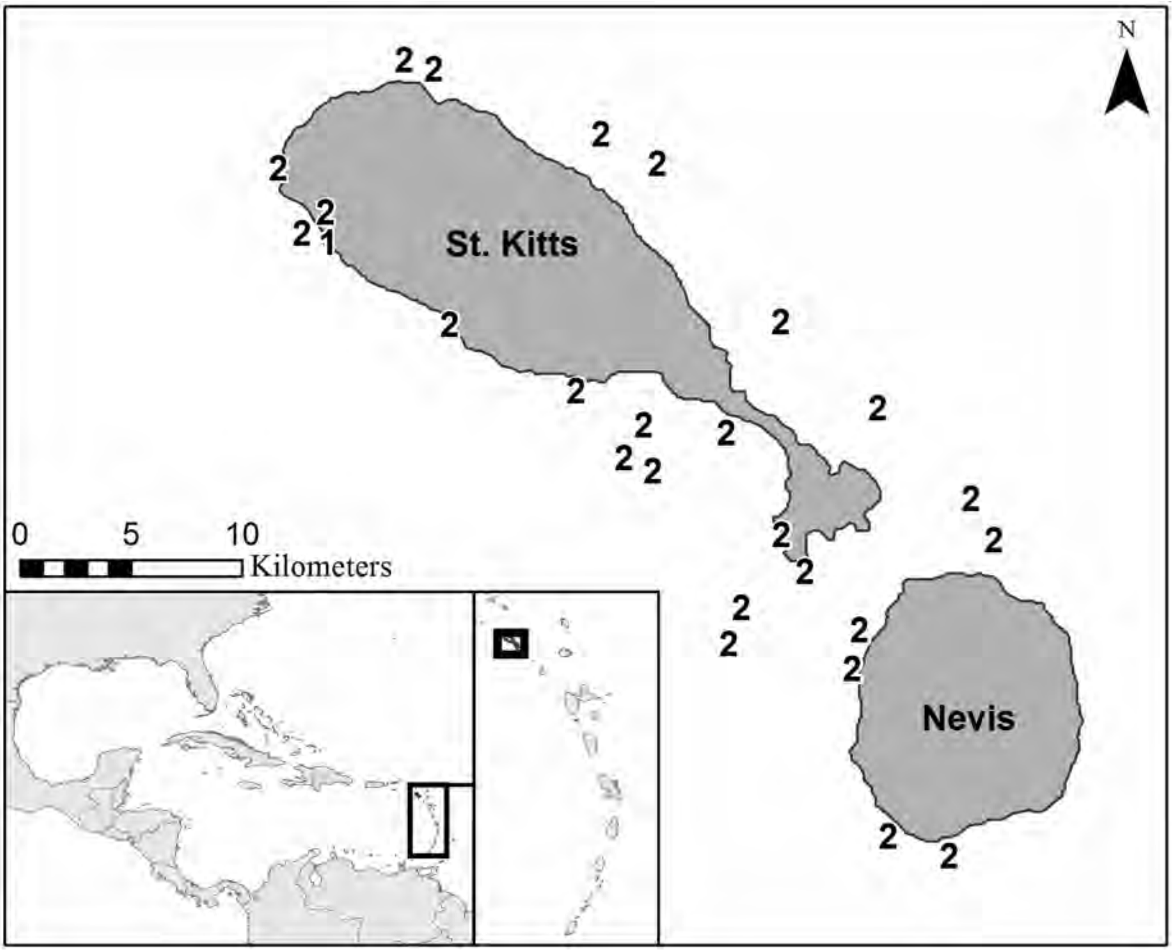

Fig. 28.1 Map of St. Kitts & Nevis, codes represent studies listed in Table 28.1. Missing map code(s) due to unavailable coordinates

Table 28.1 Data sources from St. Kitts & Nevis used in current synthesis. Map codes represent individual studies. For exact location of study, refer to Fig. 28.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Browne, Graeme/ MACC*[1,2]	St. Kitts	2007, 2009	2	X		X	
2	AGRRA/LOF*[3]	St. Kitts & Nevis	2011	1	X	X	X	X
3	Reef Check*	St. Kitts	2004	1		X		

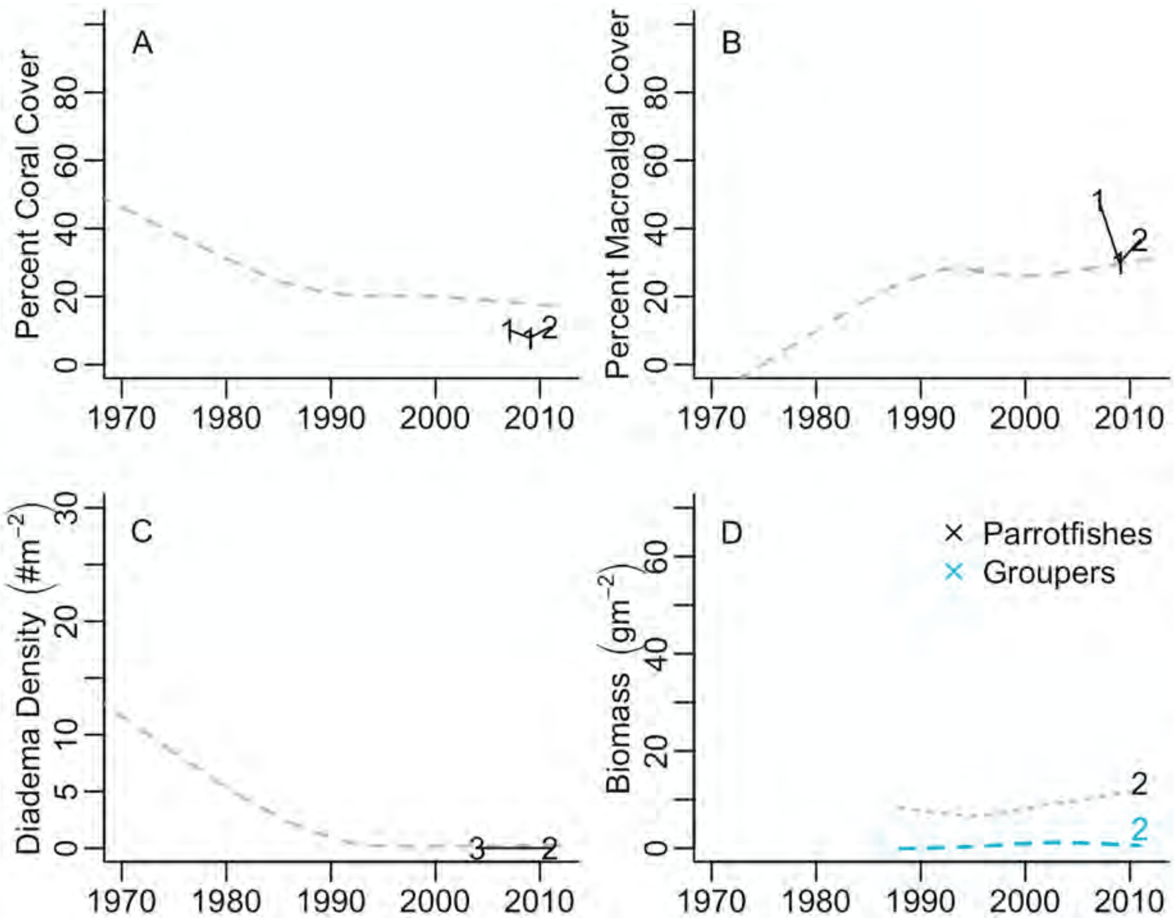

Fig. 28.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C), and biomass of parrotfishes and groupers (D) in St. Kitts & Nevis. Dotted line represents the average of Caribbean data collected for this report. (Codes same as in Table 28.1 and Figure 28.1)

Published Data Sources

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ST. LUCIA

Coauthors: Douglas Fenner, Sarah George,
St. Lucia Fisheries Department,
MACC and Reef Check

Geographic Information

Coastal Length:	163 km
Land Area:	623 km^2
Maritime Area:	15,417 km^2
Population:	172,208
Reef Area:	60 km^2
Number of hurricanes in the past 20 years:	1

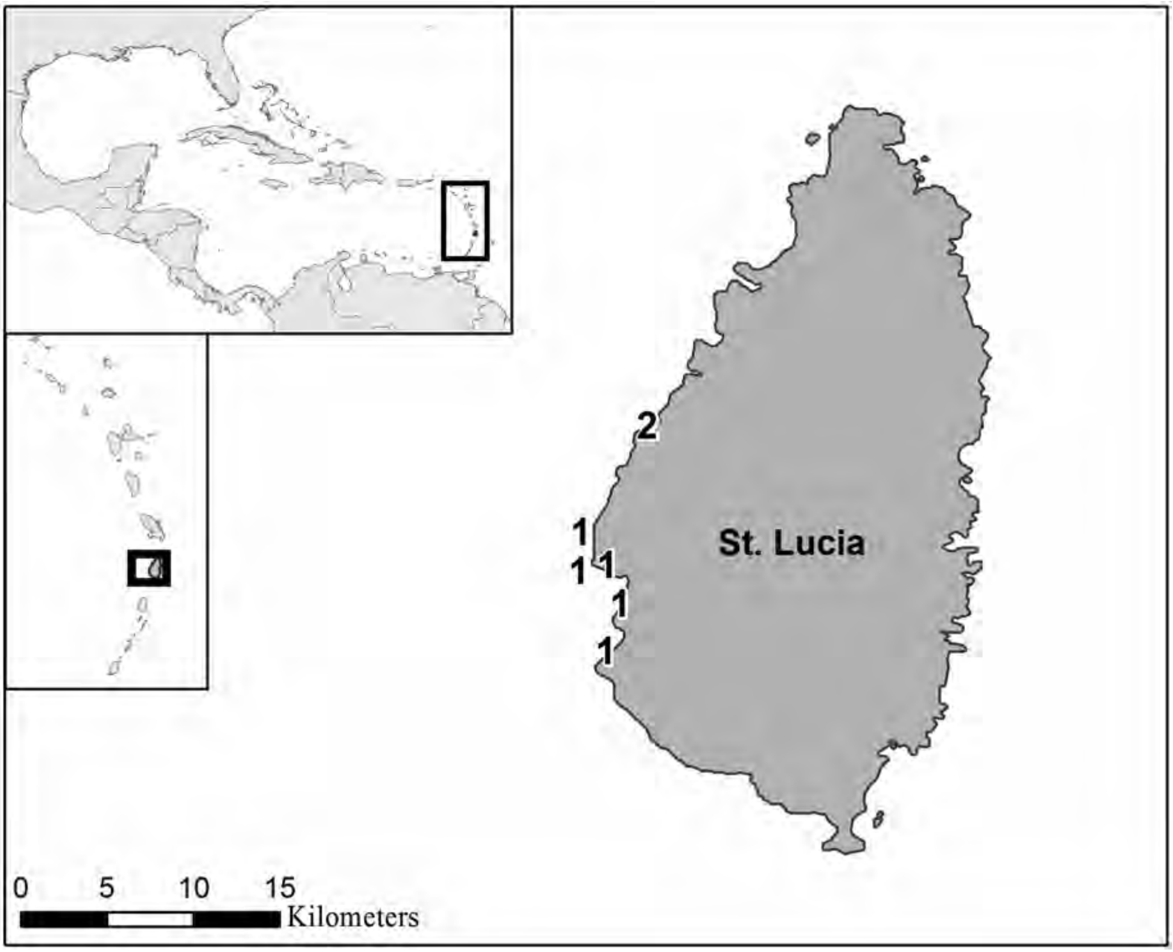

Fig. 29.1 Map of St. Lucia, codes represent studies listed in Table 29.1. Missing map code(s) due to unavailable coordinates.

Table 29.1 Data sources from St. Lucia used in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 29.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroal-gae	Fishes
1	MACC*[1,2]	2007, 2009	2	X		X	
2	Fenner, Douglas*[3]	1993	1	X			
3	Reef Check*[4]	1999-2007	9		X		

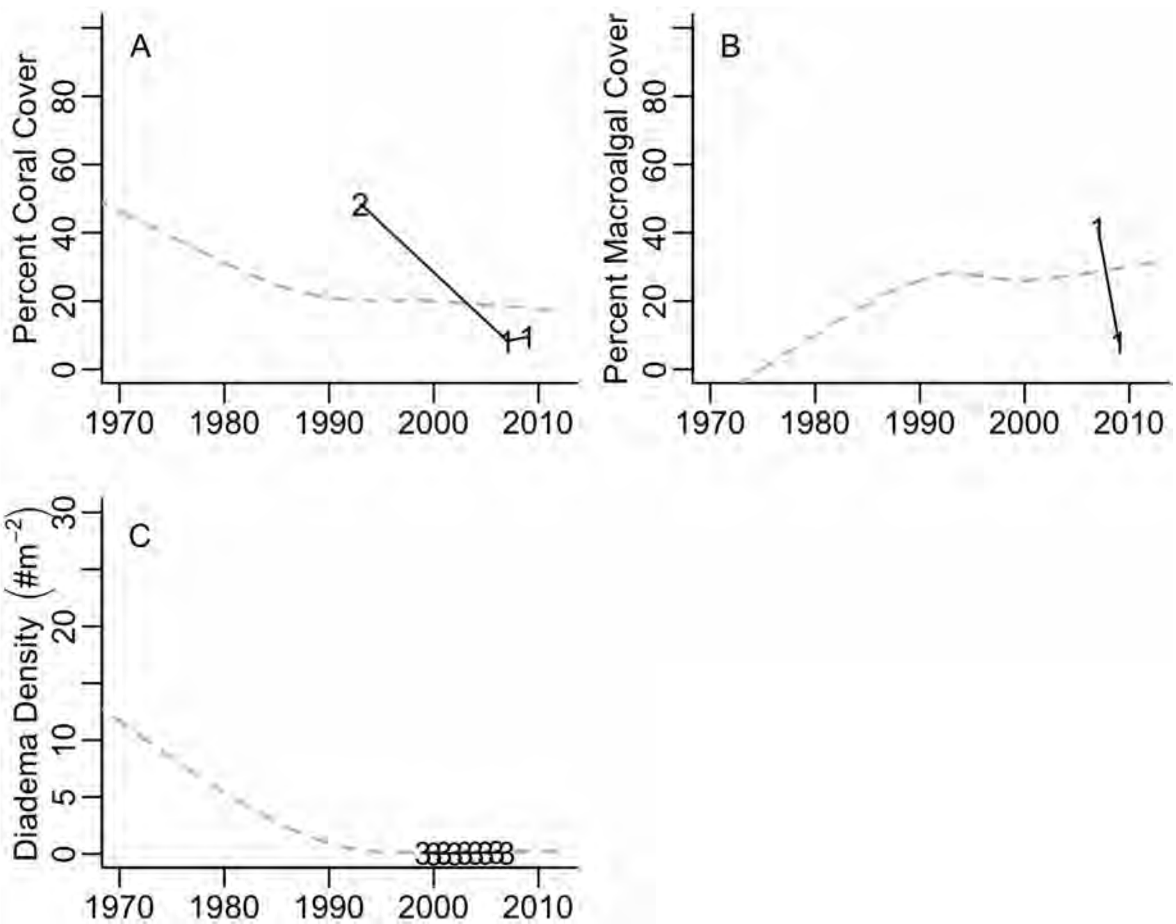

Fig. 29.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C) in St. Lucia. Dotted line represents the average of Caribbean data collected for this report. (Codes same as in Table 29.1 and Figure 29.1)

Timeline

1955:	Hurricane Janet
1950s-1970s:	Reefs ecosystems, as well as mangroves and seagrass beds in a healthy state; coral diversity and cover are high, with relatively few and low abundance of macroalgae; healthy populations of reef herbivores such as sea urchins Diadema; well balanced food chain with wide size ranges in various fish species and healthy levels of abundance of top predators (e.g., groupers; large snappers; barracuda's etc.). Fishing fleet focused mainly on near-shore fishing using wooden transoms and canoes propelled by sail, oar and small outboard engines
1960:	Hurricane Abby
1963:	Hurricane Edith
1975-1980:	Transition from the traditional wooden canoe to the more stable, fiberglass pirogue (mostly 12–25 feet) with outboard engines of 75–100 Hp
1980:	Hurricane Allen
1980s-1990s:	Change of focus for fishing fleet from near-shore fishing to offshore fishing for migratory pelagic fishes such as tunas, dolphinfish and wahoo (which started to contribute approximately 65-75% to the annual catch)
1983:	Mass mortality of *Diadema antillarum* on local reefs (over 65% mortality)
1980-1995:	significant expansion of tourism (including cruise ship arrivals and mass water-based tourism in certain locations) and banana cultivation; associated increased levels of and impacts from sedimentation in coastal areas which led to significant reef decline in many coastal areas (particularly northwest to central west coastal areas)
1984-1994:	new fisheries laws enacted with mesh size restrictions for traps and nets; size limits and close seasons for many species, e.g., lobster, conch, turtles and sea urchins; prohibitions on use of toxins and dynamite; restrictions on use of spearguns, etc.
1986:	a number of marine reserves established (many coral reefs; 2 turtle nesting beaches; several mangrove areas) but no demarcation and little day-to-day management put in place following their declaration

1990-2012: Fish Aggregating Devises tested and implemented as a means of attracting fishers away from reefs which were becoming more stressed and degraded by a combination of increased sedimentation, poor fishing practices; large–scale water-based tourism, other forms of coastal pollution and successive severe weather impacts

1992-1997: influx into the fisheries sector due to declines in the banana industry; resultant increasing fishing pressure in coastal areas, particularly during the low season when migratory pelagic fish species are not readily available (approximately July to December)

1994: Soufriere Marine Management Area is established: an 11km stretch of coastal marine space with marine reserves, multiple use areas, yacht mooring zones and recreational zones created to bring about integrated resource use and conflict resolution, sustainable resource use and reef recovery

1994: Tropical Storm Debby, very heavy rainfall event causing unprecedented flooding, sedimentation, further coral reef decline and increased algal growth on reefs

1995: Oil spill in the Cul de Sac Bay with undetermined impacts in sensitive coastal areas on the west coast

1998-99: Bleaching event in the region, including St Lucia1999: Hurricane Lenny – causes high wave activity, significant coastal destruction and physical impacts on reefs along the western coast of St Lucia.1999-2004: Reef Check assessments during this period showed a decrease in live coral cover and a general increase in rock/dead coral cover; groupers greater than 30cm only recorded on marine reserves within the SMMA; deeper reefs had a higher mean number of these fish than shallower reefs areas, with marine reserves in the SMMA having higher mean numbers than reefs in other parts of the island

2002: Tropical Storm Lili

2004: Hurricane Ivan (veered south to hit Grenada)

2005: Coral Bleaching: (40% - 83% coral was bleached at monitored sites) causes considerable mortality and resulted in further algal

2007: Point Sable Environmental Protection Area established – a coastal protected area with two of St Lucia's largest mangroves, a coral reef marine reserve, two offshore islands, extensive seagrass beds and fringing reef habitats

2010. Pitons Management Area established (later designated a World Heritage Site) with a marine component including some of the island's most valued reef habitats

2010: Hurricane Tomas – extreme rainfall event causing numerous landslides and extensive sedimentation

2011: Lionfish *Pterois volitans* (an invasive species) first recorded in local waters

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Published Data Sources

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2 Creary MC (2008) Coral Reef Monitoring for the Organization of Eastern Caribbean States and Tobago - Status of the Coral Reefs. Caribbean Community Climate Change Centre (CCCCC), Mainstreaming Adaptation to Climate Change (MACC). Kingston, Jamaica: Caribbean Coastal Data Centre, Centre for Marine Sciences, University of the West Indies Mona Campus. 94 p.

3 Fenner DP (1998) Reef topography and coral diversity of Anse Galet Reef, St. Lucia. Caribbean Marine Studies 6: 8.

4 Department of Fisheries (2004) Report on Reef Check Monitoring Programme in Saint Lucia (1999-2004). St. Lucia: Ministry of Agriculture, Lands, Fisheries and Forestry.

ST. VINCENT & THE GRENADINES

Coauthors: AGRRA, MACC and Reef Check

Geographic Information	
Coastal Length:	257 km
Land Area:	409 km^2
Maritime Area:	36,062 km^2
Population:	117,347
Reef Area:	85 km^2
Number of hurricanes in the past 20 years:	2

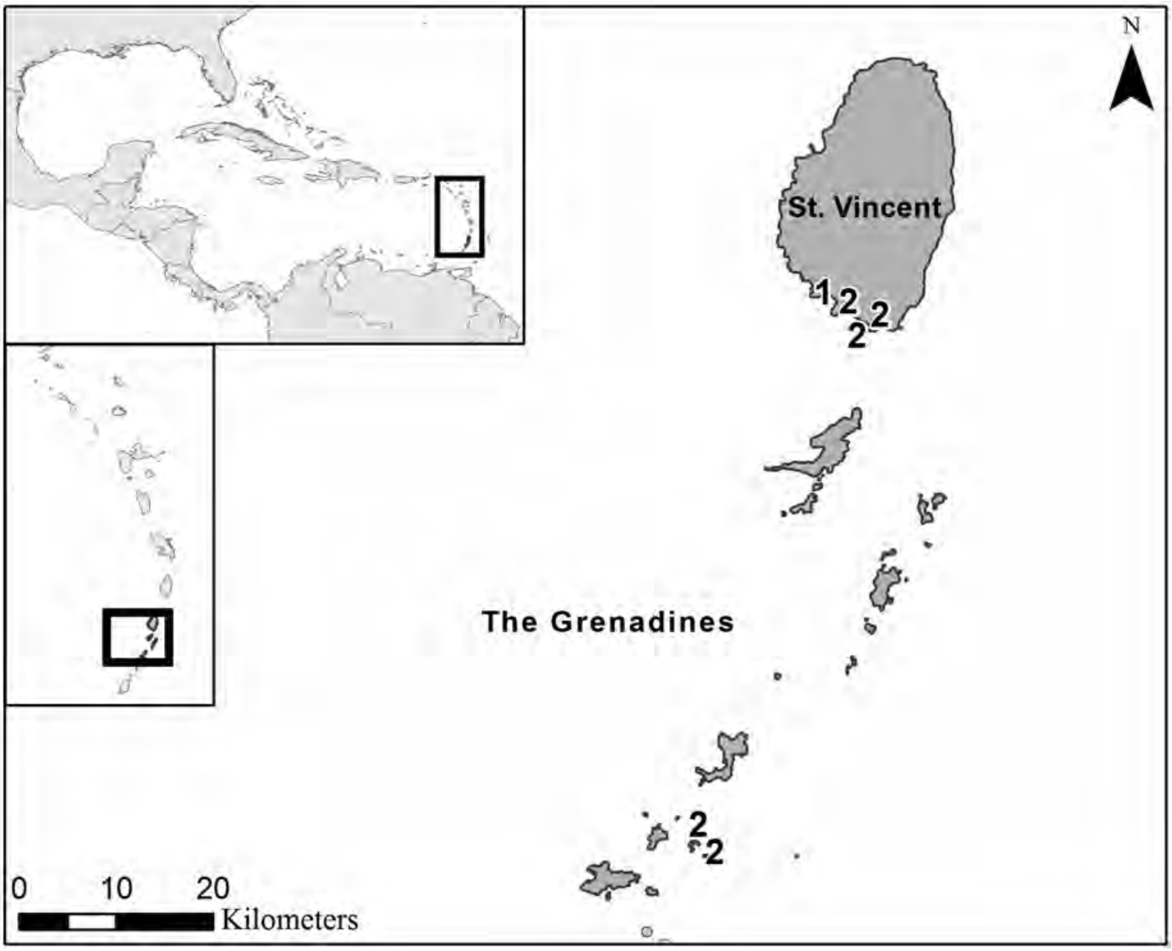

Fig. 30.1 Map of St. Vincent & the Grenadines, codes represent studies listed in Table 30.1. Missing map code(s) due to unavailable coordinates.

Table 30.1 Data sources from St. Vincent & the Grenadines. Map code represents individual studies. For exact location of study, refer to Fig. 30.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	MACC*1,2	2007, 2009	2	X		X	
2	AGRRA*3	1999, 2008	2	X	X		X
3	Reef Check*	2004-2005, 2007	3		X		

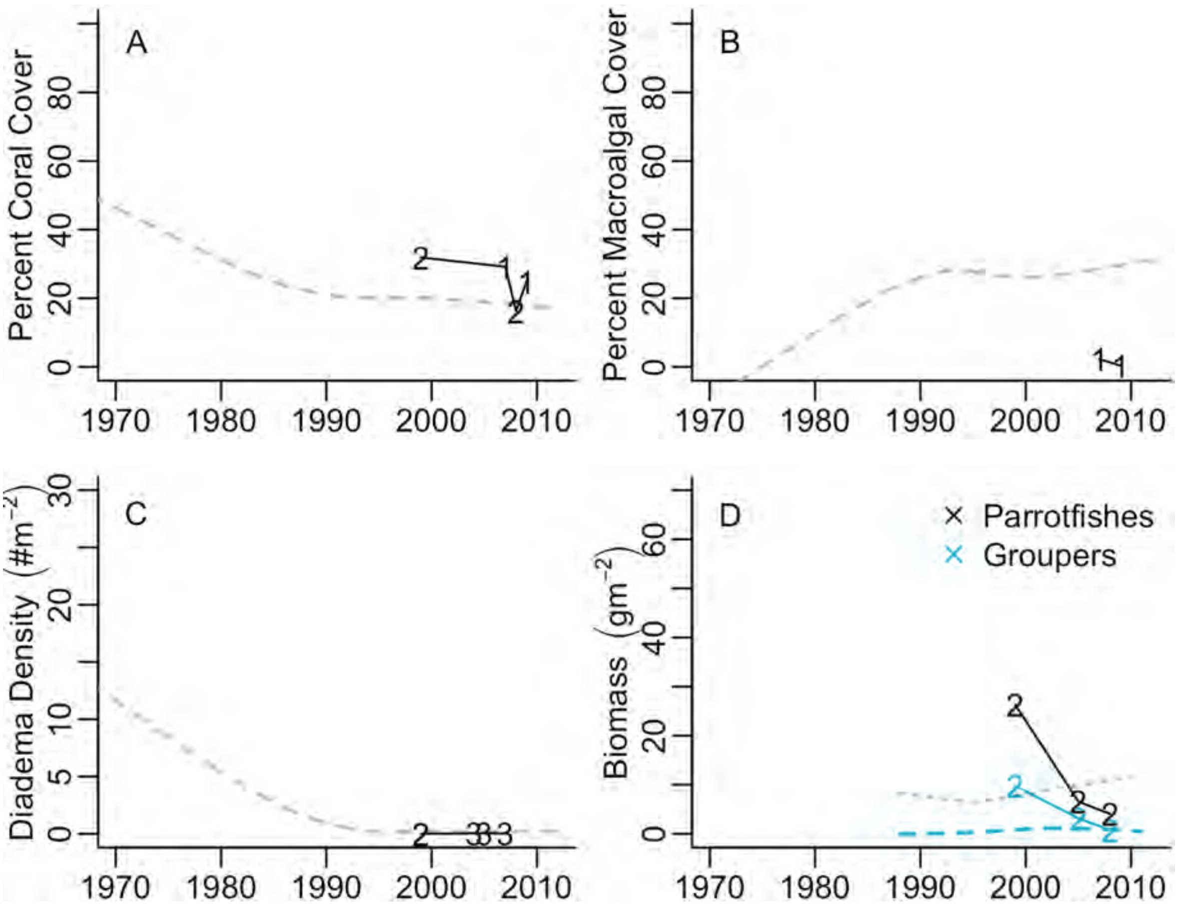

Fig. 30.2 Average percent cover of live corals (A) and macroalgae (B), density of *Diadema antillarum* (C), and biomass of parrotfishes and groupers (D) in St. Vincent & the Grenadines. Dotted line represents the average of Caribbean data collected for this report. (Codes same as in Table 30.1 and Figure 30.1)

General Literature

Adams RD (1968) The leeward reefs of St. Vincent, West Indies. The Journal of Geology 76.

Published Data Sources

[1] Creary M (2009) Coral Reef Monitoring for the Organization of Eastern Caribbean States and Tobago - Year 2. Caribbean Community Climate Change Centre (CCCCC), Mainstreaming Adaptation to Climate Change (MACC). Mona, Jamaica: The University of the West Indies

[2] Creary MC (2008) Coral Reef Monitoring for the Organization of Eastern Caribbean States and Tobago - Status of the Coral Reefs. Caribbean Community Climate Change Centre (CCCCC), Mainstreaming Adaptation to Climate Change (MACC). Kingston, Jamaica: Caribbean Coastal Data Centre, Centre for Marine Sciences, University of the West Indies Mona Campus. 94 p.

[3] Deschamps A, Desrochers A, Klomp KD (2003) A rapid assessment of the Horseshoe Reef, Tobago Cays Marine Park, St. Vincent, West Indies (Stony corals, algae and fishes). In: Lang JC, editor. Status of Coral Reefs in the Western Atlantic: Results of Initial Surveys, Atlantic and Gulf Rapid Reef Assessment (AGRRA) Program Atoll Research Bulletin 496: 439-458.

TRINIDAD & TOBAGO

Coauthors: Jahson Alemu, AGRRA, Institute of Marine Affairs, MACC and Reef Check

Geographic Information

Coastal Length:	698 km
Land Area:	5,178 km^2
Maritime Area:	73,258 km^2
Population:	1,043,790
Reef Area:	76 km^2
Number of hurricanes in the past 20 years:	1

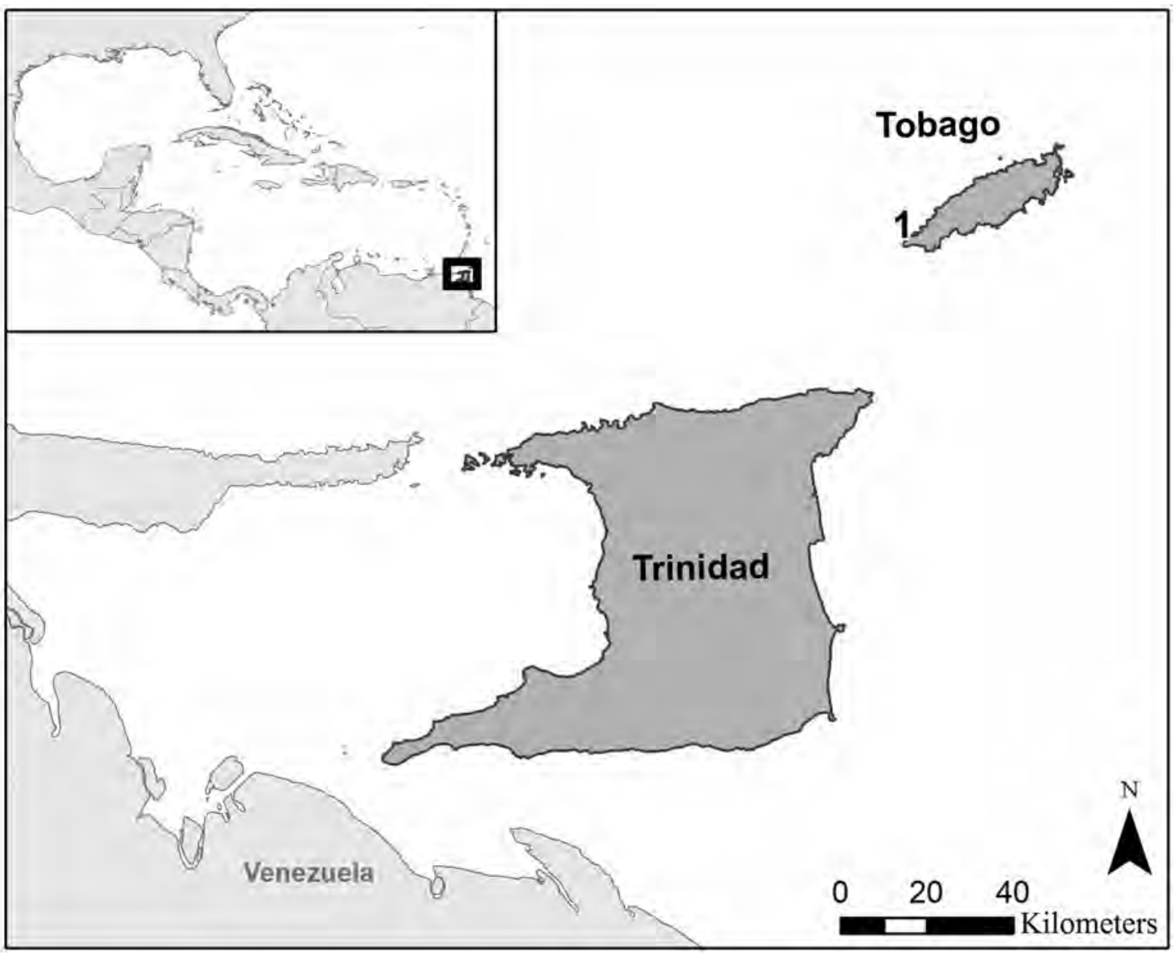

Fig. 31.1 Map of Trinidad & Tobago, codes represent studies listed in Table 31.1. Missing map code(s) due to unavailable coordinates.

Table 31.1 Data sources from Trinidad and Tobago. Map codes represent individual studies. For exact location of study, refer to Fig. 31.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Alemu, Jahson /CARICOMP*[1]	Buccoo Reef	1994-1998, 2000-2008, 2011-2012	16	X	X	X	
2	Bauer 1980[2]	Buccoo Reef	1979	1		X		
3	Lessios 1988[3]	Buccoo Reef	1983-1984	2		X		
4	Reef Check*		2007	1		X		
5	MACC*[4,5,6]	Buccoo Reef	2007-2011	5	X		X	
6	Laydoo 1985[7]	Buccoo Reef	1985	1	X		X	

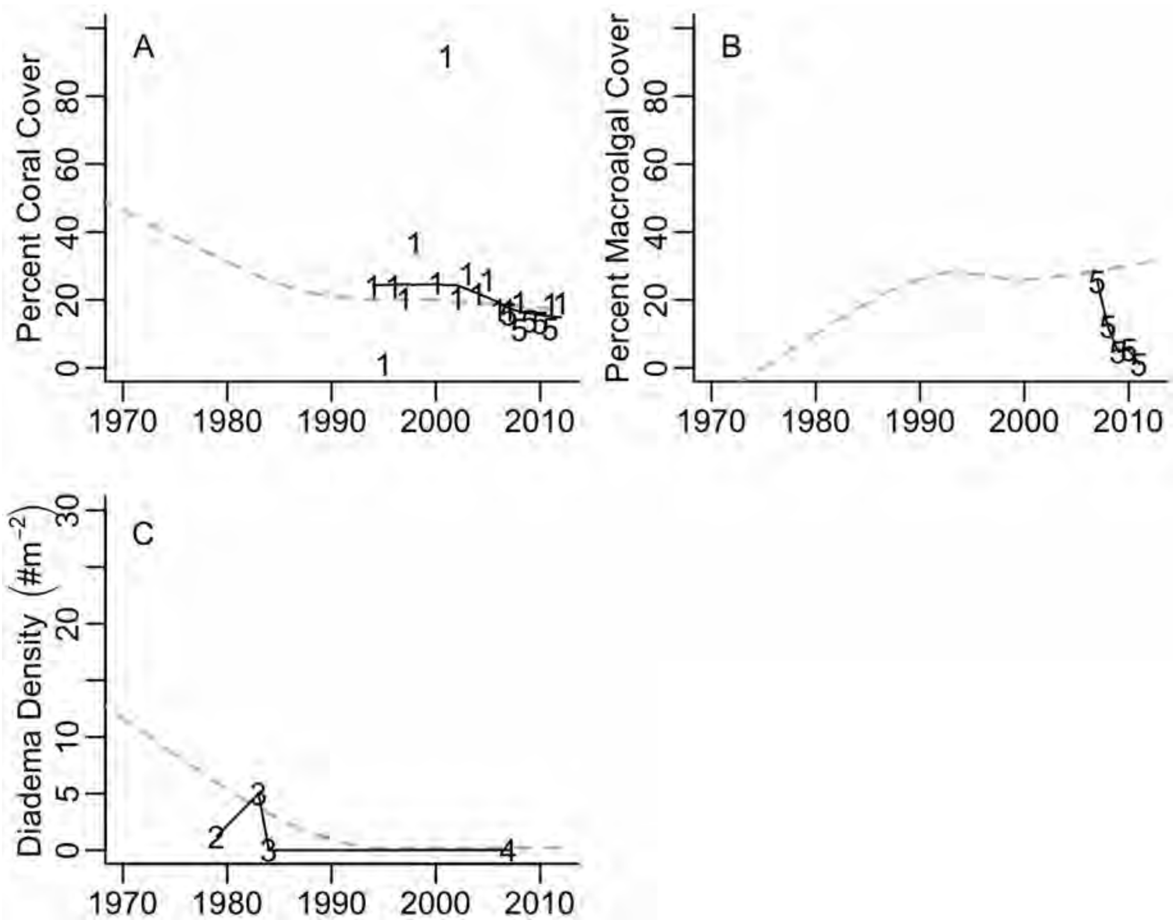

Fig. 31.2 Average percent cover of live corals (A) macroalgae (B), and density of *Diadema antillarum* (C) in Trinidad and Tobago. Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 31.1 and Figure 31.1)

Timeline

1963:	Hurricane Flora (Category 3)
1970s:	Onset of coral reef monitoring in Tobago - reef condition considered to be quite good
1973:	Declaration of Buccoo Reef as a restricted no take area (MPA, ~7km^2)
1980s-1990s:	Reef walking encouraged as a tourist activity
1983:	Mass mortality of *Diadema antillarum*
1988:	Tropical Storm Isaac

1998:	Mass coral bleaching
2004:	Hurricane Ivan (Category 3)
2002:	Mild coral bleaching event
2005:	Mass coral bleaching, mean of 66% of live hard coral cover affected, with levels over 85% observed at many sites
2008:	Mild coral bleaching event
2010:	Hurricane Tomas (Category 2)
2010:	Mass coral bleaching, affecting up to 77% of coral colonies with 10-30% coral bleaching associated mortality
2012:	Mild coral bleaching event, affecting >5% coral cover
2012:	First confirmed sighting of lionfish *Pterois volitans*
2013:	Mild bleaching event with 5-10% coral cover affected on the southern coast

General Literature

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Burke L, Greenhalgh S, Prager D, Copper E (2008) Coastal Capital - Economic Valuation of Coral Reefs in Tobago and St. Lucia. Washington DC: World Resources Institute. 76 p.

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[1] Laydoo RS, Bonair K, Alleng G (1998) Buccoo Reef and Bon Accord Lagoon, Tobago, Republic of Trinidad & Tobago. In: Kjerfve B, editor. CARICOMP - Caribbean coral reef, seagrass and mangrove sites Coastal region and small island papers 3. Paris: UNESCO. pp. 171-176.

[2] Bauer JC (1980) Observations on geographical variations in population density of the echinoid *Diadema antillarum* within the western north Atlantic. Bulletin of Marine Science 30: 509-515.

[3] Lessios HA (1988) Mass mortality of *Diadema antillarum* in the Caribbean - what we have learned. Annual Review of Ecology and Systematics 19: 371-393.

[4] Creary M (2009) Coral Reef Monitoring for the Organization of Eastern Caribbean States and Tobago - Year 2. Caribbean Community Climate Change Centre (CCCCC), Mainstreaming Adaptation to Climate Change (MACC). Mona, Jamaica: The University of the West Indies

[5] Creary MC (2008) Coral Reef Monitoring for the Organization of Eastern Caribbean States and Tobago - Status of the Coral Reefs. Caribbean Community Climate Change Centre (CCCCC), Mainstreaming Adaptation to Climate Change (MACC). Kingston, Jamaica: Caribbean Coastal Data Centre, Centre for Marine Sciences, University of the West Indies Mona Campus. 94 p.

[6] Creary MC (2011) Coral Reef Monitoring for the Organization of Eastern Caribbean States and Tobago. Status of the coral reefs of Tobago 2009-2010. Caribbean Community Climate Change Centre (CCCCC), Mainstreaming Adaptation to Climate Change (MACC). Mona, Jamaica: The University of the West Indies

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TURKS & CAICOS ISLANDS

Coauthors: Carrie Manfrino, Bernhard Riegl, AGRRA, CARICOMP and Reef Check

Geographic Information

Coastal Length:	827 km
Land Area:	1,018 km^2
Maritime Area:	148,471 km^2
Population:	21,522
Reef Area:	343 km^2
Number of hurricanes in the past 20 years:	4

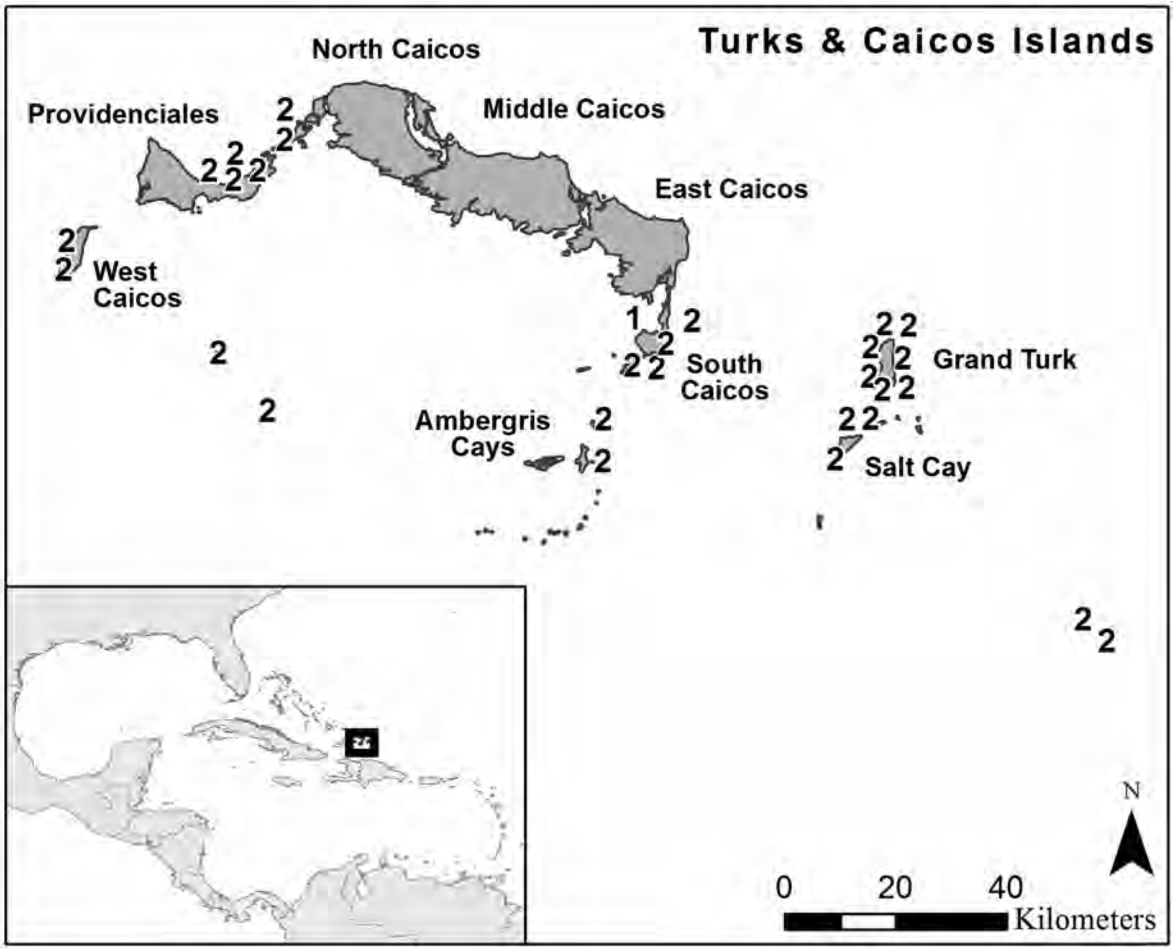

Fig. 32.1 Map of Turks & Caicos Islands, codes represent studies listed in Table 32.1. Missing map code(s) due to unavailable coordinates.

Table 32.1 Data sources from Turks & Caicos Islands in current study. Map codes represent individual studies. For exact location of study, refer to Fig. 32.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	CARICOMP*	1999	1	X		X	
2	Manfrino, Carrie, AGRRA *1,2	1999	1	X	X		X
3	Reef Check*	2004-2005, 2008	4		X		

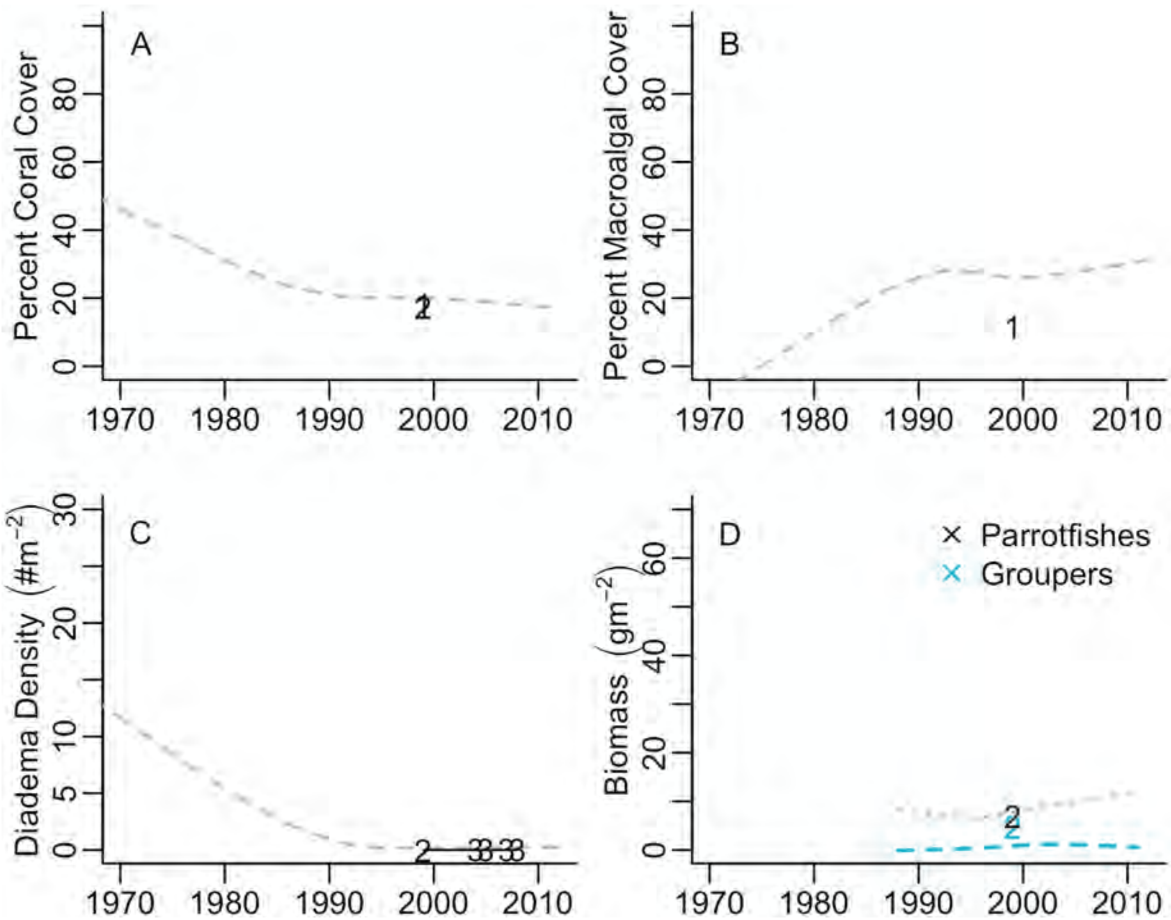

Fig. 32.2 Average percent cover of live corals (A) macroalgae (B), density of *Diadema antillarum* (C), and biomass of parrotfishes and groupers (D) in Turks & Caicos Islands. Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 32.1 and Figure 32.1)

Timeline

1950s-1970s: Reefs appeared healthy, coral cover & benthic density high, relatively few macroalgae

1970s-1980s: Mechanization of fishing fleets

1983/84: Mass mortality of *Diadema antillarum*

1990s: Rampant destructive (bleach) fishery especially on patch reefs on the banks

1998: Bleaching event, but reefs are not severely damaged

1999: White-band disease/white plague outbreaks severely reduce previously dense Montastraea populations

2005: Bleaching, affecting 45-75% coral cover

General Literature

Bene C, Tewfik A (2011) Fishing Effort Allocation and Fishermen's Decision Making Process in a Multi-Species Small-Scale Fishery: Analysis of the Conch and Lobster Fishery in Turks and Caicos Islands. Human Ecology 29: 157-186.

Dikou A, Ackerman C, Banks C, Dempsey A, Fox M, et al. (2009) Ecological assessment to detect imminent change in coral reefs of Admiral Cockburn Land and Sea National Park, Turks and Caicos Islands. Marine Ecology 30: 425-436.

Goreau TJ, Fisher T, Perez F, Lockhart K, Hibbert M, et al. (2008) Turks and Caicos Islands 2006 coral reef assessment: Large-scale environmental and ecological interactions and their management implications. Revista de Biología Tropical 56 (Suppl. 1): 25-49.

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[1] Hoshino K, Brandt M, Manfrino C, Riegl B, Steiner SCC (2003) Assessment of the coral reefs of the Turks and Caicos Islands (Part 2: fish communities). In: Lang JC, editor. Status of Coral Reefs in the Western Atlantic: Results of Initial Surveys, Atlantic and Gulf Rapid Reef Assessment (AGRRA) Program. Atoll Research Bulletin 496: 480-499.

[2] Riegl B, Manfrino C, Hermoyian C, Brandt M, Hoshino K (2003) Assessment of the coral reefs of the Turks and Caicos Islands (Part 1: stony corals and algae). In: Lang JC, editor. Status of coral reefs in the Western Atlantic: Results of Initial surveys, Atlantic and Gulf Rapid Reef Assessment (AGRRA) Program: Atoll Research Bulletin 496: 460-479.

US VIRGIN ISLANDS

Coauthors: Richard Appeldoorn, Roy Armstrong, Andrea Atkinson, Jim Beets, John Bythell, Chris Caldow, Peter Edmunds, Alan Friedlander, Barbara Kojis, Christopher F.G. Jeffrey, Don Levitan, Ian Lundgren, Jeff Miller, Richard Nemeth, Simon Pittman, Norman Quinn, Caroline Rogers, Tyler Smith, Bob Steneck, Jon Witman, AGRRA, CARICOMP, National Park Service South Florida/Caribbean Network, NOAA Biogeography Branch and Reef Check

Geographic Information	
Coastal Length:	378 km
Land Area:	370 km^2
Maritime Area:	5,895 km^2
Population:	101,328
Reef Area:	134 km^2
Number of hurricanes in the past 20 years:	7

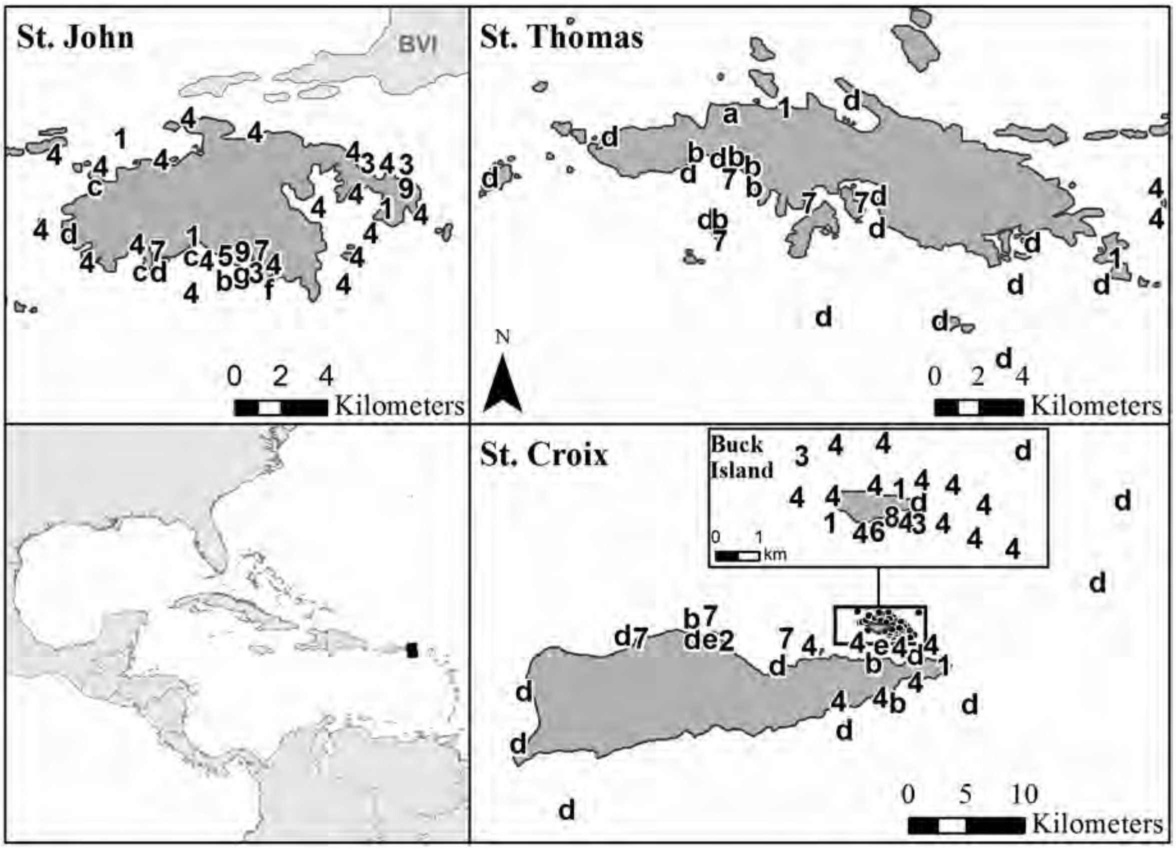

Fig. 33.1 Map of USVI showing studied sites. Codes represent individual studies, refer to Table 33.1.

Table 33.1 Data sources from USVI (Map code represent individual studies. For exact location of study, refer to Fig. 33.1; * denotes original data; for full references, refer to published literature sources in the last section)

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Antonius & Weiner 1982[1]	St. Croix, St. Thomas, St. John	1978	1	X		X	
2	Aronson et al. 1994[2]	St. Croix	1988, 1990	2	X			
3	National Park Service/SFCN*	St. Croix, St. John	2002-2011, 1999-2011	13, 3	X	X	X	
4	NOAA, Biogeography Branch*	St. Croix, St. John	2001-2011, 2008-2010	11, 3	X	X	X	X
5	Edmunds, Peter*[3,4,5,6,7,8,9,10,11]	St. John	1987-2011	25	X	X	X	
6	Gladfelter et al. 1977[12]	St. Croix	1976	1	X			
7	AGRRA*[13,14]	St. Croix, St. Thomas, St. John	1999-2000	2	X	X		X
8	Bythell, John; Lundgren, Ian*[15,16,17]	Buck Island National Reef Monument	1989-93,95-97, 99-00, 02-03, 05	13	X		X	
9	Miller, Jeff; Rogers, Caroline*[18,19]	St. John	1989-2002	14	X		X	
a	Nemeth & Nowlis2011[20]	St. Thomas	1997	1	X			
b	Rogers, Caroline[21,22,23,24,25]	St. Croix, St. Thomas, St. John	1978-1981, 1983-1986, 1999-2005	11	X	X	X	
c	Rogers & Zullo 1987[26]	St. John	1984-1985	2	X			
d	Smith, Tyler; Nemeth, Richard*[27,28]	St. Croix, St. Thomas, St. John	2001-2010	10	X	X	X	X
e	Steneck, Bob*[29]	St. Croix	1982, 1988	2	X	X	X	
f	Witman, Jon*[30]	St. John	1985, 1991	2	X			
g	Witman Jon, Edmunds, Peter*[8]	St. John	1989	1	X			
h	Friedlander, Alan; Miller, Jeff; Beets, Jim*[31]	St. John	1989-2011	22				X
k	Bauer 1980[32]	St. Croix, St. Thomas, St. John	1978-1979	2		X		
m	Carpenter 1986[33]	St. Croix	1983	1		X		
n	Carpenter 1981[34]	St. Croix	1979	1		X		
o	Carpenter 1984[35]	St. Croix, St. Thomas	1981-1986	6		X		
p	Carpenter 1988[36]	St. Croix	1983-1984	2		X		
q	Carpenter 1985[37]	St. Croix	1983-1984	2		X		
r	Carpenter 1990[38]	St. Croix	1983-1986	4		X		
s	Hay 1984[39]	St. Croix, St. Thomas	1981-1982	2		X		
t	Hay & Taylor 1985[40]	St. Croix, St. Thomas	1981-1982	2		X		
u	Levitan, Don*/Karlson & Levitan 1990[41]	St. John	1983-1990, 1992, 2009, 2010-2011	12		X		
v	Lessios 1988[42]	St. Croix, St. John	1983-1984	2		X		
w	Levitan 1988[43]	St. John	1983-1987	5		X		
x	Armstrong, Roy[44]	Hind Bank MCD	2003	1	X		X	
y	Ogden 1973[45], 1977[46]	St. Croix	1973, 1974	2			X	
z	Reef Check	St. Croix, St. Thomas, St. Croix	2002-2005	5			X	

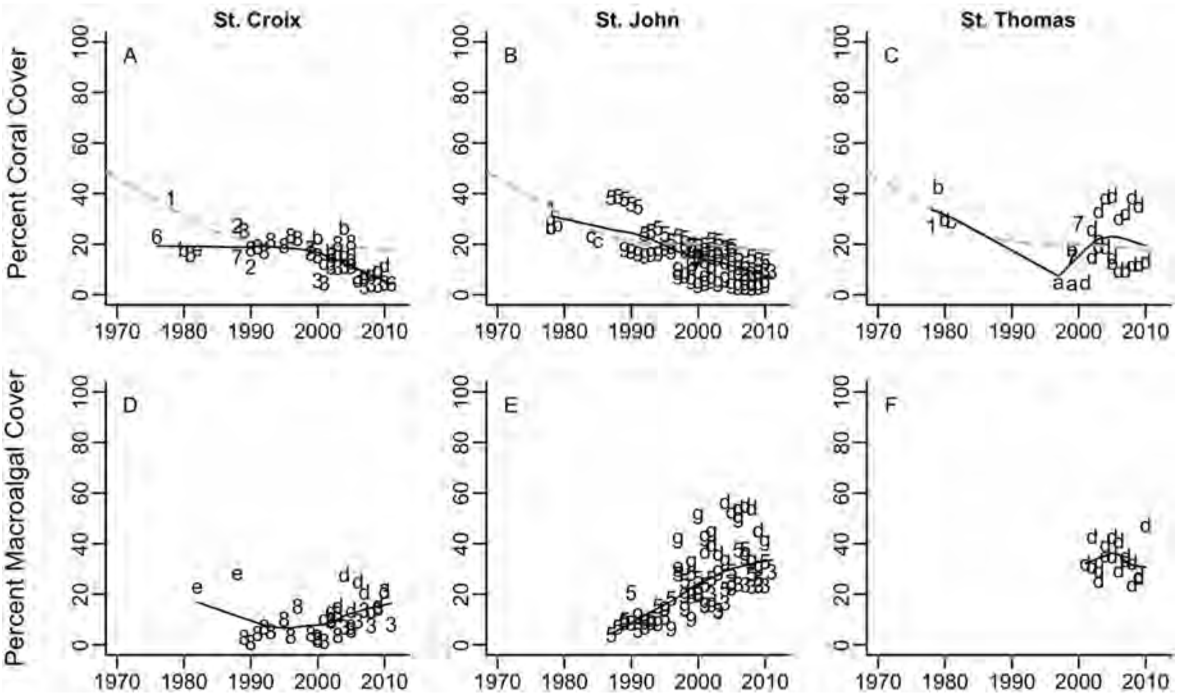

Fig. 33.2 Average percent cover of live corals and macroalgae for 3 islands in USVI: St. Croix (A & D), St. John (B & E) and St. Thomas (C & F). Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 33.1 and Figure 33.1)

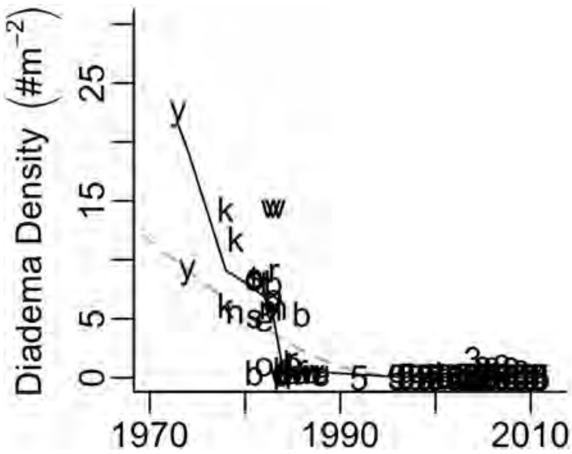

Fig. 33.3 Average density of *Diadema antillarum* for all USVI locations combined. Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 33.1 and Figure 33.1)

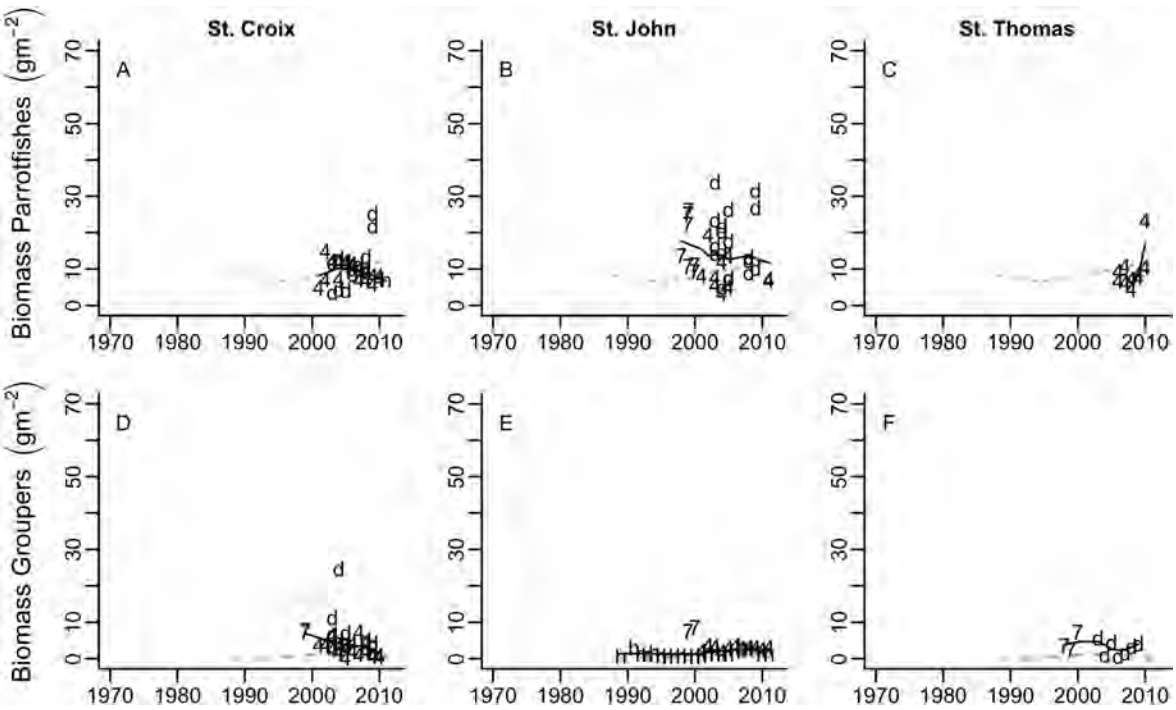

Fig. 33.4 Average biomass of parrotfishes and groupers for 3 islands in USVI: St. Croix (A & D), St. John (B & E) and St. Thomas (C & F). Dotted line represents the average of Caribbean data collected for this report; solid line is drawn through data presented. (Codes same as in Table 33.1 and Figure 33.1)

Timeline

1956:	Virgin Islands National Park established (Land only)
1958-1961:	Early research by Jack Randall and others on fish taxonomy and diet, along with first benthic habitat maps
1961:	Buck Island Reef National Monument established
1962:	Virgin Islands National Park expanded to include marine areas
1966:	Boat use in Virgin Islands NP estimated to be approx. 3 boats daily; 1st research station established, the Virgin Islands Environmental Resource Station (VIERS) in Lameshur Bay, St. John
1969-1970:	Tektite Project
1973:	*Acropora palmata* dies from white band disease off St. Croix (Robinson 1973)
1977:	White band disease identified on St. Croix (Gladfelter1977)
1979:	Hurricanes David caused a significant decrease in coral cover from 65% to 44% in Flat Cay Reef, St. Thomas (Rogers et al. 1983); Hurricane Frederic
1980:	Hurricane Allen
1983:	*Diadema* sea urchin die-off observed throughout Caribbean
1986:	Boat use in Virgin Islands NP estimated to be approx. 80 boats daily
1987:	Mass bleaching event in St. Croix and St. John
1989:	Hurricane Hugo, caused a decrease in coral cover from 20% to 12% in St. John (Rogers et al. 1991; Edmunds & Witman 1991) and also widespread damage in St. Croix; long-term transects established in Great Lameshur, St. John
1990:	Seasonal closure at Red Hind Marine Conservation District, St. Thomas; long-term transects established in Newfound, St. John
1993:	Seasonal closure at suspected mutton snapper spawning site, St. Croix; Nassau grouper protected in US Federal waters
1995:	Hurricane Luis (August) and Hurricane Marilyn (September, Category 2) caused widespread damage but no decreases in coral cover noted in Lameshur Bay or Newfound sites (Rogers & Miller 2006)
1995:	562,000 overnight visitors; 117,1000 cruise ship visits
1996:	Hurricane Bertha, minor damage
1997:	White Plague first observed on St. John
1997-2001:	Decline of *Montastrea annularis* in Tektite, St. John due to diseases (Miller et al. 2003)

1998: Bleaching event and hot water although no coral cover loss at Lameshur or Newfound; Hurricane Georges, minor damage; tourism accounts for 70% GDP

1999: Bleaching event; Hurricane Lenny, minor damage; year-round closure of Red Hind Bank Marine Conservation District, St. Thomas

2001: VIWMA (VI Waste Management Authority) leads sewer system upgrades; expansion of Buck Island Reef National Monument, St. Croix; establishment of Virgin Islands Coral Reef Monument, St. John; fishing pressure continues, overfished conditions described in St. Croix and St. John (Rogers & Beets 2001)

2003: Fee charged for overnight boat moorings in VINP (~4000 boat nights/year); establishment of East End Marine Park on St. Croix

2004: Tropical Storm Jeanne, relatively minor

2005: Most severe bleaching in USVI due to warm in-situ water temperatures with >90% coral bleached but regaining coloration in October, bleaching mortality not extensive in 4 sites in St. John and 1 off Buck Island, *A. palmata* colonies in St. John with more disease mortality than unbleached colonies, St. Croix; *Acropora palmata* and other corals bleached in VINP; seasonal closure of Grammanik Bank spawning aggregation area

2005-2007: Widespread coral disease outbreak including white plague on non-Acroporid reef-building species after bleaching (one of the most significant causes of coral mortality in USVI); *M. annularis* cover decreased by half and average coral cover loss of 60% in St. John and St. Croix (Miller et al. 2009)

2006: Nassau grouper protected in US Virgin Islands Territorial waters

2007-2011: Low disease prevalence at long-term monitoring sites in St. John and St. Croix

2008: Extreme Atlantic swell event in March damaged Elkhorn corals; Hurricane Omar, damage limited to parts of St. Croix; overfishing status continues (Pittman et al. 2008)

2010: Hurricane Earl, relatively minor and patchy impacts; rainfall exceeding 30-year records in St. John; minor territory-wide bleaching event with 62.3% coral cover bleached but no significant decline in cover; Tropical storm Tomas produced much rain, mudslides and runoff on St. Croix and other islands; VINP Fee mooring programs logs >10,000 boat-nights/year

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VENEZUELA

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Geographic Information	
Coastal Length:	6,680 km
Land Area:	911,440 km²
Maritime Area:	472,651 km²
Reef Area:	728 km²
Number of hurricanes in the past 20 years:	2

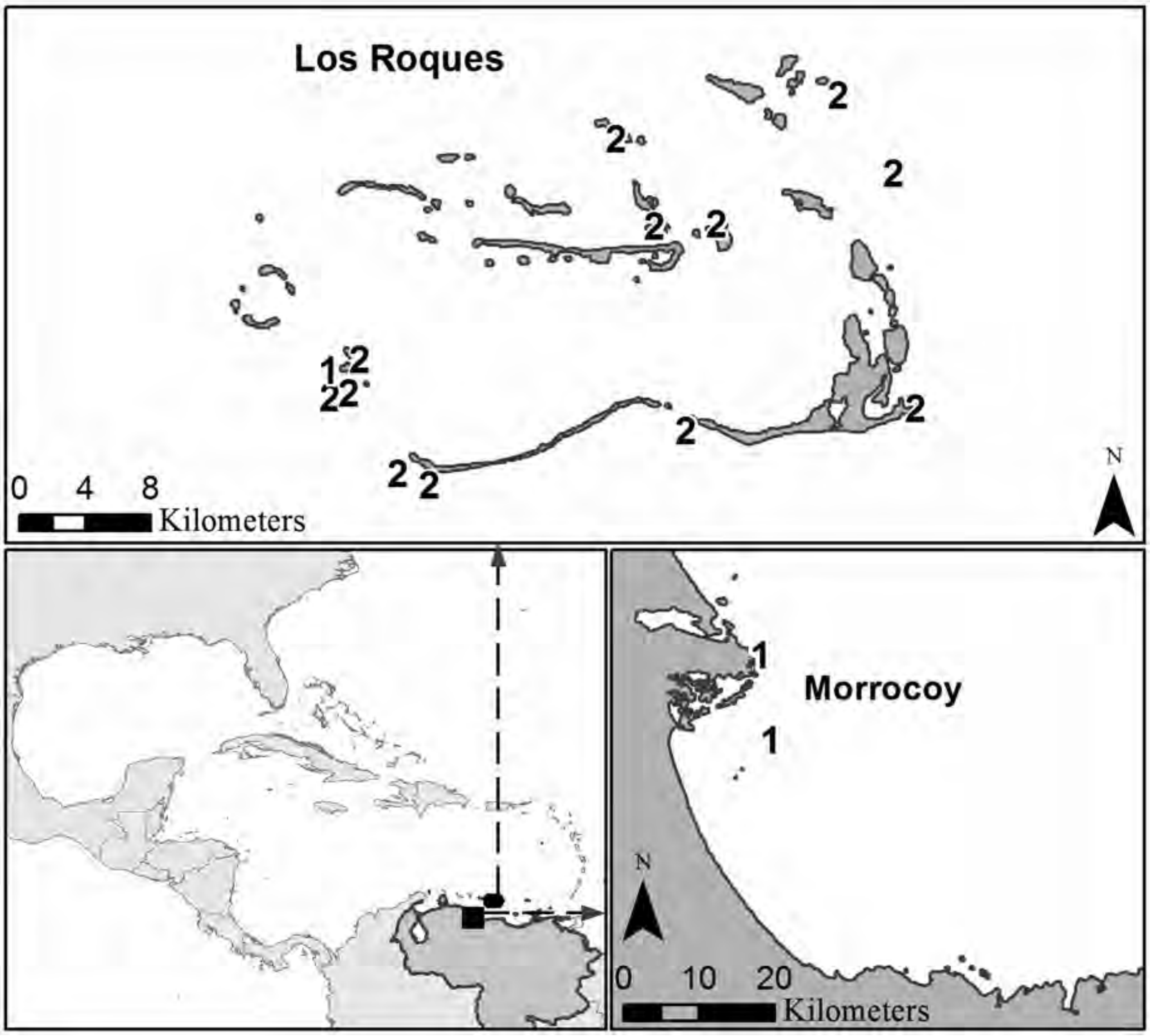

Fig. 34.1 Map of Venezuela, codes represent studies listed in Table 34.1. Missing map code(s) due to unavailable coordinates.

Table 34.1 Data sources from Venezuela. Map codes represent individual studies. For exact location of study, refer to Fig. 34.1; * denotes original data; for full references, refer to published literature sources in the last section.

Map Code	Contributor	Location	Time Period	Year Count	Coral	*Diadema antillarum*	Macroalgae	Fishes
1	Bastidas, Carolina; Cróquer, Aldo*[1,2]	Los Roques	2003-2008	6	X		X	
2	Villamizar, Estrella; Posada, Juan/AGRRA*[3,4]	Los Roques	1999	1	X	X		X
3	CARICOMP*[5,6]	Morrocoy	1996-2011	16	X		X	
4	Reef Check*		2004	1		X		

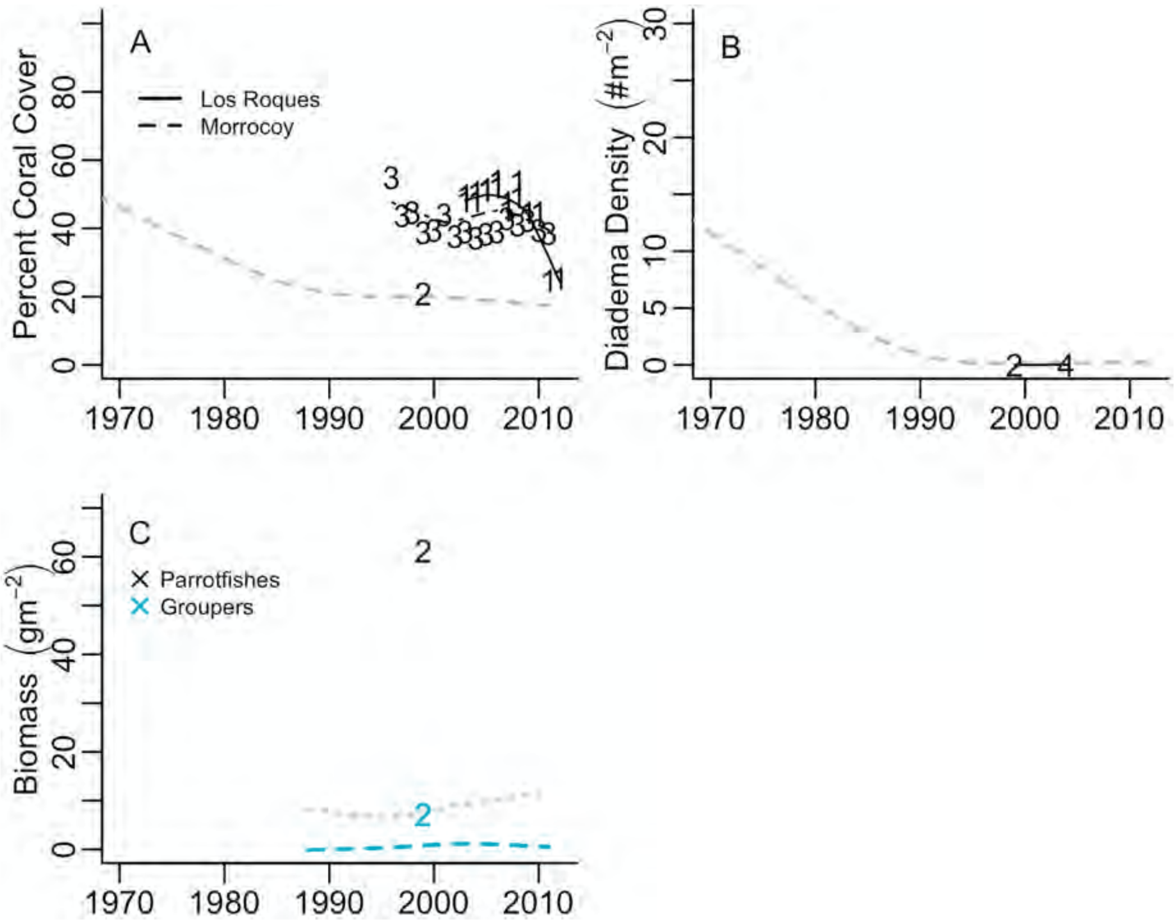

Fig. 34.2 Average percent cover of live corals (A) density of *Diadema antillarum* (B), and biomass of parrotfishes and groupers (C) in Venezuela. Dotted line represents the average of Caribbean data collected for this report; solid lines are drawn through data presented. (Codes same as in Table 34.1 and Figure 34.1)

Timeline

~1200-1950: Pre-Hispanic cultures exploded coral reef resources, both along the Venezuelan coast and the Oceanic islands such as Los Roques (Antczak & Antczak 2005). In Los Roques, piles of queen conch are distributed across the archipelago, with ages older than the first arrival of Columbus (Antczak & Antczak 2006). Most fisheries were artisanal, localized in coastal towns of low population number

1950s-1970s: First descriptions of coral reef communities providing species list, focused on intertidal corals at Las Aves archipelago and the islands of Margarita and Cubagua (Weil 2003). Anecdotal information from fishermen in Los Roques and Morrocoy show that coral reefs were healthy, corals dominated the seafloor and fish were abundant

1970s-1980s: The expansion of the industry of fisheries (three fleets: bottom trawl, tuna and Palangre) started and an exponential growth occurred up to the beginning of the 21st century. During this decade several studies aimed to describe the biology and ecology of coral reefs and their fish communities were done across the Venezuelan shelf and its oceanic islands (Weil 2003). This decade was a “golden age” of studies in coral reef ecology, biology and geology in Venezuela, for over 30 theses for bachelor, masters and PhD degrees were done, mostly in Los Roques. None of these studies reported extensive and/or recent loss of coral cover

1983: Mass mortality of *Diadema antillarum*. Weil 1980 and Weil et al. 1984 studied the population dynamics and the bioerosion rates of *D. antillarum* at Morrocoy National Park; this and other studies had ever reported a massive mortality on *Diadema antillarum*. Currently, this species is common across islands and along the Venezuelan coast where rocky reefs, coralline patches and marginal reef communities cover large and extensive areas as shown in recent surveys; however its abundance has not reached pre-die off values of density (e.g. Noriega et al. 2006). Ramos-Flores (1983) conducted the first study on coral diseases in Venezuela

1985: First study on *Acropora cervicornis* dynamics (Sandia & Medina 1987) at Los Roques National Park, no reports of white plague and major cause of mortality was associated to gastropod and fish predation and fragmentation. Nevertheless, large stands of dead *Acropora palmata* can be seen at Los Roques and other Venezuelan islands (e.g. Zubillaga et al. 2008) and in some coastal zones such as Morrocoy and San Esteban National Park; as well as the central coast of Venezuela (Martínez & Rodríquez-Quintal 2012). Presence of dead skeletons of *Acropora palmata* suggests that populations of this species in Venezuela were also impacted (no record to support the cause of mortality); however, but we cannot determine when that happened

1987-88: First quantitative record of a bleaching event in Venezuela (Lang et al. 1992), impact was only quantified in Morrocoy National Park. The event started by mid November of 1988 affecting major reef builders (e.g. C. *natans*, M. *annularis* and Agaricia spp) but by mid February the monitoring sites were almost recovered with low to no mortality associated

1988: Hurricane Gilbert (Category 3), no impacts on Venezuela

1991: Ban on queen conch exploitation (e.g. Schweizer & Posada 2006)

1993: CARICOMP starts at Morrocoy National Park. Hurricane Bret, although it was a hurricane of low intensity impacted the southern Caribbean and the coast of Venezuela. No quantification on the effects of this hurricane on Venezuelan reefs but had significant impacts on shallow-water corals in Curacao, especially on *Acropora palmata* populations (Van veghel & Houtches 1995)

1995: Mild bleaching event in Morrocoy (CARICOMP) with no significant loss of coral cover

1996: Massive mortality event wiped-out the majority of reefs at Morrocoy National Park (Laboy-Nieves et al. 2001), attributed to an abnormal upwelling event combined to lack of winds capable of mixing the water column which produced a plankton bloom which deposited into the seafloor rendering anoxic conditions that killed large number of taxa, including corals, octocorals and other sessile and non-sessile organisms. This is one of the greatest massive die-offs reported in Venezuela

1998-99: Quantitative surveys of bleaching were conducted in Morrocoy, particularly at Sombrero Key (CARICOMP Monitoring site since 1996)

1999: Massive and prolonged rainfalls, increasing terrestrial runoff, sedimentation which reduced visibility and salinity in Morrocoy (Chollett & Bone 2007, Chollett et al., 2007); first observation of coral diseases in Sombrero Key (Cróquer and Bone 2003), with yellow band and white plague being the most prevalent

2000. First coral disease epizootic event reported at Madrizqui Key at Los Roques National Park (Cróquer et al. 2003), affecting over 20 species of corals and producing significant lost of coral cover in a 2-year period (Cróquer et al. 2005)

2004: Hurricane Ivan (Category 3); octocorals and corals were transported by strong waves along the exposed keys forming large terrace deposits in Los Roques (Cróquer per. Observ.). Shallow reef sites (e.g. “La piscina of Franciski”) in Los Roques were severely affected although no formal reports that quantified the impact of this hurricane

2005: Bleaching affected oceanic and coastal reefs along the Venezuelan coast, but no significant loss of coral cover was observed (Rodríguez et al. 2010). This bleaching event has minor consequences in Venezuela, compared with other Caribbean localities (Eakin et al. 2010)

2007: Hurricane Dean and Felix (Category 5); impacted the coast of Venezuela and their oceanic islands. Effects were similar to Ivan (Cróquer, personal observation)

2008: Lionfish *Pterois volitans* first documented

2009: Earthquake (6.2 magnitude) affected the western coast of Venezuela, with the epicenter in the adjacencies of Morrocoy National Park (FUNVISIS 2009); fractures in the reef framework and minor mortality due to coral break and overturned of colonies; trawling banned in entire country; new laws that prohibited trawl fisheries were created and implemented in 2009, today industrial fisheries such as tuna and palambre can only operate offshore in Venezuela

2010: Bleaching event reducing over 20% of the live coral cover in Los Roques from November 2010 to February 2011 (Bastidas et al. 2012). The cover continued to drop up to June 2011. The bleaching was massive in most of the archipelago, being observed across the park and in shallow (1m) and deep environments (up to 35 m, Cróquer Pers. observation). Major reef builders such as Montastraea spp, Colpophyllia spp, Diploria spp and Agaricia spp were severely affected. Bleached colonies became infected first by black band disease, then by white band disease and lately by yellow band disease (Cróquer, per. Observation). Surprisingly, *Acropora palmata* was bleached but seldom died compared to Montastraea spp and Colpophyllia spp. Reports of bleaching from diving operators across the country were common. The effect of this bleaching event had no precedent in the recent history of coral reef catastrophes in Venezuela

2011: Inshore waters heavily overfished, fishermen forced to fish in open waters

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