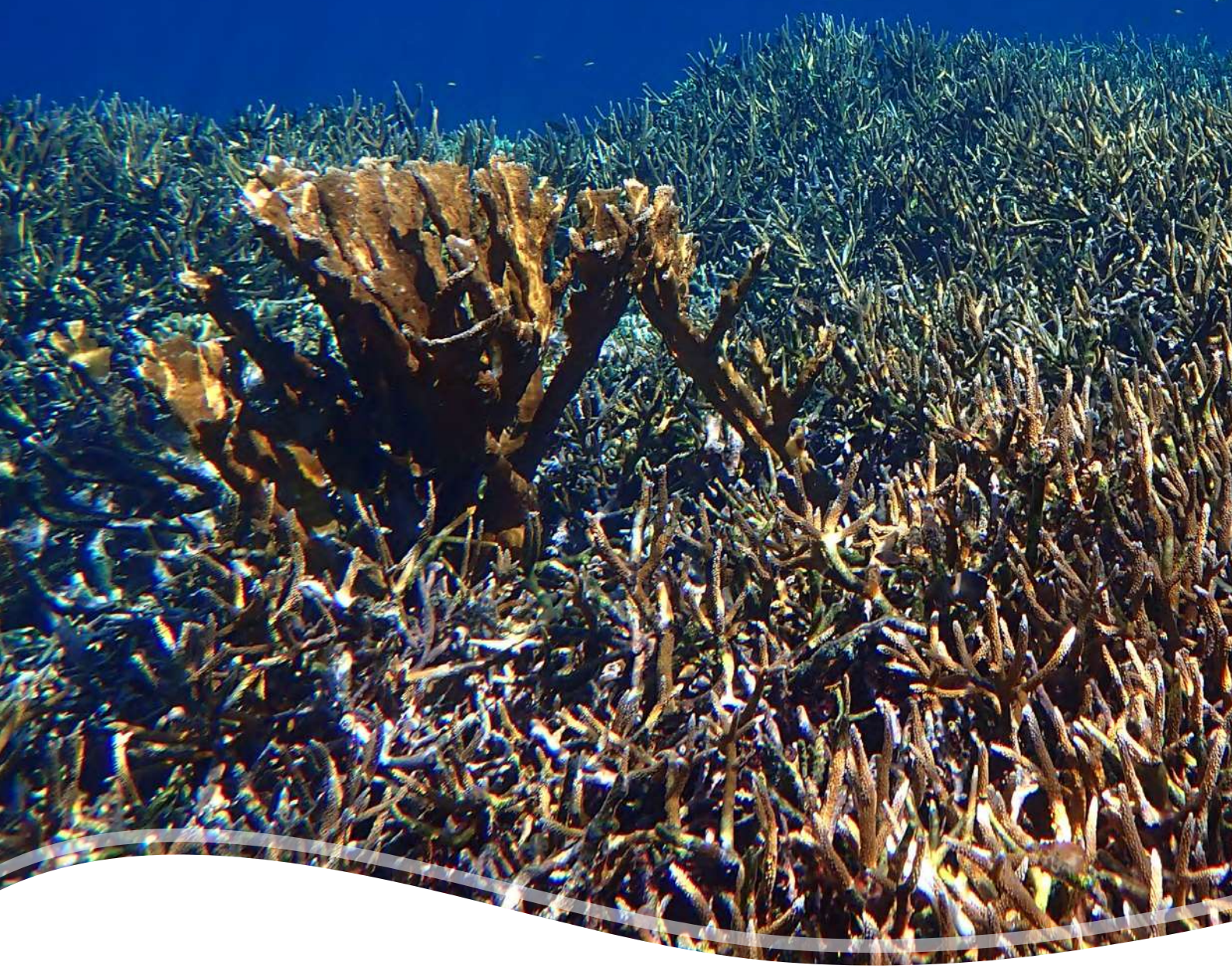
GCRMN
Global Coral Reef Monitoring Network

ICRI
International Coral Reef Initiative

STATUS & TRENDS OF **CARIBBEAN** CORAL REEFS:

1970 - 2024

Edited by Jérémy Wicquart,
Laëtitia Mathon, Auriane Petit,
Andrea Rivera-Sosa & Melanie McField



UN environment programme

Caribbean Environment Programme

SPAW • RAC

Healthy Reefs for healthy people

Global Fund for Coral Reefs

DEDICATION

This report is dedicated to the numerous individuals who have worked to study, conserve and protect our coral reefs. We also recognize the International Coral Reef Initiative and partners, and particularly the people of all territories and nations throughout the Wider Caribbean region who continue to strive for the existence of healthy coral reefs for future generations.

CITATION

Wicquart, J., Mathon, L., Petit, A., Rivera-Sosa, A., and McField, M. (eds.), 2025. Status and Trends of Caribbean Coral Reefs: 1970 – 2024. Global Coral Reef Monitoring Network (GCRMN) and International Coral Reef Initiative (ICRI). DOI: https://doi.org/10.59387/BDHF9180

CITATION FOR INDIVIDUAL CHAPTERS

Authors. Chapter title. In Wicquart, J., Mathon, L., Petit, A., Rivera-Sosa, A., & McField, M. (eds.), 2025. Status and Trends of Caribbean Coral Reefs: 1970 – 2024. Global Coral Reef Monitoring Network (GCRMN) and International Coral Reef Initiative (ICRI). DOI: https://doi.org/10.59387/BDHF9180

DOI

https://doi.org/10.59387/BDHF9180

NOTE

This report has been edited and organized by the Global Coral Reef Monitoring Network (GCRMN) Caribbean Regional Coordinators in collaboration with the GCRMN National Coordinators of countries in the Caribbean Region and does not necessarily reflect the policies or positions of contributing governments. This publication may be reproduced in whole or in part and in any form for educational or non-profit services, provided acknowledgement of the source is made. No use of this publication may be made for resale or any other commercial purpose without prior permission in writing.

DESIGNATIONS

The designations employed and the presentation of material, including the maps, nomenclature and associated data, do not imply the expression of any opinion whatsoever on the part of the Global Coral Reef Monitoring Network (GCRMN), the International Coral Reef Initiative (ICRI), or participating organisations concerning the legal status of any country, territory, city or area, or of its authorities, or concerning the delimitation of its frontiers or boundaries.

FRONT COVER

Healthy reef community of *Acropora cervicornis*, Roatan, Honduras, 2019.
© Melina Soto / Healthy Reefs for Healthy People

GRAPHIC DESIGN Julia Bos - Ecodyssey

GCRMN
Global Coral Reef Monitoring Network

ICRI
International Coral Reef Initiative

GCRMN Global Coral Reef Monitoring Network
www.gcrmn.net

ICRI International Coral Reef Initiative
www.icriforum.org

ACKNOWLEDGMENTS

FUNDING

The publication of this report would not have been possible without the generous financial support of the United Nations through three different projects and initiatives supporting coral reef conservation, marine protected areas, and Caribbean regional cooperation.

First, we would like to express our sincere gratitude to the **Global Fund for Coral Reefs (GFCR)** for their invaluable contribution. Co-led by the United Nations Capital Development Fund (UNCDF), the United Nations Development Programme (UNDP) and the United Nations Environment Programme (UNEP),, the GFCR is a public-private coalition deploying a range of financial solutions and support for sustainable businesses to improve the resilience of coastal ecosystems, economies, and communities.

Secondly, we would like to express our profound gratitude to the **ACP MEAs programme.** The ACP MEAs programme is a joint partnership between the European Union, the Organization of African, Caribbean and Pacific States, UN Environment Programme and the Food and Agriculture Organization of the United Nations which aims at building capacity in 79 countries in Africa, Caribbean, and Pacific (ACP) to support them fulfill their obligations as parties to Multilateral Environmental Agreements (MEAs) to tackle the environmental issues they face.

Thirdly, we would like to express our warmest thanks to the **Global Environment Facility (GEF) project "Implementation of the Strategic Action Program for the Gulf of Mexico Large Marine Ecosystem"**. This GEF project is supported by the United Nations Environment Programme (UNEP) as the executing agency and is implemented by the United Nations Office for Project Services (UNOPS). Through component 3 of the project, aimed at conserving and restoring the quality of coastal and marine ecosystems and greater bilateral cooperation, the UNEP Cartagena Convention Secretariat is taking action in this regard by strengthening Marine Protected Areas (MPA) networks in the Wider Caribbean, including the Gulf of Mexico, and contributing to initiatives such as the preparation of this report in collaboration with the SPAW-RAC and the GCRMN Caribbean.

We would like to express our sincere gratitude to the Government of France and the Government of Monaco for their support for the report elaboration and dissemination.

COORDINATION

We would like to give our acknowledgements to the Specially Protected Areas and Wildlife - Regional Activity Center (SPAW-RAC) for coordinating the production of this report. Their efforts included, among others, organizing numerous videoconference meetings and the in-person GCRMN Caribbean Data Workshop, along with Healthy Reefs for Healthy People, serving as the co-chair of GCRMN Caribbean for the past three years. We warmly thank the Universidad Nacional Autónoma de México (UNAM) and the Instituto de Ciencias del Mar y de Limnología for their generous hospitality in hosting this workshop, held in Puerto Morelos from 1 to 4 October 2024. We are also grateful to the university and Healthy Reefs for Healthy People staff for their assistance in welcoming and supporting participants, and particularly Melina Soto. Special thanks to the Coral Reef Alliance and Dr. Miguel Ángel Zarate for hosting Dr. Andrea Rivera-Sosa at the Laboratorio de Corales Pétreos y Arrecifes Coralinos from the Colegio de la Frontera Sur (ECOSUR) Unidad Chetumal, and the Secretaría de Ciencia, Humanidades, Tecnología e Innovación (SECIHTI) for their support. Finally, we wish to acknowledge the International Coral Reef Initiative (ICRI) Secretariat for their continuous guidance and support throughout the preparation of this report.

DATA CONTRIBUTION

This report owes much to the dedicated efforts of all those who have collected data in Caribbean coral reefs over the past decades. We extend our deepest appreciation to these contributors for generously sharing their data, which were instrumental in the development of this report. Their individual contributions are recognized in the *Authors' Contributions* section. We are particularly grateful to the many data contributors to the Atlantic and Gulf Rapid Reef Assessment (AGRRA) Database, and to Patricia Kramer and Lynnette Roth for their kind guidance in the use of this database. We also thank the Reef Check Foundation, and its volunteers who have spent countless hours collecting these data.

DATA ANALYSIS

We wish to thank all participants of the GCRMN Caribbean Data Workshop, both those who attended in person and those involved in the Data Task Force. We are especially grateful to Lorenzo Alvarez-Filip, Erica Towle, Aaron Israel Muñiz-Castillo, Mark Vermeij, Robert Steneck, Aldo Croquer, Patricia Kramer, Jean-Philippe Maréchal, Tyler Smith, and Fabio Quesada-Perez for their valuable advice and insights throughout the preparation of this report.

FOREWORD

Christopher Corbin,

Coordinator UNEP Cartagena Convention Secretariat

Caribbean coral reefs are more than just underwater marvels; they are the bedrock of biodiversity, coastal protection, and economic vitality for our region. It is therefore my great pleasure to present the report Status and Trends of Caribbean Coral Reefs: 1970–2024.

This publication is a landmark effort, bringing together over five decades of data and knowledge on one of our most vital ecosystems.

Its production was coordinated by the GCRMN-Caribbean handled by the Specially Protected Areas and Wildlife Regional Activity Centre (SPAW-RAC) and Healthy Reefs for Healthy People, with funding support from the UNEP Cartagena Convention Secretariat through the GEF project "Implementation of the Strategic Action Program of the Gulf of Mexico Large Marine Ecosystem", UNEP, and the Global Fund for Coral Reefs. Their leadership and collaboration have been essential in ensuring that the information presented here is both comprehensive and accessible.

The Cartagena Convention and its Protocols remind us that partnership is the foundation of progress. The SPAW Protocol, in particular, provides the legal framework that underpins regional cooperation for the conservation of coral reefs and associated biodiversity.

This report exemplifies that spirit of collaboration, uniting governments, scientists, civil society, and regional organizations to monitor, understand, and safeguard our reefs. By harmonizing monitoring approaches and building strong networks, we have produced not just a report, but a shared regional picture of reef health and resilience.

The findings are sobering: Caribbean reefs have lost more than half of their live coral cover since 1970, with mounting pressures from climate change, pollution, and overuse. In this context, reliable data is not optional — it is essential. Sound science must guide the policies and investments needed for recovery. These findings will help inform strategies that protect biodiversity while also securing food, livelihoods, and cultural identity for the millions who depend on reefs.

On behalf of the Cartagena Convention Secretariat, I extend sincere thanks to SPAW-RAC, the Healthy Reefs for Healthy People, the authors and all GCRMN-Caribbean contributors. This report is both a call to action and a reminder that with commitment, collaboration, and science on our side, we can still secure a future where Caribbean reefs thrive and continue to inspire generations to come.

Best,

Christopher Corbin

TABLE OF **CONTENTS**

ACKNOWLEDGMENTS **3**

FOREWORD **6**

EXECUTIVE SUMMARY **10**
RÉSUMÉ EXÉCUTIF **18**
RESUMEN EJECUTIVO **26**

LIST OF ACRONYMS **34**

INTRODUCTION **35**

PART I. SYNTHESIS FOR THE WIDER CARIBBEAN REGION **39**

1. Coral reefs in the Caribbean 40
2. Threats to coral reefs of the Caribbean 46
3. Temporal trends in benthic cover 67
4. Recommendations 80

PART II. SYNTHESES FOR COUNTRIES AND TERRITORIES **91**

1. Anguilla 92
2. Antigua and Barbuda 96
3. Aruba 100
4. The Bahamas 103
5. Barbados 106
6. Belize 110
7. Bermuda 115
8. Bonaire 118
9. British Virgin Islands 125
10. Cayman Islands 129
11. Colombia 133
12. Costa Rica 139
13. Cuba 143
14. Curaçao 147
15. Dominica 151
16. Dominican Republic 154
17. Florida 162
18. East and West Flower Garden Banks 166
19. Grenada 170
20. Guadeloupe 174
21. Guatemala 178
22. Haiti 183
23. Honduras 186
24. Jamaica 190
25. Martinique 193
26. Mexico (Caribbean Sea) 198
27. Mexico (Gulf Of Mexico) 206

28. Montserrat 212
29. Nicaragua 215
30. Caribbean Panama 219
31. Puerto Rico 224
32. Saba 230
33. Saint Kitts and Nevis 234
34. Saint Lucia 237
35. Saint Vincent and The Grenadines 240
36. Saint-Barthélemy 244
37. Saint-Martin 248
38. Seaflower Biosphere Reserve 252
39. Sint-Eustatius 257
40. Sint Maarten 261
41. Trinidad and Tobago 265
42. Turks and Caicos Islands 269
43. United States Virgin Islands 273
44. Venezuela 282

CASE STUDIES

1. A tale of two islands: Lessons from monitoring at fixed sites with the same researchers 122
2. Colombia's Paradoxical Varadero Reef 137
3. The value of a joint coral-monitoring program to improve the outcome of coral restoration programs at the Southeast Reefs Marine Sanctuary of the Dominican Republic 159
4. Protecting the Mesoamerican Reef through Blended Finance: The MAR+Invest Case Study 203
5. Safeguarding Unique Coral Reef Ecosystems in the Gulf of Mexico 210
6. Coral reef fish abundance and reef rugosity in Bocas del Toro, Panama 223
7. Socioeconomic Trends in the U.S. Caribbean 228
8. Forty years of research and monitoring documents the success of a large marine protected area in rebuilding a red hind spawning aggregation in the U.S. Virgin Islands 278
9. Road to recovery for the endangered Nassau grouper in the US Virgin Islands 280

MATERIALS & METHODS 287

AUTHORS' CONTRIBUTIONS 308

BIBLIOGRAPHY 316

SUPPLEMENTARY MATERIALS 341

EXECUTIVE **SUMMARY**

RÉSUMÉ EXÉCUTIF

RESUMEN EJECUTIVO

Brain coral © Carlos Diaz / Ocean Image Bank

EXECUTIVE SUMMARY

KEY POINTS

- Coral reefs of the GCRMN Caribbean region cover **24,230 km²,** which represent **9.7%** of the world's coral reef extent.
- Hard coral cover **declined by 48%** from 1980 to 2024, meaning that hard corals now occupy half of the benthic cover they occupied in 1980.
- Hard coral cover **declined** sharply in **1998** (-9.0%), **2005** (-17.1%), and **2023** (-16.9%) due to bleaching events induced by thermal stress, and coral disease.
- *Acropora* cover remained low between 1980 and 2024, averaging around **1.8%**, far below the values of the early 1970s, which were closer to **16%**. In contrast to these branching species, the benthic cover of *Orbicella* and *Porites*, which encompass more massive hard corals, has remained stable or even increased over the same period. This shift in hard coral species dominance has likely reduced the **structural complexity** of Caribbean coral reefs.
- Macroalgae cover **increased by 84.8%** from 1980 to 2024 likely due to the decline in herbivorous species (*e.g.* parrotfish, sea urchins) and the increase in nutrients.
- Mean sea surface temperature over coral reef areas across the Caribbean **increased by +1.07°C** between 1985 and 2024, driven by climate change, representing a warming rate of **0.27°C** per decade.
- The number of people living within 20 km from coral reefs increased by **27.6%** from 2000 to 2020 at the regional scale, representing an increase of **13 million people**.

Introduction

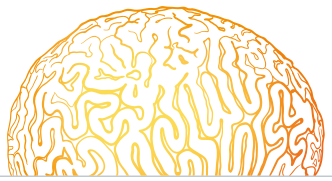
Coral reefs are among the most diverse ecosystems on Earth, supporting around one third of all named marine species[1]. Beyond their ecological importance, they provide essential ecosystem services to human societies, protecting coastlines from storms, supporting food security through fisheries, and generating income from tourism. However, coral reefs are increasingly threatened by a range of direct and indirect human pressures, including climate change, pollution, and unsustainable fishing practices.

Recognizing the urgency of these threats, global, regional, and national frameworks, such as the Kunming–Montreal Global Biodiversity Framework, must be implemented to ensure the effective conservation and sustainable management of coral reef ecosystems. Measuring progress toward these goals, and adjusting strategies where necessary, requires robust quantitative syntheses conducted at large spatial scales.

Established in 1995 as an operational network of the International Coral Reef Initiative (ICRI), the Global Coral Reef Monitoring Network (GCRMN) plays a central role in this effort. By providing the best available scientific data, the GCRMN informs policy, strengthens management, and strengthens capacity for coral reef conservation worldwide. Organized through ten regional nodes, including the Caribbean, the network regularly publishes global and regional assessments on the status and trends of coral reefs.

The previous regional assessment for the Caribbean, "*Status and Trends of Caribbean Coral Reefs: 1970–2012*", was published in 2014 and incorporated data up to 2012. Since then, numerous local studies have documented ongoing coral reef degradation across the region. More recently, the *Status of Coral Reefs of the World – 2020* report confirmed the continued decline of Caribbean reefs. In response to these alarming observations, the Specially Protected Areas and Wildlife - Regional Activity Center (SPAW-RAC) and the United Nations Environment Programme (UNEP) initiated the development of a new regional assessment, supported by the Global Fund for Coral Reefs (GFCR).

The present report, "*Status and Trends of Caribbean Coral Reefs: 1970–2024*", is the culmination of nearly two years of collaborative work involving nearly 300 contributors across the region. It provides an updated, comprehensive overview of the evolution of key threats affecting Caribbean coral reefs and examines the major changes observed in benthic community composition over the past four decades. This report also highlights urgent priorities for conservation and management across the Caribbean.



[1] Fisher, R., O'Leary, R. A., Low-Choy, S., Mengersen, K., Knowlton, N., Brainard, R. E., & Caley, M. J. (2015). Species richness on coral reefs and the pursuit of convergent global estimates. Current Biology, 25(4), 500-505.

Regional context

The GCRMN Caribbean region hosts 24,230 km² of coral reefs, which represent 9.7% of the world's coral reef extent. The coral reefs in this region fall under the jurisdiction of 44 different countries and territories, with coral reef extent varying significantly from 4 km² in Sint Maarten to 4,931 km² in Cuba.

We estimated that 47 million inhabitants lived within 20 km from a coral reef in the Caribbean in 2020. Many eastern Caribbean island nations, like Barbados, Antigua and Barbuda, or Saint Lucia, have nearly their entire populations living within 20 km from a coral reef. On the contrary, in continental countries like Mexico, Honduras, and Venezuela, only a small percentage of the total population lives close to the sea, and therefore to the reefs. From 2000 to 2020, the number of people living within 20 km of a coral reef of the Caribbean grew by 27.6%, representing 13 million additional people.

Coral reefs in the Caribbean provide multiple ecosystem services, and are valued at $6.2 billion annually in fisheries and tourism benefits. Along with adjacent coastal ecosystems (mangroves, seagrass), their value can increase to $15 billion annually. These reefs support diverse fisheries, including species like snappers, groupers, and spiny lobsters, that are essential sources of protein and micronutrients, particularly in small island developing states. Coral reefs also act as natural barriers that mitigate the impacts of wave energy and storm surges, thereby protecting coastal communities and ecosystems. Finally, by stimulating tourism and sometimes even being the driving force behind it, coral reefs provide significant economic benefits to coastal populations. Coral reefs are also important from a social and cultural perspective, influencing how people throughout the region rely on and connect with the marine environment.

Methods used

We estimated the temporal trends in five major benthic categories (hard coral, macroalgae, turf algae, coralline algae, and other fauna), and three hard coral genera (*Acropora*, *Orbicella*, *Porites*), through the integration of 72 datasets. Together, these datasets represent a total of 13,864 monitoring sites, on which 23,742 surveys were conducted between 1973 and 2024. The temporal trends were estimated using machine learning models to take the bias of the non-homogeneous distribution of monitoring data over space and time into account.

Results

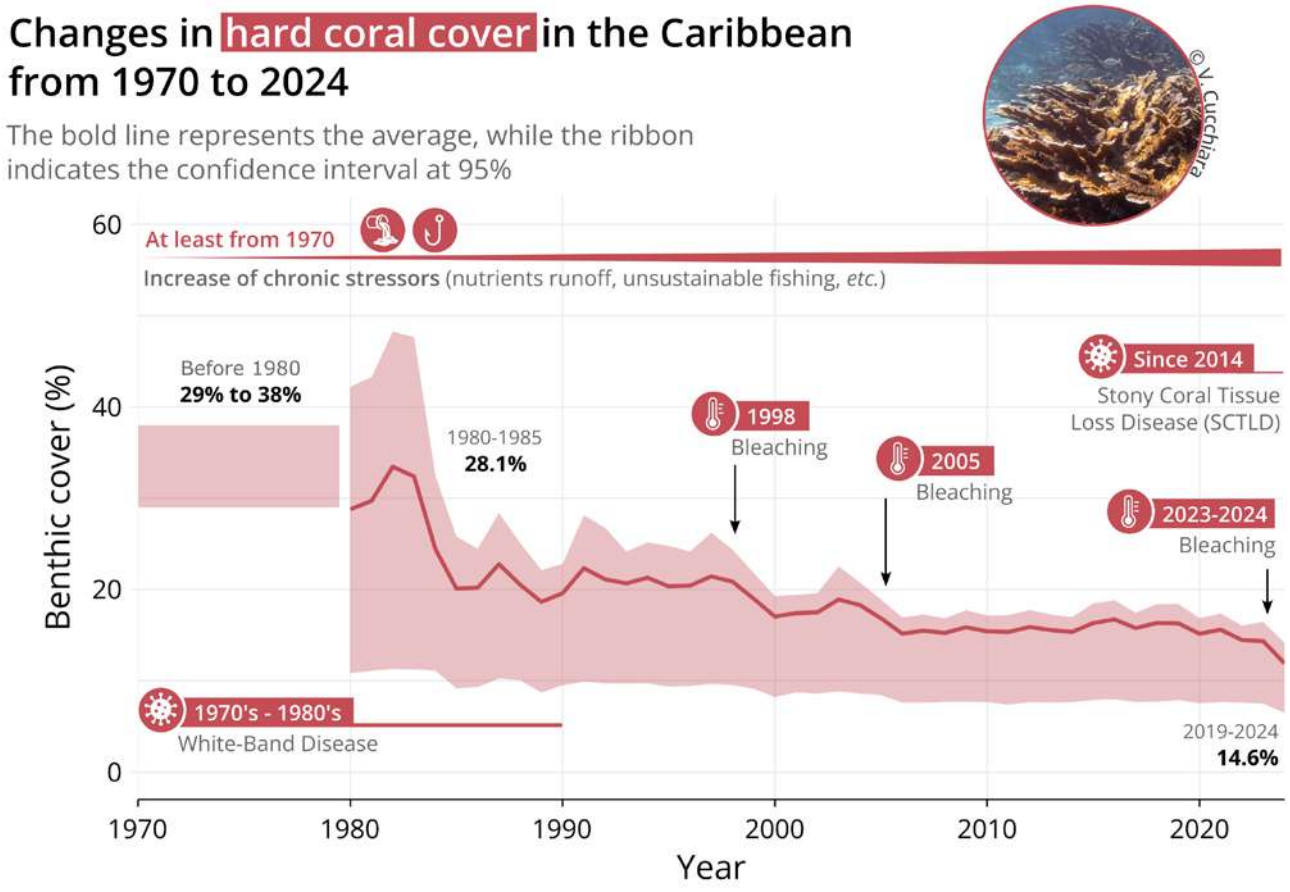
Decrease in hard coral cover

Early monitoring data indicate that hard coral cover across the Caribbean ranged between 29% and 38% before 1983[2]. While higher than present hard coral cover, these values do not represent pristine coral reef conditions in the region, as human stressors were already affecting these ecosystems at that time. Between 1983 and 2024, our analyses reveal that hard coral cover has declined by half, from 28.1% in 1980–1985 to 14.6% in 2019–2024. Major declines coincided with region-wide mass bleaching events driven by thermal stress, notably in 1998 (-9%), 2005 (-17.1%), and 2023–2024 (-16.9%)[3]. Given the limited data available for 2024, the impact of the most recent, and continuing, fourth global bleaching event[4], is likely underestimated in this assessment. In addition, hard coral diseases such as white band disease and stony coral tissue loss disease have contributed to the long-term decline in hard coral cover, with more chronic and spatially variable impacts than bleaching events.

[2] Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (Eds.) (2014). Status and Trends of Caribbean Coral Reefs: 1970–2012. Global Coral Reef Monitoring Network.

[3] The percentages indicate the relative decline, not the absolute decline, in hard coral cover.

[4] Reimer, J. D., Peixoto, R. S., Davies, S. W., Traylor-Knowles, N., Short, M. L., Cabral-Tena, R. A., ... & Voolstra, C. R. (2024). The fourth global coral bleaching event: where do we go from here?. *Coral Reefs, 43*(4), 1121-1125.



Change in hard coral assemblage

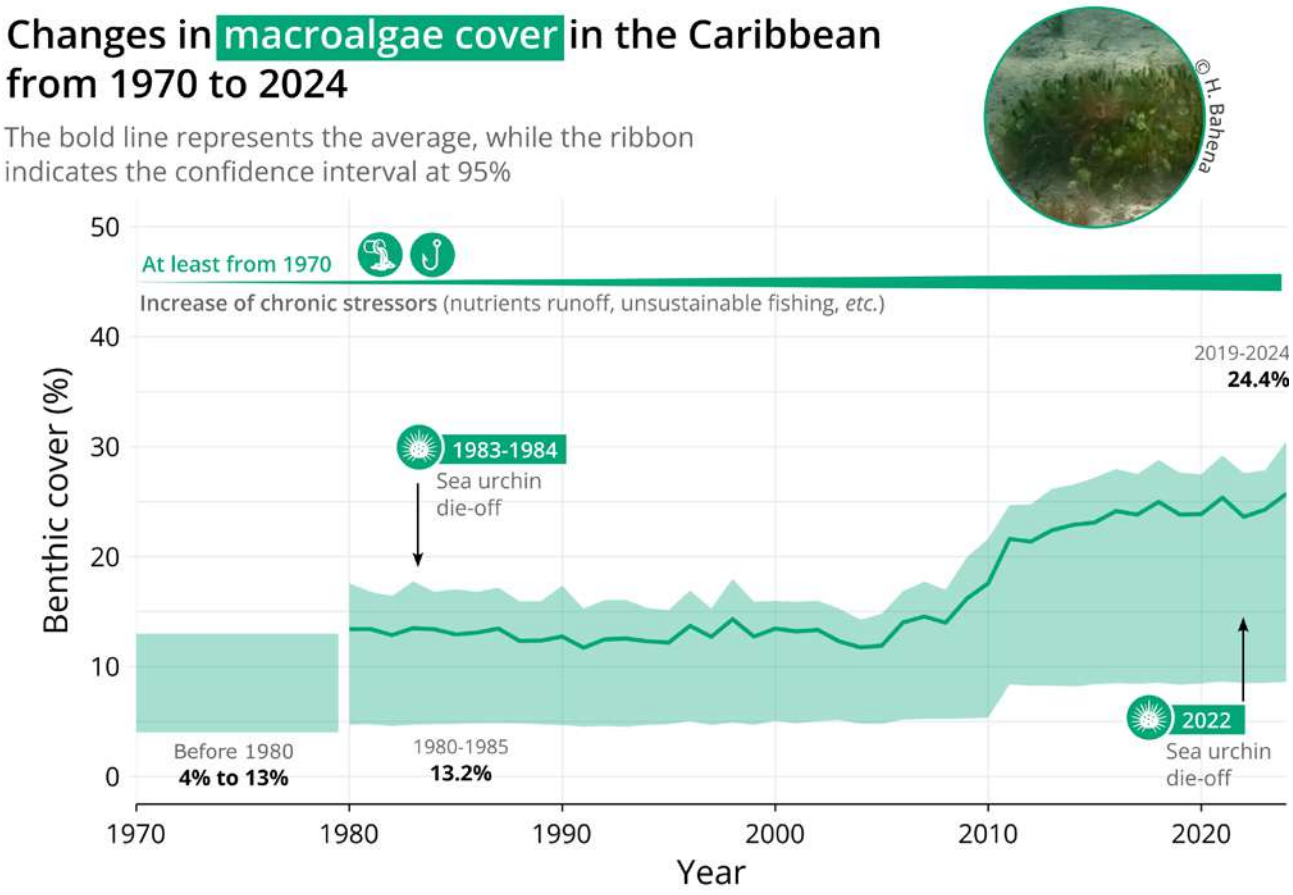
The overall decline in hard coral cover observed over the study period conceals contrasting trajectories among the major hard coral genera in the region. Previous studies have shown that *Acropora* spp., which represented approximately 16% of the benthic cover in the 1970s, declined sharply before 1980 due to anthropogenic stressors and widespread outbreaks of white band disease[5]. Since then, our results indicate that the benthic cover of these branching hard coral species has remained low since 1980, averaging around 1.8%. *Orbicella* spp. cover, which includes massive hard coral species, declined following the 1998 bleaching event and stabilized from 2009 onward at approximately 5%. In contrast, *Porites* spp. cover increased by 164% between 1980 and 2024, reaching an average percentage cover of 3.7% in 2019-2024. Together, these trends suggest a profound reorganization of hard coral assemblages over the past decades, marked by the loss of branching species and the relative rise of more stress-tolerant, massive forms. This compositional shift toward stress-tolerant massive hard corals likely reduced habitat complexity and biodiversity, thus affecting ecosystem services.

Increase in macroalgae cover

Early monitoring data indicate that macroalgae covered between 4% and 13% of Caribbean reefs before 1983[6]. In contrast to hard corals, macroalgal cover has increased by 84.8% between 1980 and 2024. After a period of high variability in the first two decades of monitoring, turf algae also expanded, rising by 64.7% since 2001. Two main factors have been suggested to explain these trends. First, the loss of herbivory following the decline of parrotfish and surgeonfish populations, coupled with mass die-offs of the long-spined sea urchin (*Diadema antillarum*) in 1983-1984 and again in 2022. Second, declining water quality, reflected in increased nutrient loads, sedimentation, and turbidity, has further favored macroalgae and turf algae proliferation. Together, these processes have reduced grazing pressure while stimulating algal growth, leading to sustained increases in macroalgae and turf algae cover. Because these algae compete directly with hard corals for space, this expansion further limits coral recruitment and growth, hindering hard coral cover recovery across the Caribbean.

[5] Cramer, K. L., Jackson, J. B., Donovan, M. K., Greenstein, B. J., Korpanty, C. A., Cook, G. M., & Pandolfi, J. M. (2020). Widespread loss of Caribbean acroporid corals was underway before coral bleaching and disease outbreaks. *Science Advances, 6*(17), eaax9395.

[6] Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (Eds.) (2014). Status and Trends of Caribbean Coral Reefs: 1970–2012. Global Coral Reef Monitoring Network.



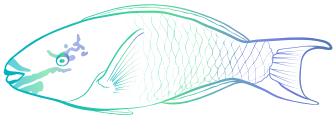
Increase of threats

Climate change

Climate change is an escalating threat to Caribbean coral reefs, driven by rising atmospheric greenhouse gas concentrations over the past century. Between 1985 and 2024, sea surface temperatures across Caribbean coral reefs increased by an average of 1.07°C, representing a warming rate of 0.27°C per decade. Coral reefs from all countries and territories in the region experienced this warming, with the largest long-term increases recorded in Mexico (Gulf of Mexico), Guatemala, and the Flower Garden Banks National Marine Sanctuary, where sea surface temperatures rose by about 1.4°C.

In parallel, marine heatwaves—periods of unusually high and prolonged sea surface temperatures—have become more frequent, intense, and longer-lasting under climate change. These events, especially those in 1987, 1998, 2005, 2010, 2015–2017, 2020, and 2023–2024, have led to bleaching events.

Caribbean coral reefs have also been exposed to more intense hurricanes as a consequence of climate change. From 1980 to 2024, we estimated that 171 hurricanes passed within 100 km of coral reef systems, with notable peaks in 1995 (11 hurricanes), 2005 (15), and 2017 (10). While the impacts of these storms vary by intensity and location, they often result in severe structural damage to coral reefs. Climate projections indicate that while the overall frequency of hurricanes may remain stable toward the end of the century, their intensity is expected to increase, potentially leading to increased damage of coral reefs.



Water quality

Beyond the global impacts of climate change, Caribbean coral reefs have been increasingly threatened by local and direct human activities. The significant increase in human population throughout the region in the last decades has led to greater pressure on coastal ecosystems. Nutrient and sediment inputs from agriculture and coastal development have increased substantially, contributing to a decline in water quality and a widespread rise in macroalgae cover. Chemical and plastic pollution have increased significantly, with microplastic densities in oceans doubling each decade since 1970. This pollution poses a serious threat to coral reefs by exacerbating bleaching and diseases. In parallel, the expansion of maritime traffic, including cargo and cruise shipping, has increased physical damage while contributing to the degradation of water quality.

Diseases

Caribbean coral reefs have faced increasing frequency and severity of disease outbreaks in recent decades, driving widespread mortality among hard corals and reef organisms. Linked to rising sea temperatures, pollution, and other human impacts, these diseases are accelerating reef degradation and biodiversity loss. During the 1990s, several diseases emerged in the Caribbean: the yellow-band disease led to massive loss of *Orbicella* spp., the white-band disease affected acroporids massively, while the white plague led to the loss of multiple species. Stony Coral Tissue Loss Disease (SCTLD), first detected by the Port of Miami, Florida in 2014, is the latest disease affecting over 30 species of corals in at least 30 Caribbean countries, with extremely high mortality. It is considered the most devastating coral disease ever recorded, as even minimal infection can trigger outbreaks and threaten long-term coral recovery by affecting coral recruits. Populations of the sea urchin *Diadema antillarum,* once the most abundant herbivore whose grazing reduces turf and macroalgae, have collapsed due to two extreme mortality events, in 1983-84 and 2022, leading to a severe reduction in grazing pressure across the Caribbean.

Unsustainable fishing

Coral reefs across the Caribbean are increasingly threatened by unsustainable fisheries targeting carnivorous fish, herbivores, and invertebrates. Overfishing of top predators like groupers, snappers, and sharks has disrupted food webs and reduced reef resilience. Herbivorous fish, such as parrotfish and surgeonfish, are essential to maintain a low algal cover and enhance coral recruitment. However, they have been increasingly targeted by small-scale fisheries, exacerbating algal overgrowth. Key invertebrate species, including spiny lobsters and queen conch, also face steep declines due to intense fishing, habitat degradation, pollution and weak regulations and enforcement. These combined impacts threaten marine biodiversity, ecosystem functions, and the livelihoods and food security of coastal communities throughout the region.

Invasive species

Finally, Caribbean coral reefs are under threats of invasive species, notably Lionfish (*Pterois volitans*), for almost two decades, and more recently of soft coral species *Unomia stolonifera* (first recorded in Venezuela in 2014), *Xenia umbellata* (first reported in Puerto Rico in 2023) and *Latissimia ningalooensis* (reported in Puerto Rico in 2025). These soft coral invasions have caused major ecological disruptions on Caribbean reefs by overgrowing hard corals and shifting benthic community structure, due to their resilience and rapid growth. The rapid expansion of these species might have unforeseen negative consequences for coral conservation/restoration efforts.

Recommendations

To reverse the decline of Caribbean coral reefs, coordinated action is urgently required across five key areas: integrating coral reefs into national and regional strategies, reducing local and global threats, strengthening protection and management of protected areas, maintaining and improving monitoring systems, and scaling up restoration efforts.

1. Integrate coral reef protection and restoration into climate and biodiversity strategies

Coral reefs must be embedded in global and national climate and biodiversity frameworks such as the United Nations Framework Convention on Climate Change (UNFCCC) and Convention on Biological Diversity, by supporting Governments to incorporate reefs into Nationally Determined Contributions (NDCs), National Adaptation Plans (NAPs), National Biodiversity Strategies and Action Plans (NBSAPs), and National Biodiversity Finance Plans (BFPs). As natural infrastructure and nature-based solutions, reefs provide mitigation, adaptation, and biodiversity benefits. A stronger regional and global collective effort including climate and biodiversity finance, and targeted capacity development is needed to support Caribbean states, which are composed of ⅔ of Small Island Developing States, in protecting biodiversity, reducing pollution (Cartagena Convention), strengthening regional cooperation (CARICOM, OECS, OCM, ACS), and advancing blue economy strategies that translate these commitments into tangible projects.

2. Reduce local threats and greenhouse gas emissions

Countries must focus on reducing the threats affecting coral reefs. According to global climate commitments under the UNFCCC and the Paris Agreement, all countries share responsibility for accelerating the transition to renewable energy, reducing reliance on fossil fuels, and protecting key carbon sinks such as mangroves and seagrasses. While Caribbean nations are among the least responsible for historical greenhouse gas emissions, they have nonetheless committed to advancing these goals through their own nationally determined contributions and regional initiatives. Reducing local pressures requires sustainable coastal development through Integrated Coastal Zone Management, Marine Spatial Planning, strong environmental impact assessments, and improved wastewater and solid waste management to prevent eutrophication and pollution. The spread of invasive species could be curbed by strengthening the legal and regulatory framework governing the aquarium trade. Sustainable fisheries should be promoted through seasonal closures, catch limits, and herbivore protection, while tourism must transition toward low-impact, certified sustainable models. Blended-finance initiatives, such as those under the Global Fund for Coral Reefs, can support these transitions.

3. Strengthen protection through effective area-based management tools

Achieving the Kunming–Montreal Global Biodiversity Framework "30x30" target is essential to ensure a sustainable future for coral reefs, and will require expanding and effectively managing ecologically connected Area-based Management Tools, including marine protected areas (MPAs), with enforced fully-protected zones and co-managed areas. Strengthened governance, stakeholder engagement, and community-based management are critical for long-term success. Governments should prioritize integrated management plans, ensure robust enforcement, and secure sustainable funding through partnerships. Capacity-development, educational programs and cooperation networks (*e.g.*, SPAW protected area manager network, MPA Connect, Red Golfo) should be strengthened to improve management effectiveness across the region.

4. Maintain and enhance coral reef monitoring

Comprehensive and long-term monitoring is essential for understanding reef trajectories and guiding adaptive management, including restoration. Monitoring should be spatially representative and standardized across the region, encompassing indicators such as hard coral cover by species, hard coral demography, abundance and size of key functional fish groups, water quality, invasive species, and disease prevalence. Decision-oriented monitoring frameworks should link data to management actions. Data sharing should follow FAIR (Findable, Accessible, Interoperable, Reusable) data principles, using platforms like AGRRA, ReefCloud, and MERMAID, and capacity should be built for local and citizen-led data collection. Developing or improving analytical tools that allow more straightforward analysis of the data collected would be helpful for future region-wide assessments.

5. Scale up restoration to enhance resilience

Restoration has matured into a vital tool for boosting reef recovery. Restoration efforts should be embedded in wider management plans and not implemented independently. Enhancing sexual reproduction, genetic diversity and thermotolerance and improving key ecological processes that increase early-stage coral survival can strengthen ecosystem resilience to stress. Efforts could also focus on expanding coral gardening and larval propagation, deploying artificial reefs aligned with marine spatial planning, and enhancing herbivore populations—particularly *Diadema antillarum* and alternative grazers. Partnerships with the tourism sector and innovative financing models such as "Restoration-as-a-Service" can align ecological goals with economic benefits. Scaling up these initiatives requires long-term financing, standardized monitoring, and regional collaboration to maximize ecological, social, and economic outcomes.

RÉSUMÉ EXÉCUTIF

POINTS CLÉS

- Les récifs coralliens de la région GCRMN Caraïbes couvrent **24 230 km²**, ce qui représente **9,7 %** de l'étendue des récifs coralliens du monde.
- La couverture en coraux durs a **diminué de 48 %** entre 1980 et 2024, ce qui signifie que les coraux durs n'occupent aujourd'hui que la moitié de la surface benthique qu'ils couvraient en 1980.
- La couverture en coraux durs a chuté brutalement en **1998** (-9,0 %), **2005** (-17,1 %) et **2023** (-16,9 %) en raison d'épisodes de blanchissement induits par le stress thermique, ainsi que de maladies coralliennes.
- La couverture en *Acropora* est restée faible entre 1980 et 2024, avec une moyenne d'environ **1,8 %**, très en dessous des valeurs du début des années 1970, qui avoisinaient **16 %**. Contrairement à ces espèces branchues, la couverture benthique d'*Orbicella* et de *Porites* — qui regroupent des coraux durs principalement massifs — est restée stable, voire a augmenté sur la même période. Ce changement dans la dominance des espèces de coraux durs a probablement réduit la **complexité structurelle** des récifs coralliens caribéens.
- La couverture en macroalgues a **augmenté de 84,8 %** entre 1980 et 2024, probablement en raison du déclin des espèces herbivores (ex. poissons-perroquets, oursins) et de l'augmentation des nutriments.
- La température moyenne de la surface de la mer dans les zones récifales des Caraïbes a augmenté de **+1,07 °C** entre 1985 et 2024 sous l'effet du changement climatique, ce qui correspond à un rythme de réchauffement de **0,27 °C** par décennie.
- Le nombre de personnes vivant à moins de 20 km des récifs coralliens a augmenté de **27,6 %** entre 2000 et 2020 à l'échelle régionale, soit une hausse de **13 millions de personnes**.

Introduction

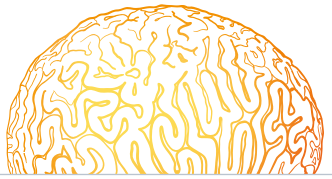
Les récifs coralliens comptent parmi les écosystèmes les plus diversifiés de la planète, abritant environ un tiers de toutes les espèces marines décrites[1]. Au-delà de leur importance écologique, ils fournissent des services écosystémiques essentiels aux sociétés humaines, en protégeant les côtes contre les tempêtes, en soutenant la sécurité alimentaire grâce aux pêcheries et en générant des revenus à travers le tourisme. Cependant, les récifs coralliens sont de plus en plus menacés par un ensemble de pressions humaines directes et indirectes, notamment le changement climatique, la pollution et les pratiques de pêche non durables.

Reconnaissant l'urgence de ces menaces, des cadres mondiaux, régionaux et nationaux, tels que le Cadre mondial de la biodiversité de Kunming-Montréal, ont été mis en œuvre afin d'assurer la conservation efficace et la gestion durable des écosystèmes coralliens. Mesurer les progrès réalisés vers ces objectifs, et ajuster les stratégies lorsque cela est nécessaire, exige des synthèses quantitatives robustes menées à grande échelle spatiale.

Créé en 1995 en tant que réseau opérationnel de l'Initiative internationale pour les récifs coralliens (ICRI), le Réseau mondial de surveillance des récifs coralliens (GCRMN) joue un rôle central dans cet effort. En fournissant les meilleures données scientifiques disponibles, le GCRMN éclaire les politiques, renforce la gestion et consolide les capacités de conservation des récifs coralliens à l'échelle mondiale. Organisé en dix nœuds régionaux, dont celui des Caraïbes, le réseau publie régulièrement des évaluations mondiales et régionales sur l'état et les tendances des récifs coralliens.

La précédente évaluation régionale pour les Caraïbes, « *Status and Trends of Caribbean Coral Reefs: 1970–2012* », a été publiée en 2014 et intégrait des données allant jusqu'en 2012. Depuis, de nombreuses études locales ont documenté la dégradation continue des récifs dans la région. Plus récemment, le rapport « *Status of Coral Reefs of the World – 2020* » a confirmé le déclin persistant des récifs caribéens. En réponse à ces observations alarmantes, le Centre d'activités régionales pour les aires spécialement protégées et la faune (SPAW-RAC) et le Programme des Nations Unies pour l'environnement (PNUE) ont lancé l'élaboration d'une nouvelle évaluation régionale, soutenue par le Fonds mondial pour les récifs coralliens (GFCR).

Le présent rapport, « *Status and Trends of Caribbean Coral Reefs: 1970–2024* », est l'aboutissement de près de deux années de travail collaboratif impliquant près de 300 contributeurs à travers la région. Il fournit une mise à jour complète sur l'évolution des principales menaces pesant sur les récifs coralliens des Caraïbes et examine les changements majeurs observés dans la composition des communautés benthiques au cours des quatre dernières décennies. Ce rapport met également en lumière les priorités urgentes pour la conservation et la gestion dans toute la région caribéenne.



[1] Fisher, R., O'Leary, R. A., Low-Choy, S., Mengersen, K., Knowlton, N., Brainard, R. E., & Caley, M. J. (2015). Species richness on coral reefs and the pursuit of convergent global estimates. Current Biology, 25(4), 500-505.

Contexte régional

La région GCRMN Caraïbes abrite 24 230 km² de récifs coralliens, ce qui représente 9,7 % de l'étendue totale des récifs coralliens dans le monde. Les récifs de cette région relèvent de la juridiction de 44 pays et territoires différents, avec une superficie récifale variant considérablement, allant de 4 km² pour Sint-Maarten à 4 931 km² pour Cuba.

Nous avons estimé qu'en 2020, 47 millions d'habitants vivaient à moins de 20 km d'un récif corallien dans les Caraïbes. Dans de nombreux États insulaires des Petites Antilles, comme la Barbade, Antigua-et-Barbuda ou Sainte-Lucie, presque toute la population vit à moins de 20 km d'un récif corallien. À l'inverse, dans certains pays continentaux comme le Mexique, le Honduras et le Venezuela, seule une petite proportion de la population totale vit près de la mer et donc des récifs coralliens. Entre 2000 et 2020, le nombre de personnes vivant à moins de 20 km d'un récif corallien dans les Caraïbes a augmenté de 27,6 %, ce qui représente 13 millions de personnes supplémentaires.

Les récifs coralliens des Caraïbes fournissent de multiples services écosystémiques, dont ceux liés aux pêcheries et au tourisme sont évalués à 6,2 milliards de dollars par an. En tenant compte des écosystèmes côtiers adjacents (mangroves et herbiers marins), leur valeur peut atteindre 15 milliards de dollars par an. Les récifs coralliens soutiennent des pêcheries diversifiées, incluant des espèces comme les vivaneaux, les mérous et les langoustes, qui constituent des sources essentielles de protéines et de micronutriments, en particulier dans les petits États insulaires en développement. Les récifs coralliens agissent également comme des barrières naturelles qui atténuent les effets de l'énergie des vagues et des ondes de tempête, protégeant ainsi les populations et les écosystèmes côtiers. Enfin, en stimulant le tourisme, et parfois même en en constituant le moteur principal, les récifs coralliens apportent d'importants avantages économiques aux populations littorales. Les récifs coralliens revêtent aussi une importance sociale et culturelle.

Méthodes utilisées

Nous avons estimé les tendances temporelles de cinq grandes catégories benthiques (coraux durs, macroalgues, algues gazonnantes, algues corallines et autre faune), ainsi que de trois genres de coraux durs (*Acropora*, *Orbicella*, *Porites*), grâce à l'intégration de 72 jeux de données. Ensemble, ces jeux de données représentent un total de 13 864 sites de suivi, sur lesquels 23 742 relevés ont été réalisés entre 1973 et 2024. Les tendances temporelles ont été estimées à l'aide de modèles d'apprentissage automatique afin de prendre en compte le biais lié à la distribution non homogène des données de suivi dans l'espace et dans le temps.

Résultats

Diminution de la couverture corallienne

Les premières données de suivi indiquent que la couverture en coraux durs dans les Caraïbes variait entre 29 % et 38 % avant 1983[2]. Bien que supérieures aux valeurs actuelles, ces estimations ne représentent pas des conditions récifales vierges dans la région, car les pressions anthropiques affectaient déjà ces écosystèmes à cette époque. Entre 1983 et 2024, nos analyses révèlent que la couverture en coraux durs a diminué de moitié, passant de 28,1 % en 1980–1985 à 14,6 % en 2019–2024. Les déclins majeurs ont coïncidé avec des épisodes de blanchissement massifs à l'échelle régionale causés par le stress thermique, notamment en 1998 (-9 %), 2005 (-17,1 %) et 2023–2024 (-16,9 %)[3]. Étant donné le nombre limité de données disponibles pour 2024, l'impact du quatrième épisode mondial de blanchissement[4], le plus récent et encore en cours, est probablement sous-estimé dans cette évaluation. En outre, des maladies affectant les coraux durs, telles que la maladie de la bande blanche ou la maladie de perte de tissus (SCTLD), ont contribué au déclin à long terme de la couverture corallienne, avec des impacts plus chroniques et spatialement variables que ceux des épisodes de blanchissement.

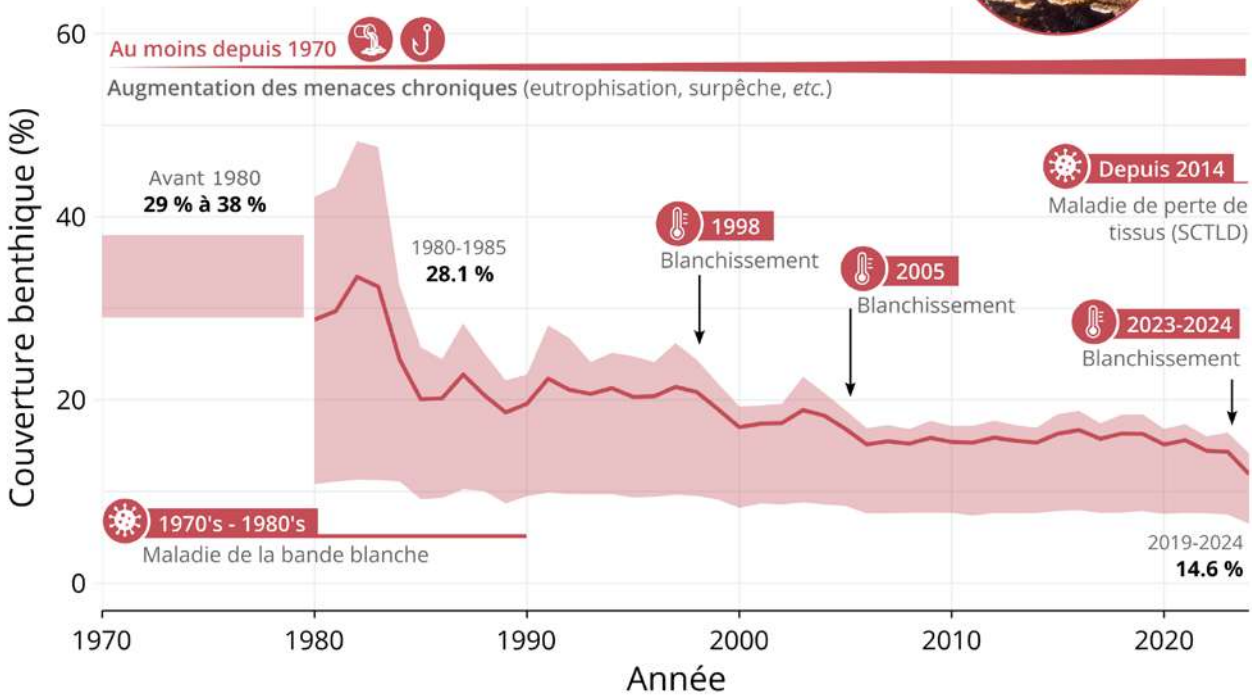
[2] Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (Eds.) (2014). Status and Trends of Caribbean Coral Reefs: 1970–2012. Global Coral Reef Monitoring Network.

[3] Les pourcentages indiquent le déclin relatif, et non absolu, de la couverture en coraux durs.

[4] Reimer, J. D., Peixoto, R. S., Davies, S. W., Traylor-Knowles, N., Short, M. L., Cabral-Tena, R. A., ... & Voolstra, C. R. (2024). The fourth global coral bleaching event: where do we go from here?. *Coral Reefs, 43*(4), 1121-1125.

Évolution de la couverture corallienne dans les Caraïbes de 1970 à 2024

La ligne en gras représente la moyenne, tandis que le ruban représente l'intervalle de confiance à 95 %



Changement de l'assemblage corallien

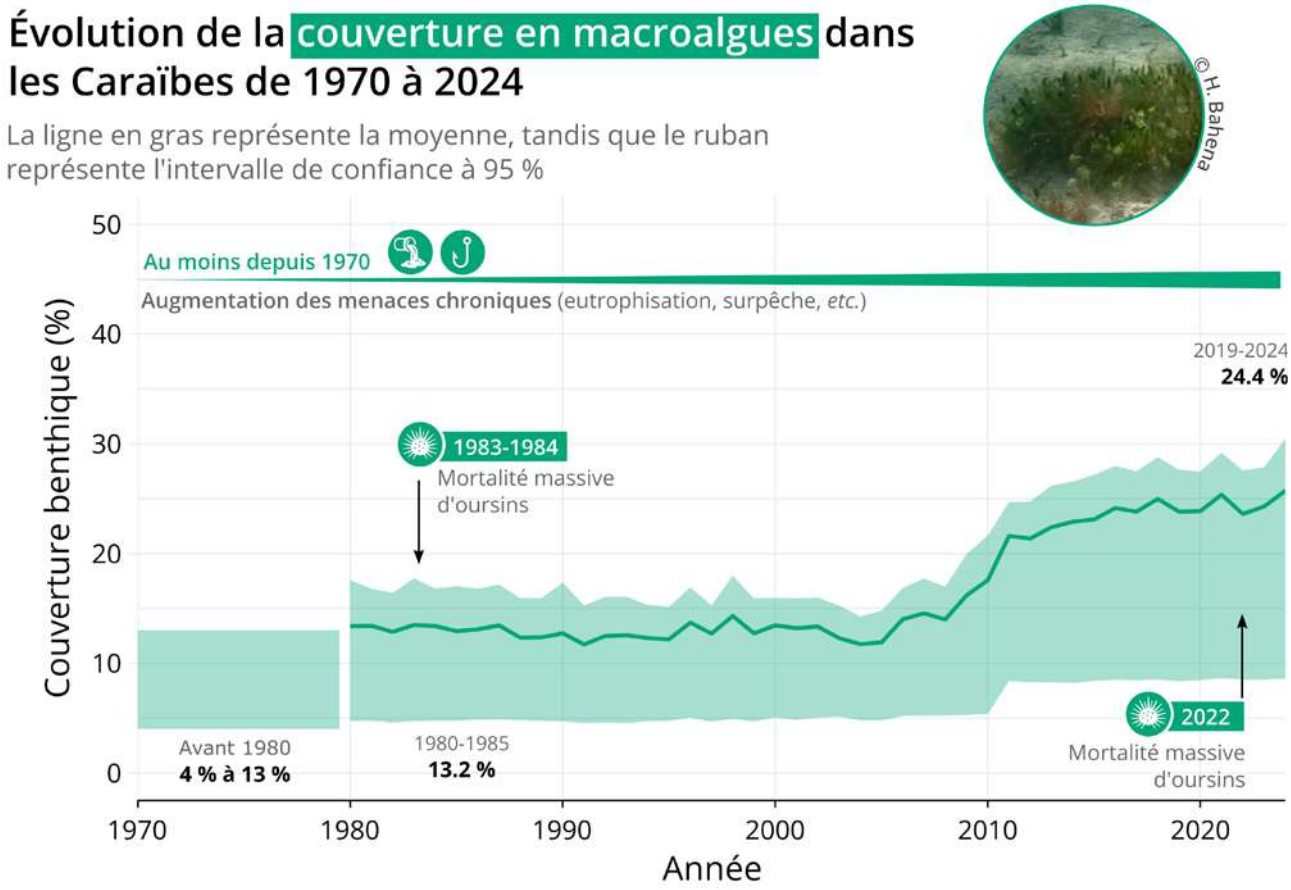
Le déclin général de la couverture en coraux durs observé sur la période d'étude masque des trajectoires contrastées parmi les principaux genres de coraux durs de la région. Des études antérieures ont montré que les *Acropora* spp., qui représentaient environ 16 % de la couverture benthique dans les années 1970, ont fortement décliné avant 1980 en raison de pressions anthropiques et d'épidémies de maladie de la bande blanche[5]. Depuis, nos résultats indiquent que la couverture benthique de ces espèces de coraux durs branchus est restée faible, se stabilisant autour de 1,8 % depuis 1980. La couverture d'*Orbicella* spp., qui comprend des espèces de coraux durs massifs, a diminué après l'épisode de blanchissement de 1998 et s'est stabilisée à partir de 2009 autour de 5 %. En revanche, la couverture de *Porites* spp. a augmenté de 164 % entre 1980 et 2024, atteignant une couverture moyenne de 3,7 % en 2019–2024. Dans l'ensemble, ces tendances suggèrent une profonde réorganisation des assemblages de coraux durs au cours des dernières décennies, marquée par la perte d'espèces branchues et l'augmentation relative de formes massives et plus tolérantes au stress. Ce changement de composition, en faveur de coraux durs massifs résistants, a probablement réduit la complexité de l'habitat et la biodiversité, affectant ainsi les services écosystémiques.

Augmentation de la couverture en macroalgues

Les premières données de suivi indiquent que les macroalgues couvraient entre 4 % et 13 % des récifs caribéens avant 1983[6]. Contrairement aux coraux durs, la couverture en macroalgues a augmenté de 84,8 % entre 1980 et 2024. Après une période de forte variabilité durant les deux premières décennies de suivi, les algues gazonnantes ont également progressé, augmentant de 64,7 % depuis 2001. Deux facteurs principaux ont été avancés pour expliquer ces tendances. Premièrement, la perte d'herbivorie à la suite du déclin des populations de poissons-perroquets et chirurgiens, combinée aux mortalités massives de l'oursin à longues épines (*Diadema*

[5] Cramer, K. L., Jackson, J. B., Donovan, M. K., Greenstein, B. J., Korpanty, C. A., Cook, G. M., & Pandolfi, J. M. (2020). Widespread loss of Caribbean acroporid corals was underway before coral bleaching and disease outbreaks. *Science Advances, 6*(17), eaax9395.

[6] Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (Eds.) (2014). Status and Trends of Caribbean Coral Reefs: 1970–2012. Global Coral Reef Monitoring Network.



antillarum) en 1983–1984 puis à nouveau en 2022. Deuxièmement, la dégradation de la qualité de l'eau — reflétée par l'augmentation des apports en nutriments, de la sédimentation et de la turbidité — qui a favorisé la prolifération des macroalgues et des algues gazonnantes. Ces deux processus ont réduit la pression de broutage tout en stimulant la croissance algale, entraînant une augmentation progressive de la couverture en macroalgues et en algues gazonnantes. Comme ces algues sont en compétition directe avec les coraux durs pour l'espace, leur expansion limite le recrutement et la croissance des coraux, entravant la récupération de la couverture corallienne dans les Caraïbes.

Augmentation des menaces

Changement climatique

Le changement climatique constitue une menace croissante pour les récifs coralliens des Caraïbes, en raison de l'augmentation des concentrations de gaz à effet de serre dans l'atmosphère au cours du siècle dernier. Entre 1985 et 2024, la température de surface de la mer au niveau des récifs coralliens des Caraïbes a augmenté en moyenne de 1,07°C, soit un réchauffement de 0,27 °C par décennie. Les récifs coralliens de tous les pays et territoires de la région ont subi ce réchauffement, les augmentations les plus importantes à long terme ayant été enregistrées au Mexique (golfe du Mexique), au Guatemala et dans le sanctuaire marin national de Flower Garden Banks, où la température de surface de la mer a augmenté d'environ 1,4°C sur la période.

Parallèlement, les vagues de chaleur marines—périodes de températures de surface anormalement élevées et prolongées—sont devenues plus fréquentes, plus intenses et plus longues en raison du changement climatique. Ces événements, en particulier ceux de 1987, 1998, 2005, 2010, 2015-2017, 2020 et 2023-2024, ont entraîné des phénomènes de blanchissement.

Les récifs coralliens des Caraïbes ont également été exposés à des ouragans plus intenses, toujours du fait du changement climatique. De 1980 à 2024, nous avons estimé que 171 ouragans sont passés à moins de 100 km des récifs coralliens, avec des pics notables en 1995 (11 ouragans), 2005 (15) et 2017 (10). Si les impacts de ces évènements météorologiques varient en fonction de leur intensité et de leur emplacement, elles entraînent souvent de graves dommages structurels aux récifs coralliens. Les projections climatiques indiquent que si la fréquence globale des ouragans devrait rester stable jusqu'à la fin du siècle, leur intensité devrait augmenter, ce qui pourrait entraîner une augmentation des dommages causés aux récifs.

Qualité de l'eau

Au-delà des impacts mondiaux du changement climatique, les récifs coralliens des Caraïbes sont de plus en plus menacés par les activités humaines locales et directes. L'augmentation significative de la population humaine dans toute la région au cours des dernières décennies a entraîné une pression accrue sur les écosystèmes côtiers. Les apports en nutriments et en sédiments provenant de l'agriculture et du développement côtier ont considérablement augmenté, contribuant à une détérioration de la qualité de l'eau et à une augmentation généralisée de la couverture en macroalgues. La pollution chimique et plastique a fortement augmenté, la densité des microplastiques dans les océans ayant doublé chaque décennie depuis 1970. Cette pollution constitue une menace sérieuse pour les récifs coralliens en exacerbant le blanchissement et les maladies. Parallèlement, l'expansion du trafic maritime, notamment le transport de marchandises et les croisières, a accru les dommages physiques tout en contribuant à la dégradation de la qualité de l'eau.

Maladies

Au cours des dernières décennies, les récifs coralliens des Caraïbes ont été confrontés à une augmentation de la fréquence et de la gravité des maladies, entraînant une mortalité généralisée parmi les coraux durs et les organismes récifaux. Liées à la hausse des températures océaniques, à la pollution et à d'autres impacts humains, ces maladies accélèrent la dégradation des récifs coralliens et la perte de biodiversité. Au cours des années 1990, plusieurs maladies sont apparues dans les Caraïbes : la maladie de la bande jaune a entraîné la disparition massive d'*Orbicella* spp., la maladie de la bande blanche a touché massivement les *Acropora* spp., tandis que la peste blanche a entraîné la disparition de plusieurs espèces. La maladie de la perte de tissus (SCTLD), détectée pour la première fois par le port de Miami, en Floride, en 2014, est la dernière maladie en date qui touche une trentaine d'espèces de coraux dans au moins 30 pays des Caraïbes, avec un taux de mortalité extrêmement élevé. Elle est considérée comme la maladie corallienne la plus dévastatrice jamais enregistrée, car même une infection minime peut déclencher une épidémie et menacer le rétablissement à long terme des coraux durs en affectant les recrues coralliennes.

Les populations d'oursins *Diadema antillarum*, autrefois les herbivores les plus abondants et dont le broutage réduit la couverture en macroalgues, se sont effondrées en raison de deux événements de mortalité massive, en 1983-1984 et en 2022, entraînant une forte réduction de la pression de broutage dans les Caraïbes.

Pêche non durable

Les récifs coralliens des Caraïbes sont de plus en plus menacés par la pêche non durable ciblant les poissons carnivores, les herbivores et les invertébrés. La surpêche des grands prédateurs tels que les mérous, les vivaneaux et les requins a perturbé les réseaux trophiques et réduit la résilience des récifs. Les poissons herbivores, tels que les poissons-perroquets et les poissons-chirurgiens sont quant à eux essentiels pour maintenir une faible couverture algale et favoriser le recrutement des coraux. Cependant, ils sont de plus en plus ciblés par les pêcheries artisanales, ce qui aggrave la prolifération des algues. Des espèces invertébrées, notamment les langoustes et les lambis, sont également confrontées à un déclin rapide en raison de la pêche intensive, de la dégradation de leur habitat, de la pollution et de la faiblesse des réglementations et de leur application. Ces impacts combinés menacent la biodiversité marine, les fonctions écosystémiques, ainsi que les moyens de subsistance et la sécurité alimentaire des communautés côtières de toute la région.

Espèces envahissantes

Enfin, les récifs coralliens des Caraïbes sont menacés depuis près de deux décennies par des espèces envahissantes, notamment le poisson-lion (*Pterois volitans*), et plus récemment par des espèces de coraux mous telles que *Unomia stolonifera* (observée pour la première fois au Venezuela en 2014), *Xenia umbellata* (signalée pour la première fois à Porto Rico en 2023) et *Latissimia ningalooensis* (signalée à Porto Rico en 2025). Ces invasions de coraux mous ont provoqué des perturbations écologiques majeures sur les récifs des Caraïbes en recouvrant les coraux durs et en modifiant la structure des communautés benthiques, en raison de leur croissance rapide. L'expansion de ces espèces pourrait avoir des conséquences négatives imprévues sur les efforts de conservation/restauration des coraux.

Recommandations

Pour inverser le déclin des récifs coralliens des Caraïbes, une action coordonnée est nécessaire de toute urgence dans cinq domaines clés : intégration des récifs coralliens dans les stratégies nationales et régionales, réduction les menaces locales et mondiales, renforcement de la protection et de la gestion des zones protégées, maintien et amélioration des programmes de suivi, et intensification des efforts de restauration.

1. Intégrer la protection et la restauration des récifs coralliens dans les stratégies sur le climat et la biodiversité

Les récifs coralliens doivent être intégrés dans les cadres mondiaux et nationaux relatifs au climat et à la biodiversité, tels que la Convention-cadre des Nations unies sur les changements climatiques (CCNUCC) et la Convention sur la diversité biologique, en aidant les gouvernements à inclure les récifs dans les contributions déterminées au niveau national (CDN), les plans d'adaptation nationaux (PAN), les stratégies et plans d'action nationaux pour la biodiversité (SPANB) et les plans nationaux de financement de la biodiversité (PFB). En tant qu'infrastructures naturelles et solutions fondées sur la nature, les récifs offrent des avantages en matière d'atténuation, d'adaptation et de biodiversité. Un effort collectif régional et mondial plus important—notamment en termes de financement pour le climat et la biodiversité et de renforcement ciblé des capacités—est nécessaire pour aider les États des Caraïbes, dont les deux tiers sont des petits États insulaires en développement, à protéger la biodiversité, à réduire la pollution (Convention de Carthagène), à renforcer la coopération régionale (CARICOM, OECO, OCM, AEC) et à promouvoir des stratégies d'économie bleue qui traduisent ces engagements en projets concrets.

2. Réduire les menaces locales et les émissions de gaz à effet de serre

Les pays doivent se focaliser sur la réduction des menaces qui pèsent sur les récifs coralliens. Conformément aux engagements climatiques mondiaux pris dans le cadre de la CCNUCC et de l'Accord de Paris, tous les pays partagent la responsabilité d'accélérer la transition vers les énergies renouvelables, de réduire la dépendance aux combustibles fossiles et de protéger les principaux puits de carbone tels que les mangroves et les herbiers marins. Bien que les pays des Caraïbes soient parmi les moins responsables des émissions historiques de gaz à effet de serre, ils se sont néanmoins engagés à faire progresser ces objectifs par le biais de leurs propres contributions déterminées au niveau national et d'initiatives régionales. La réduction des pressions locales implique un développement côtier durable grâce à la gestion intégrée des zones côtières, à l'aménagement de l'espace marin, à des évaluations rigoureuses de l'impact environnemental et à une meilleure gestion des eaux usées et des déchets solides afin de prévenir l'eutrophisation et la pollution. La propagation des espèces envahissantes pourrait être freinée en renforçant le cadre juridique et réglementaire régissant le commerce des aquariums. Il convient également de promouvoir une pêche durable grâce à des fermetures saisonnières, des limites de capture et la protection des herbivores, tandis que le tourisme doit évoluer vers des modèles certifiés durables et à faible impact. Des initiatives de financement mixte, telles que celles du Fonds mondial pour les récifs coralliens, peuvent soutenir ces transitions.

3. Renforcer la protection grâce à des outils de gestion spatialisée efficaces

La réalisation de l'objectif « 30x30 » du Cadre mondial de Kunming-Montréal pour la biodiversité est essentielle pour assurer un avenir durable aux récifs coralliens. Elle nécessitera l'extension et la gestion efficace d'outils de gestion spatialisée écologiquement connectés, notamment les aires marines protégées (AMP), avec des zones entièrement protégées et des zones cogérées. Le renforcement de la gouvernance, l'engagement des parties prenantes et la gestion communautaire sont essentiels pour garantir le succès à long terme. Les gouvernements devraient donner la priorité aux plans de gestion intégrée, veiller à leur application rigoureuse et garantir un financement durable grâce à des partenariats. Le renforcement des capacités, les programmes éducatifs et les réseaux de coopération (par exemple, le réseau des gestionnaires d'aires protégées SPAW, MPA Connect, Red Golfo) devraient être renforcés afin d'améliorer l'efficacité de la gestion dans toute la région.

4. Maintenir et développer le suivi des récifs coralliens

Un suivi complet et à long terme est essentiel pour comprendre l'évolution des récifs coralliens et orienter la gestion adaptative, y compris la restauration. Le suivi doit être représentatif spatialement et standardisé dans toute la région, et inclure des indicateurs tels que la couverture corallienne par espèce, la démographie des coraux durs, l'abondance et la taille des principaux groupes fonctionnels de poissons, la qualité de l'eau, les espèces envahissantes et la prévalence des maladies. Les cadres de suivi axés sur la prise de décision doivent lier les données aux mesures de gestion. Le partage des données doit respecter les principes FAIR (facile à trouver, accessible, interopérable, réutilisable), en utilisant des plateformes telles que AGRRA, ReefCloud et MERMAID, et il convient de renforcer les capacités en matière de collecte de données au niveau local et par les citoyens. Il serait utile pour les futures évaluations à l'échelle régionale de développer ou d'améliorer des outils analytiques permettant une analyse rigoureuse des données collectées.

5. Intensifier la restauration pour renforcer la résilience

La restauration est devenue un outil essentiel pour stimuler la régénération des récifs. Les efforts de restauration doivent s'inscrire dans des plans de gestion plus larges et ne pas être mis en œuvre de manière indépendante. Le renforcement de la reproduction sexuée, de la diversité génétique et de la thermotolérance, ainsi que l'amélioration des processus écologiques clés qui augmentent la survie des coraux à un stade précoce, peuvent renforcer la résilience des écosystèmes face au stress. Les efforts pourraient également se concentrer sur l'expansion des pépinières de coraux et la propagation des larves, le déploiement de récifs artificiels en adéquation avec la planification spatiale marine, et l'augmentation des populations d'herbivores, en particulier *Diadema antillarum* et d'autres brouteurs. Les partenariats avec le secteur du tourisme et les modèles de financement innovants tels que « Restoration-as-a-Service » (la restauration en tant que service) peuvent aligner les objectifs écologiques sur les avantages économiques. L'intensification de ces initiatives nécessite un financement à long terme, un suivi standardisé et une collaboration régionale afin de maximiser les résultats écologiques, sociaux et économiques.

RESUMEN EJECUTIVO

PUNTOS CLAVE

- Los arrecifes de coral en la región Caribe del GCRMN cubren **24,230 km²**, lo que representa el **9.7%** de la extensión mundial de arrecifes coralinos.
- La cobertura de coral pétreo **disminuyó 48%** entre 1980 y 2024, lo que significa que los corales pétreos ahora ocupan la mitad del sustrato que ocupaban en 1980.
- La cobertura de coral pétreo se redujo drásticamente en **1998** (-9.0%), **2005** (-17.1%) y en **2023** (-16.9%) debido a eventos de blanqueamiento masivos inducidos por estrés térmico, y por enfermedades coralinas.
- La cobertura de *Acropora* permaneció baja entre 1980 y 2024, con un promedio cercano a **1.8%**, muy debajo de los valores de los inicios de la década de 1970, los cuales rondaban alrededor del **16%**. En contraste con estas especies ramificadas, la cobertura bentónica de *Orbicella* y *Porites*, que incluyen corales pétreos masivos, se ha mantenido estable o incluso aumentado durante el mismo periodo. Este cambio en la dominancia de especies de coral pétreo probablemente ha reducido la **complejidad estructural** de los arrecifes del Caribe.
- La cobertura de macroalgas **aumentó un 84.8%** entre 1980 y 2024, probablemente debido al declive de especies herbívoras (por ejemplo, peces loros, erizos de mar) y por el incremento de nutrientes.
- La temperatura superficial del mar en áreas arrecifales del Caribe aumentó en **+1.07°C** entre 1985 y 2024, impulsada por el cambio climático, lo que representa una tasa de calentamiento de **0.27°C** por década.
- El número de personas que viven a menos de 20 km de los arrecifes coralinos aumentó **27.6%** del año 2000 al 2020 a escala regional, lo que representa un incremento de **13 millones de personas**.

INTRODUCCIÓN

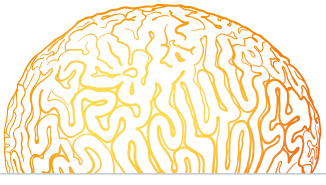
Los arrecifes de coral se encuentran entre los ecosistemas más diversos del planeta, albergando alrededor de un tercio de todas las especies marinas descritas[1]. Más allá de su importancia ecológica, brindan servicios ecosistémicos esenciales para las sociedades humanas, protegiendo las costas de las tormentas, contribuyendo a la seguridad alimentaria a través de la pesca y generando ingresos por turismo. Sin embargo, los arrecifes de coral enfrentan crecientes amenazas derivadas de diversas presiones humanas directas e indirectas, incluido el cambio climático, la contaminación y las prácticas pesqueras no sostenibles.

Ante la urgencia de estas amenazas, es necesario implementar marcos globales, regionales y nacionales, como el Marco Global de Biodiversidad de Kunming–Montreal, para asegurar la conservación efectiva y el manejo sostenible de los ecosistemas arrecifales. Medir el avance hacia estos objetivos y ajustar las estrategias cuando sea necesario requiere síntesis cuantitativas sólidas a grandes escalas espaciales.

Establecida en 1995 como una red operativa de la Iniciativa Internacional para los Arrecifes Coralinos (ICRI por sus siglas en inglés), la Red Mundial de Monitoreo de Arrecifes de Coral (GCRMN por sus siglas en inglés) desempeña un papel central en este esfuerzo. Al proporcionar la mejor información científica disponible, la GCRMN orienta las políticas, fortalece la gestión y refuerza las capacidades para la conservación de arrecifes a nivel mundial. Organizada en diez nodos regionales, incluido el Caribe, la red regularmente publica evaluaciones globales y regionales sobre el estado y las tendencias de los arrecifes de coral.

La evaluación regional previa para el Caribe, "*Estado y Tendencias de Arrecifes del Caribe: 1970–2012*", se publicó en 2014 e incorpora datos hasta el año 2012. Desde entonces, numerosos estudios locales han documentado la continua degradación de los arrecifes en la región. Recientemente, el informe *Estado de los Arrecifes del Mundo – 2020* confirmó el persistente declive de los arrecifes del Caribe. En respuesta a estas tendencias preocupantes, el Centro de Actividad Regional para Áreas y Vida Silvestre Especialmente Protegidas (SPAW-RAC por sus siglas en inglés) y el Programa de las Naciones Unidas para el Medio Ambiente (PNUMA) inició el desarrollo de una nueva evaluación regional, con el apoyo del Fondo Global para los Arrecifes de Coral (GFCR por sus siglas en inglés).

El presente informe, "*Estado y tendencias de los Arrecifes Coralinos del Caribe 1970–2024*", es el resultado de casi dos años de trabajo colaborativo que involucró a más de 300 colaboradores en la región. Ofrece una síntesis actualizada y comprensiva sobre la evolución de las principales amenazas que afectan a los arrecifes del Caribe y analiza los cambios más significativos en la composición bentónica durante las últimas cuatro décadas. Asimismo, destaca prioridades urgentes para la conservación y la gestión en el Caribe.



[1] Fisher, R., O'Leary, R. A., Low-Choy, S., Mengersen, K., Knowlton, N., Brainard, R. E., & Caley, M. J. (2015). Species richness on coral reefs and the pursuit of convergent global estimates. Current Biology, 25(4), 500-505.

CONTEXTO REGIONAL

La región del GCRMN Caribe alberga 24,230 km² de arrecifes de coral, lo que representa 9.7% de la extensión mundial de estos ecosistemas. Los arrecifes de la región se encuentran bajo la jurisdicción de 44 países y territorios, con extensiones que varían considerablemente, desde 4 km² en Sint Maarten hasta 4,931 km² en Cuba.

En 2020, en el Caribe se estima que 47 millones de habitantes viven a menos de 20 km de un arrecife coralino. En varias islas del Caribe oriental, como Barbados, Antigua y Barbuda o Santa Lucía, casi toda la población vive dentro de este rango. En contraste, en países continentales como México, Honduras y Venezuela, solo un pequeño porcentaje de la población reside cerca del mar y, por lo tanto, de los arrecifes. Entre 2000 y 2020, la población que vive a menos de 20 km de un arrecife en el Caribe aumentó 27.6%, lo que equivale a 13 millones de personas adicionales.

Los arrecifes de coral del Caribe brindan múltiples servicios ecosistémicos y se valoran en 6.2 mil millones de dólares anuales por sus beneficios en pesca y turismo. Junto con los ecosistemas costeros adyacentes (manglares y pastos marinos), su valor puede alcanzar los 15 billones de dólares anuales. Estos arrecifes sustentan pesquerías diversas, incluyendo pargos, meros y langostas espinosas, los cuales son fuentes esenciales de proteína y micronutrientes, especialmente en los pequeños Estados insulares en desarrollo. Los arrecifes también actúan como barreras naturales que mitigan los impactos de la energía del oleaje y las marejadas, protegiendo a las comunidades y ecosistemas costeros. Finalmente, al impulsar el turismo, y en algunos casos ser su motor principal, los arrecifes generan beneficios económicos significativos para las poblaciones costeras. Además, tienen una importancia social y cultural, influyendo en la forma en que las comunidades de la región dependen del entorno marino y se relacionan con él.

MÉTODOS

Estimamos las tendencias temporales de cinco categorías bentónicas principales (corales pétreos, macroalgas, algas filamentosas *turf*, algas coralinas y otra fauna), así como en tres géneros de corales pétreos (*Acropora*, *Orbicella* y *Porites*), mediante la integración de 72 bases de datos. En conjunto, estos datos representan un total de 13,864 sitios de monitoreo, en los cuales se realizaron 23,742 evaluaciones entre 1973 y 2024. Las tendencias temporales se estimaron utilizando modelos de aprendizaje automatizados para considerar el sesgo derivado de la distribución no homogénea de los datos de monitoreo en el espacio y en el tiempo.

RESULTADOS

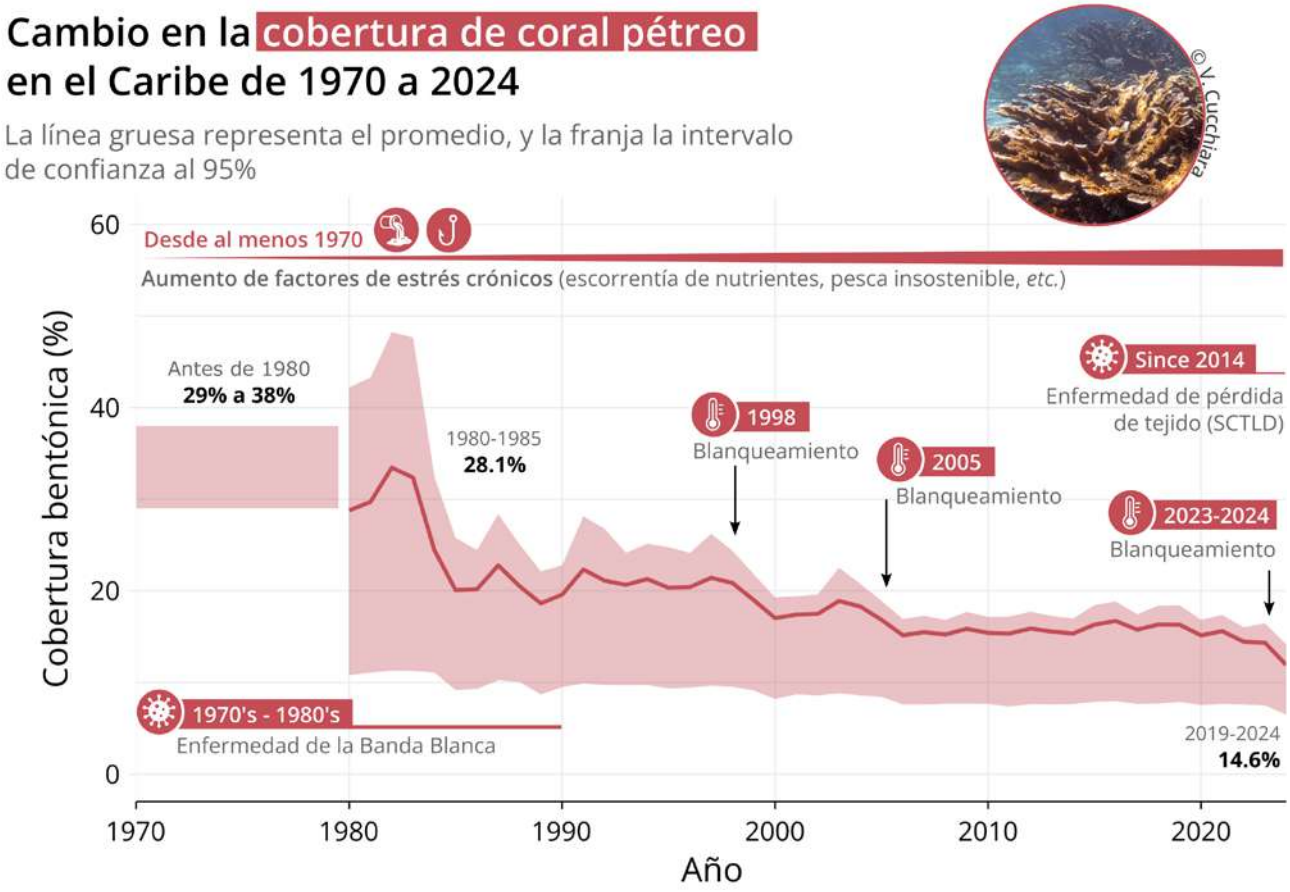
Reducción en la cobertura de coral pétreo

Los datos de los monitoreos en los años 1983, indicaron que la cobertura de coral pétreo en el Caribe fluctuó entre 29% y 38%[2]. Aunque estos valores son superiores a los niveles actuales, no representan condiciones prístinas de los arrecifes de la región, ya que los factores de estrés de origen humano ya estaban afectando a estos ecosistemas en ese momento. Entre 1983 y 2024, nuestros análisis muestran que la cobertura de coral pétreo se ha reducido a la mitad, pasando de 28.1% en 1980–1985 a 14.6% en 2019–2024. Las mayores disminuciones coincidieron con eventos de blanqueamiento masivo a escala regional impulsados por estrés térmico, destacando los de 1998 (-9%), 2005 (-17.1%) y 2023–2024 (-16.9%)[3]. Dado a el número limitado de datos disponibles para el año 2024, es probable que el impacto del cuarto evento de blanqueamiento global más reciente[4], el cual aún se encuentra en curso, esté subestimado en esta evaluación. Adicionalmente, enfermedades de corales pétreos como la enfermedad de la banda blanca y la enfermedad de pérdida de tejido de coral, han contribuido al declive a largo plazo de la cobertura de coral, con impactos más crónicos y espacialmente variables que los eventos de blanqueamiento.

[2] Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (Eds.) (2014). Status and Trends of Caribbean Coral Reefs: 1970–2012. Global Coral Reef Monitoring Network.

[3] The percentages indicate the relative decline, not the absolute decline, in hard coral cover.

[4] Reimer, J. D., Peixoto, R. S., Davies, S. W., Traylor-Knowles, N., Short, M. L., Cabral-Tena, R. A., ... & Voolstra, C. R. (2024). The fourth global coral bleaching event: where do we go from here?. Coral Reefs, 43(4), 1121-1125.



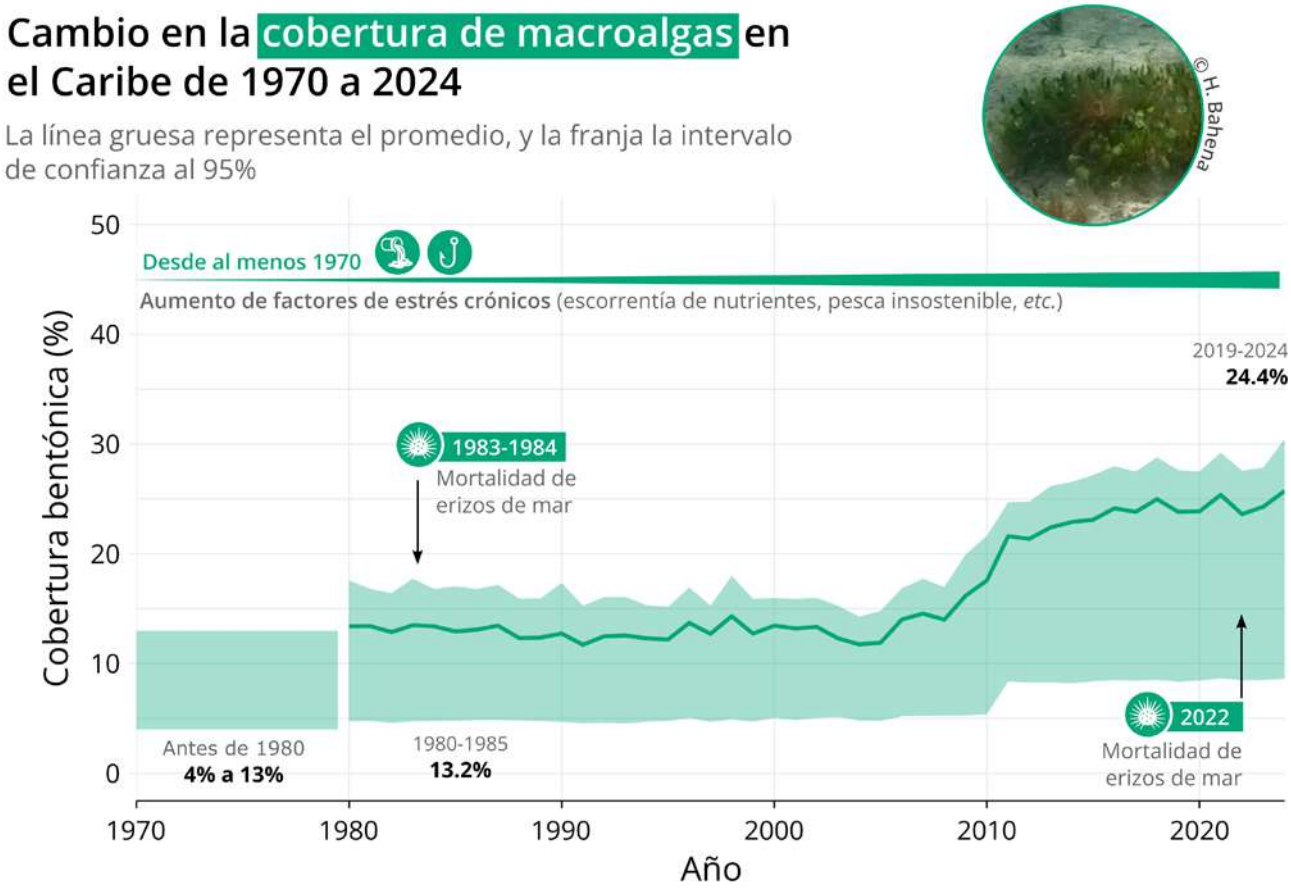
Cambios en la composición de corales pétreos

El declive general de la cobertura de coral pétreo observado durante el período de estudio oculta trayectorias contrastantes entre los principales géneros de corales pétreos en la región. Estudios previos han mostrado que *Acropora* spp., que representaba aproximadamente el 16% de la cobertura bentónica en la década de 1970, disminuyó abruptamente antes de 1980 debido a factores de estrés antropogénicos y brotes generalizados de la enfermedad de la banda blanca. Desde entonces, nuestros resultados indican que la cobertura bentónica de estas especies ramificadas de corales pétreos se ha mantenido baja desde 1980, con un promedio de alrededor del 1.8%. La cobertura de *Orbicella* spp., que incluye especies masivas de corales pétreos, disminuyó tras el evento de blanqueamiento del 1998 y se estabilizó a partir del 2009 en aproximadamente 5%. En contraste, la cobertura de *Porites* spp. aumentó en un 164% entre 1980 y 2024, alcanzando un promedio de 3.7% en el período 2019–2024. En conjunto, estas tendencias sugieren una profunda reorganización de los ensamblajes de corales pétreos en las últimas décadas, marcada por la pérdida de especies ramificadas y el aumento relativo de formas masivas más tolerantes al estrés. Este cambio composicional hacia corales pétreos masivos y tolerantes probablemente ha reducido la complejidad del hábitat y la biodiversidad, afectando así los servicios ecosistémicos.

Incremento en la cobertura de macroalgas

Los datos de monitoreo antes del año 1983, indicaron que las macroalgas cubrían entre el 4% y el 13% de cobertura en los arrecifes del Caribe[5]. En contraste con los corales pétreos, la cobertura de macroalgas ha aumentado un 84.8% entre 1980 y 2024. Tras un período de alta variabilidad en las dos primeras décadas de monitoreo, las algas filamentosas (*turf*) también se expandieron, aumentando un 64.7% desde 2001. Se han propuesto dos factores principales para explicar estas tendencias. Primero, la pérdida de herbivoría tras el declive de las poblaciones de peces loro y peces cirujanos, junto con las mortalidades masivas del erizo de espinas largas (*Diadema antillarum*)

[5] Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (Eds.) (2014). Status and Trends of Caribbean Coral Reefs: 1970–2012. Global Coral Reef Monitoring Network.



en los años 1983–1984 y nuevamente en el año 2022. Segundo, la disminución en la calidad del agua, reflejada en mayores cargas de nutrientes, sedimentación y turbidez, ha favorecido aún más la proliferación de macroalgas y algas filamentosas. En conjunto, estos procesos han reducido la presión de los procesos de herbivoría clave, que al mismo tiempo estimulan el crecimiento algal, lo que ha llevado a aumentos sostenidos en la cobertura de macroalgas y algas filamentosas *turf*. Debido a que estas algas compiten directamente con los corales pétreos por espacio, esta expansión limita aún más el reclutamiento y crecimiento de corales, obstaculizando la recuperación de la cobertura de coral pétreo en todo el Caribe.

INCREMENTO DE LAS AMENAZAS

Cambio Climático

El cambio climático es una amenaza creciente para los arrecifes coralinos del Caribe, impulsada por el aumento de las concentraciones atmosféricas de gases de efecto invernadero durante el último siglo. Entre 1985 y 2024, las temperaturas superficiales del mar en los arrecifes caribeños aumentaron en un promedio de 1.07°C, lo que representa una tasa de calentamiento de 0.27°C por década. Los arrecifes de todos los países y territorios de la región experimentaron este calentamiento, con los mayores aumentos registrados en México (Golfo de México), Guatemala y el Flower Garden Banks National Marine Sanctuary, donde las temperaturas superficiales del mar aumentaron aproximadamente 1.4°C. Paralelamente, las olas de calor marinas —periodos de temperaturas superficiales del mar inusualmente altas y prolongadas— se han vuelto más frecuentes, intensas y duraderas bajo el cambio climático. Estos eventos, especialmente los ocurridos en los años 1987, 1998, 2005, 2010, 2015–2017, 2020 y 2023–2024, han provocado episodios de blanqueamiento coralino. Los arrecifes del Caribe también han estado expuestos a huracanes más intensos como consecuencia del cambio climático. Entre 1980 y 2024, estimamos que 171 huracanes pasaron a menos de 100 km de los sistemas arrecifales, con picos notables en los años 1995 (11 huracanes), 2005 (15) y 2017 (10). Si bien los impactos de estas tormentas varían según su intensidad y ubicación, a menudo resultan en daños estructurales severos a los arrecifes. Las proyecciones climáticas indican que, aunque la frecuencia total de huracanes podría mantenerse estable hacia finales de siglo, su intensidad probablemente aumentará, lo que podría generar daños cada vez mayores en los arrecifes de coral.

Calidad del agua

Más allá de los impactos globales del cambio climático, los arrecifes coralinos del Caribe se ven cada vez más amenazados por las actividades humanas locales y directas. El aumento significativo de la población humana en toda la región en las últimas décadas ha provocado una mayor presión sobre los ecosistemas costeros. Las aportaciones de nutrientes y sedimentos procedentes de la agricultura y el desarrollo costero han aumentado considerablemente, lo que ha contribuido a la disminución de la calidad del agua y al aumento generalizado de la cobertura de macroalgas. La contaminación química y plástica ha aumentado considerablemente, y la densidad de microplásticos en los océanos se ha duplicado cada década desde 1970. Esta contaminación supone una grave amenaza para los arrecifes de coral, ya que agrava el blanqueamiento y las enfermedades. Paralelamente, la expansión del tráfico marítimo, incluido el transporte de mercancías y los cruceros, ha aumentado los daños físicos y ha contribuido a la degradación de la calidad del agua.

Enfermedades

Los arrecifes coralinos del Caribe han enfrentado brotes de enfermedades cada vez más frecuentes y graves en las últimas décadas, lo que ha provocado mortalidades generalizadas entre los corales pétreos y organismos arrecifales. Estas enfermedades, relacionadas con el aumento de la temperatura del mar, la contaminación y otros impactos humanos, están acelerando la degradación de los arrecifes y la pérdida de biodiversidad. Durante la década de 1990, surgieron varias enfermedades en el Caribe: la enfermedad de la banda amarilla provocó una pérdida masiva de *Orbicella* spp., la enfermedad de la banda blanca afectó enormemente a *Acropora*, mientras que la plaga blanca provocó la pérdida de múltiples especies. La enfermedad de pérdida de tejido de los corales pétreos (SCTLD), detectada por primera vez en el puerto de Miami, Florida, en 2014, es la enfermedad más reciente que ha afectado a más de 30 especies de corales en al menos 30 países del Caribe, con una mortalidad extremadamente alta. Se considera la enfermedad de los corales más devastadora que se ha registrado, ya que incluso una infección mínima puede desencadenar brotes y amenazar la recuperación a largo plazo, al afectar a los reclutas de corales. Las poblaciones del erizo de mar *Diadema antillarum*, el cual fue uno de los herbívoros más abundantes y cuya herbivoría reduce los césped de algas turf y las macroalgas, se han colapsado debido a dos episodios de mortalidad extrema, en los años 1983-84 y 2022, lo que ha provocado una grave reducción de la herbivoría en todo el Caribe.

Pesca no sostenible

Los arrecifes de coral del Caribe se ven cada vez más amenazados por la pesca insostenible dirigida a peces carnívoros, herbívoros e invertebrados. La sobrepesca de depredadores superiores como meros, pargos y tiburones ha alterado las redes tróficas y reducido la resiliencia de los arrecifes. Los peces herbívoros, como los peces loro y los peces cirujano, son esenciales para mantener una baja cobertura de algas y mejorar el reclutamiento de corales. Sin embargo, son cada vez más objeto de la pesca a pequeña escala, lo que agrava el crecimiento excesivo de algas. Especies clave de invertebrados, como las langostas espinosas y el caracol reina, también se enfrentan a un fuerte declive debido a la pesca intensiva, la degradación del hábitat, la contaminación y la aplicación laxa de las regulaciones. Estos impactos combinados amenazan la biodiversidad marina, las funciones de los ecosistemas y los medios de vida y la seguridad alimentaria de las comunidades costeras de toda la región.

Especies invasoras

Por último, los arrecifes de coral del Caribe se ven amenazados por especies invasoras, en particular el pez león (*Pterois volitans*), desde hace casi dos décadas, y más recientemente por especies de corales blandos como *Unomia stolonifera* (registrada por primera vez en Venezuela en 2014), *Xenia umbellata* (registrada por primera vez en Puerto Rico en 2023) y *Latissimia ningalooensis* (registrada en Puerto Rico en 2025). Estas invasiones de corales blandos han causado importantes alteraciones ecológicas en los arrecifes del Caribe al crecer en exceso sobre los corales duros y cambiar la estructura de la comunidad bentónica, debido a su resistencia y rápido crecimiento. La rápida expansión de estas especies podría tener consecuencias negativas imprevistas para los esfuerzos de conservación y restauración de los corales.

RECOMENDACIONES

Para revertir el deterioro de los arrecifes coralinos del Caribe, es necesario adoptar medidas coordinadas con carácter urgente en cinco ámbitos clave: integrar los arrecifes de coral en las estrategias nacionales y regionales, reducir las amenazas locales y globales, reforzar la protección y la gestión de las zonas protegidas, mantener y mejorar los sistemas de monitoreo, y escalar las iniciativas de restauración adecuadas.

1. Integrar la protección y restauración de los arrecifes coralinos en las estrategias climáticas y de biodiversidad

Los arrecifes de coral deben integrarse en los marcos climáticos y de biodiversidad mundiales y nacionales, como la Convención Marco de las Naciones Unidas sobre el Cambio Climático (CMNUCC) y el Convenio sobre la Diversidad Biológica, apoyando a los gobiernos para que incorporen los arrecifes en las contribuciones determinadas a nivel nacional (NDC), los planes nacionales de adaptación (NAP), las estrategias y planes de acción nacionales sobre biodiversidad (NBSAP) y los planes nacionales de financiación de la biodiversidad (BFP). Como infraestructura natural y fuente de soluciones basadas en la naturaleza, los arrecifes proporcionan beneficios en materia de mitigación, adaptación y biodiversidad. Se necesita un esfuerzo colectivo regional y mundial más intenso, que incluya la financiación del clima y la biodiversidad y el fortalecimiento de capacidades específicas, para apoyar a los Estados del Caribe, que están compuestos por de los pequeños Estados insulares en desarrollo, en la protección de la biodiversidad, la reducción de la contaminación (Convención de Cartagena), el fortalecimiento de la cooperación regional (CARICOM, OECO, OCM, ACS) y el avance de estrategias de economía azul que traduzcan estos compromisos en proyectos tangibles.

2. Reducir las amenazas locales y las emisiones de gases de efecto invernadero

Los países deben centrarse en reducir las amenazas que afectan a los arrecifes de coral. De acuerdo con los compromisos climáticos globales en virtud de la CMNUCC y el Acuerdo de París, todos los países comparten la responsabilidad de acelerar la transición hacia las energías renovables, reducir la dependencia de los combustibles fósiles y proteger los sumideros de carbono clave, como los manglares y las praderas marinas. Si bien las naciones del Caribe se encuentran entre las menos responsables de las emisiones históricas de gases de efecto invernadero, se han comprometido a promover estos objetivos a través de sus propias contribuciones determinadas a nivel nacional e iniciativas regionales. La reducción de las presiones locales requiere un desarrollo costero sostenible mediante la gestión integrada de las zonas costeras, la planificación espacial marina, evaluaciones rigurosas del impacto ambiental y una mejor gestión de las aguas residuales y los residuos sólidos para prevenir la eutrofización y la contaminación. La propagación de especies invasoras se podría frenar, reforzando el marco jurídico y normativo que regula el comercio de acuarios. Debe promoverse la pesca sostenible mediante vedas estacionales, límites de captura y protección de los herbívoros, mientras que el turismo debe pasar a modelos certificados como sostenibles y de bajo impacto. Las iniciativas de financiación combinada, como las del GFCR pueden apoyar estas transiciones.

3. Fortalecer la protección mediante medidas efectivas de conservación basadas en áreas

Alcanzar el objetivo «30x30» del Marco Mundial para la Diversidad Biológica de Kunming-Montreal es esencial para garantizar un futuro sostenible para los arrecifes coralinos. Este requiere la ampliación y la gestión eficaz de las medidas efectivas de conservación basadas en áreas que se encuentren ecológicamente conectadas, incluidas las áreas marinas protegidas (AMP), con zonas totalmente protegidas y áreas de co-manejo. El fortalecimiento de la gobernanza, la participación de las partes interesadas y la gestión basada en la comunidad son fundamentales para el éxito a largo plazo. Los gobiernos deben dar prioridad a los planes de gestión integrada, garantizar una aplicación rigurosa y asegurar una financiación sostenible mediante asociaciones. Deben reforzarse el fortalecimiento de capacidades, los programas educativos y las redes de cooperación (por ejemplo, la red de gestores de áreas protegidas SPAW, MPA Connect, Red Golfo) para mejorar la eficacia de la gestión en toda la región.

4. Mantener y mejorar el monitoreo de los arrecifes de coral

El monitoreo integral y a largo plazo es esencial para comprender las trayectorias de los arrecifes y orientar la gestión adaptativa, incluida la restauración. El monitoreo debe ser espacialmente representativo y estandarizado en toda la región, y abarcar indicadores como la cobertura de corales pétreos por especie, la demografía de los corales pétreos, la abundancia y el tamaño de los grupos de peces funcionales clave, la calidad del agua, las especies invasoras y la prevalencia de enfermedades. Los marcos de seguimiento orientados a la toma de decisiones deben vincular los datos con las medidas de gestión. El intercambio de datos debe seguir los principios FAIR por sus siglas en inglés (localizables, accesibles, interoperables y reutilizables), utilizando plataformas como AGRRA, ReefCloud y MERMAID, y se debe fortalecer la capacidad para la recopilación de datos a nivel local y dirigida por los ciudadanos. El desarrollo o la mejora de herramientas analíticas que permitan un análisis más sencillo de los datos recopilados es indispensable para futuras evaluaciones a nivel regional.

5. Escalar la restauración para mejorar la resiliencia

La restauración se ha convertido en una herramienta fundamental para impulsar la recuperación de los arrecifes. Las iniciativas de restauración deben integrarse en planes de gestión más amplios y no aplicarse de forma independiente. Mejorar la reproducción sexual, la diversidad genética y la tolerancia térmica, así como los procesos ecológicos clave que aumentan la supervivencia de los corales en sus primeras etapas, puede reforzar la resiliencia del ecosistema frente al estrés. Los esfuerzos también podrían centrarse en ampliar los jardines de corales y la propagación de larvas, desplegar arrecifes artificiales alineados con la planificación espacial marina y mejorar las poblaciones de herbívoros, en particular *Diadema antillarum* y otros herbívoros. Las asociaciones con el sector turístico y los modelos de financiación innovadores, como "Restauración-como-servicio", pueden alinear los objetivos ecológicos con los beneficios económicos. El escalamiento de estas iniciativas requiere financiación a largo plazo, un seguimiento estandarizado y la colaboración regional para maximizar los resultados ecológicos, sociales y económicos.

LIST OF ACRONYMS

ABMT	Area-Based Management Tools
ACA	Allen Coral Atlas
AGRRA	Atlantic and Gulf Rapid Reef Assessment
CI	Confidence Interval
EEZ	Exclusive Economic Zone
ENSO	El Niño Southern Oscillation
GBM	Gradient Boosting Machine
GCRMN	Global Coral Reef Monitoring Network
GEE	Google Earth Engine
HRHP	Healthy Reefs for Healthy People
ICRI	International Coral Reef Initiative
IPCC	Intergovernmental Panel on Climate Change
ML	Machine Learning
MPA	Marine Protected Area
NBSAP	National Biodiversity Strategies and Action Plans
NAP	National Adaptation Plans
NDC	Nationally Determined Contributions
NOAA	National Ocean and Atmospheric Administration
RMSE	Root Mean Squared Error
SCTLD	Stony Coral Tissue Loss Disease
SIDS	Small Island Developing State
SST	Sea Surface Temperature
SPAW-RAC	Specially Protected Areas and Wildlife - Regional Activity Center
WORMS	World Register of Marine Species
WRI	World Resources Institute

INTRODUCTION

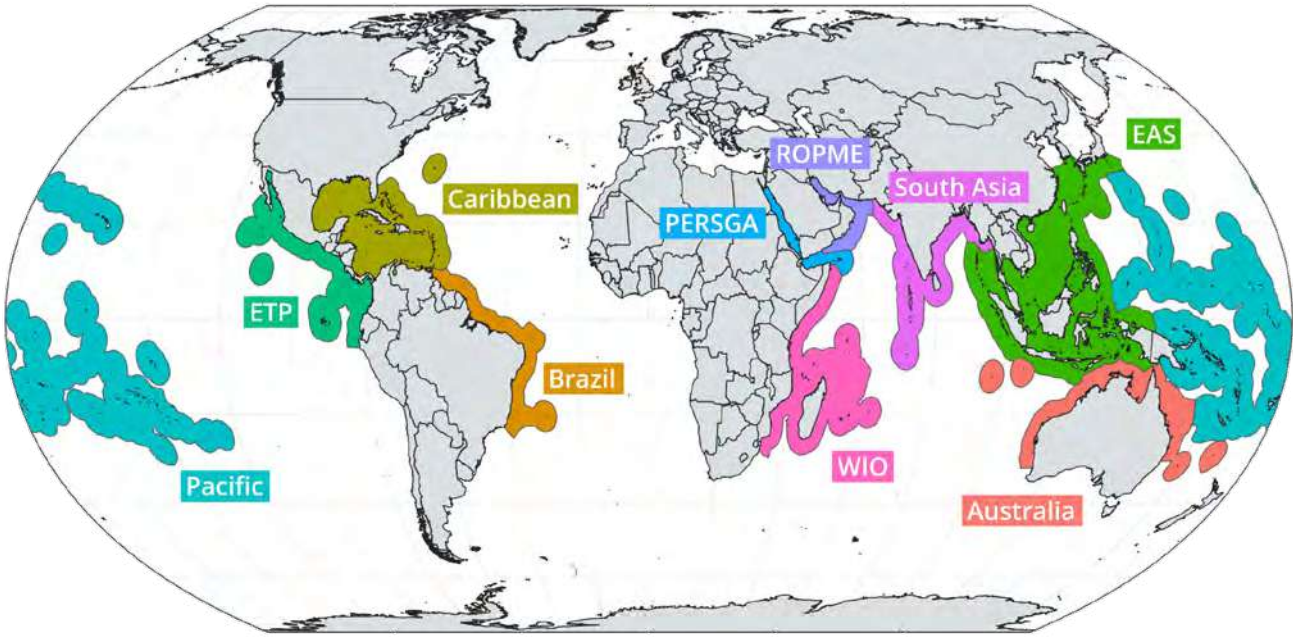
Gorgonians © Valentina Cucchiara

CONTEXT AND OBJECTIVES OF THE GCRMN

In the Anthropocene, coral reefs are exposed to a wide range of direct and indirect anthropogenic stressors (Good et al., 2021). The cumulative impacts of these pressures have profoundly altered, and continue to alter, the structure, composition, and functioning of reef ecosystems (Hughes et al., 2017). Numerous studies have documented widespread declines in hard coral cover (Emslie et al., 2024), fish biomass (Rummer and Illing, 2022), and coral accretion rates (Moris et al., 2022) across reefs worldwide, including those located in remote areas with minimal direct human influence (Baumann et al., 2022). These anthropogenic stressors not only compromise biodiversity but also threaten the ecosystem services that coral reefs provide to human populations (Woodhead et al., 2019).

To reverse, or limit, the global decline of coral reefs, Nations have developed national strategies or commitments, including National Biodiversity Strategies and Action Plans whilst Parties to global multilateral environmental agreements have adopted frameworks and international targets. For example, Parties to the Convention on Biological Diversity (CBD) adopted the Aichi targets (2011 - 2020) in October 2010 including Aichi Target 10 that aimed to minimize the anthropogenic pressures affecting coral reefs to maintain their integrity and function (UNEP/CBD/COP/10/27, 2011). In December 2022, Parties to the CBD adopted the Kunming-Montreal Global Biodiversity Framework, which sets ambitious targets and goals for 2030, and 2050, of which many targets are highly applicable to coral reefs (CBD/COP/DEC/15/4, 2022). Quantitative syntheses at large spatial scales are crucial to measure targeted progress of these international and national initiatives (Orr et al., 2022).

The Global Coral Reef Monitoring Network (GCRMN) aims to produce regular syntheses about the status and trends of coral reefs to inform policy makers, managers, and scientists. Created in 1995, as an operational network of the International Coral Reef Initiative (ICRI), the GCRMN operates through 10 regional nodes: Australia, Brazil, Caribbean, East Asian Seas (EAS), Eastern Tropical Pacific (ETP), Pacific, Red Sea and Gulf of Aden (PERSGA), Regional Organization for the Protection of the Marine Environment (ROPME) Sea Area, South Asia, and Western Indian Ocean.



Map of the ten GCRMN regions. EAS: East Asian Seas, ETP: Eastern Tropical Pacific, PERGSA: Red Sea and Gulf of Aden, ROPME: Regional Organization for the Protection of the Marine Environment, WIO: Western Indian Ocean.

The GCRMN was established under three primary goals, with capacity building as a fourth, overarching priority, notably to:

- Improve understanding of coral reef status and trends, globally and regionally,
- Analyze and communicate coral reef biophysical, social and economic trends, providing science-based recommendations in support of raising awareness, management and policy development,
- Enable and facilitate greater utilization of coral reef data, including in research,
- Build human and technical capacity to collect, analyze and report biophysical and socio-economic data on coral reefs.

HISTORY OF GCRMN REPORTS

Since its foundation, six global reports in 1998 (Wilkinson, 1998), 2000 (Wilkinson, 2000), 2002 (Wilkinson, 2002), 2004 (Wilkinson, 2004), 2008 (Wilkinson, 2008) were published, with the most recent, and the first global quantitative analysis, being released in 2021—the Status of Coral Reefs of the World: 2020 (Souter et al., 2021). In addition to these global reports, the GCRMN has also published multiple regional and topical reports on the status and trends of coral reefs of the Caribbean (*e.g.* Jackson et al., 2014), the Western Indian Ocean (Obura et al., 2017), the Pacific (*e.g.* Moritz et al., 2018; Wicquart et al., 2025), and the East Asian Seas (*e.g.* Kimura et al., 2022). Regional reports provide information at the level of countries and territories. Hence, contrary to global reports which target high-level policy decision-makers, regional reports are mainly targeting an audience of national policy-makers and managers.

THE 2014 GCRMN CARIBBEAN REPORT

The last GCRMN regional report for the Caribbean, "Status and Trends of Caribbean Coral Reefs: 1970–2012", was published in 2014 and included data up to 2012 (Jackson et al., 2014). The main findings of this report highlighted a dramatic decline in hard coral cover across coral reefs of the region, occurring in parallel with a marked increase in macroalgae cover. The report also documented significant declines in *Acropora* spp. cover, *Diadema antillarum* densities, and large parrotfish abundances. Multiple drivers—including overfishing, ocean warming, and coastal pollution—were identified as key contributors to the changes observed in Caribbean coral reefs over the 40-year period. The "Status and Trends of Caribbean Coral Reefs: 1970–2012" report provided a comprehensive synthesis of regional coral reef changes and established an essential scientific foundation for the development of effective conservation and management strategies.

Since no comparable synthesis has been produced for the Caribbean in the past decade, the editors, supported by several dozen co-authors from the region, have undertaken the publication of the current "*Status and Trends of Caribbean Coral Reefs: 1970–2024*" GCRMN report.

OBJECTIVES OF THE PRESENT REPORT

The objectives of the present report are to describe and analyze the changes that have occurred on shallow coral reefs across the wider Caribbean GCRMN region from 1970 to 2024. Specifically, the report aims to address two primary questions:

1. How have threats to coral reefs in the Caribbean changed between 1970 and 2024? To examine this question, we quantified long-term changes in the thermal regime and hurricane indicators over coral reefs. In addition, we compiled qualitative syntheses of other significant threats, such as invasive species, coral diseases, and other stressors known to impact Caribbean reef ecosystems.

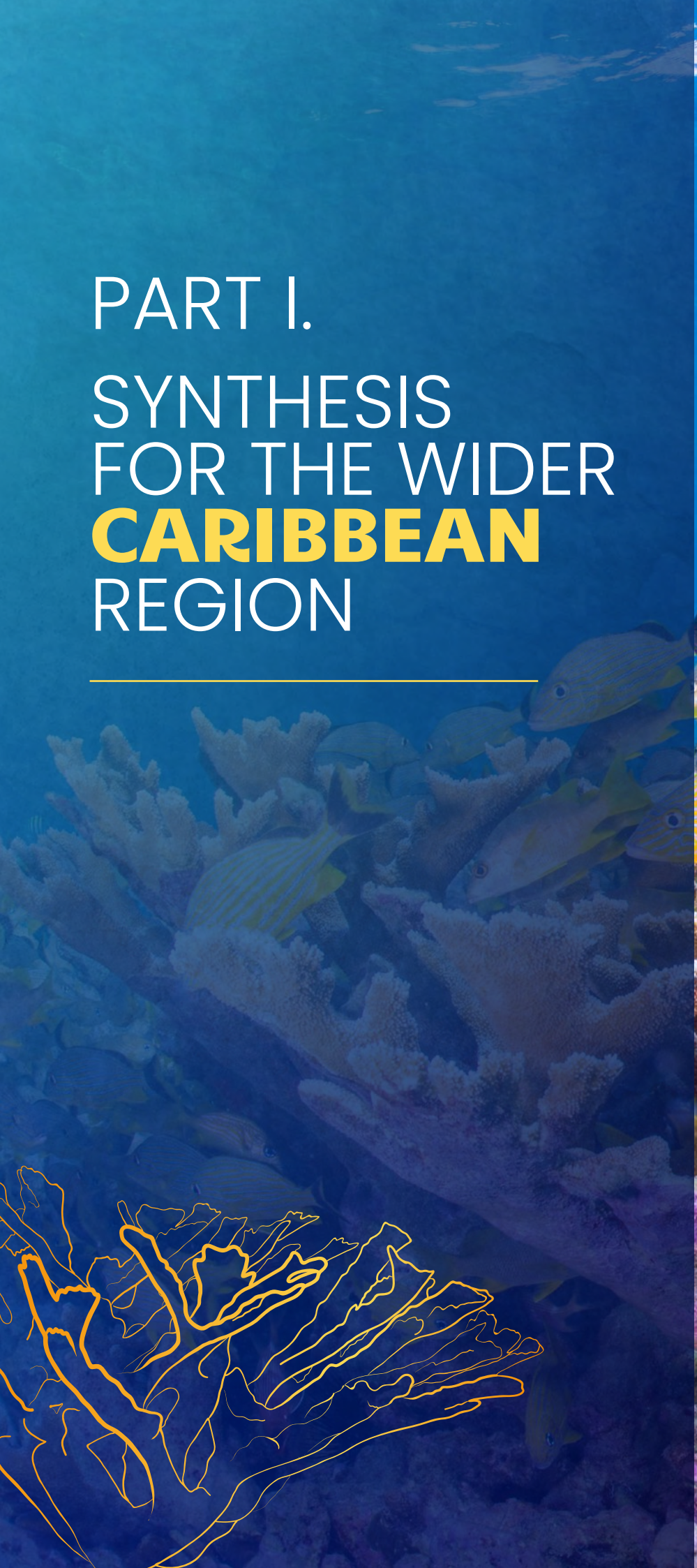
2. How has the percentage cover of major benthic categories changed over this period? To address this question, eight biological indicators associated with coral reef benthic organisms were examined. In particular, temporal trends were reported for the benthic cover of hard coral, macroalgae, turf algae, coralline algae, and other benthic fauna. Additionally, temporal trends in the cover of three key hard coral genera—*Acropora*, *Orbicella*, and *Porites*—were analyzed to assess broad shifts in the composition of the hard coral assemblage.

STRUCTURE OF THE CURRENT REPORT

As for previous GCRMN regional reports, the present document is structured into two main parts. **Part 1 – Synthesis for the wider Caribbean region** provides an overview of coral reef conditions across the wider Caribbean. It is organized into four sections: a general description of the region's coral reefs, an outline of the main threats they face, an analysis of temporal trends in the percentage cover of key benthic categories, and a set of recommendations aimed at improving the protection and management of these ecosystems. **Part 2 – Syntheses for countries and territories** compiles individual assessments of coral reefs for each of the 44 Caribbean countries and territories. Each chapter includes a brief introduction, a quantitative description of the thermal regime and hurricane activity of coral reef environments, and a qualitative evaluation of other major threats. Chapters conclude with the results of estimated temporal trends in hard coral and macroalgal cover. Case studies are presented at various points in the report to highlight specific aspects that were not addressed by the main results, such as the effectiveness of management measures or local perceptions of coral reef condition. The report also includes a **Supplementary Materials** chapter, which provides further detail on the results presented, and a **Materials and Methods** chapter, which describes the data sources and analytical approaches that were used to obtain the results presented in this study.

PART I.

SYNTHESIS FOR THE WIDER **CARIBBEAN** REGION



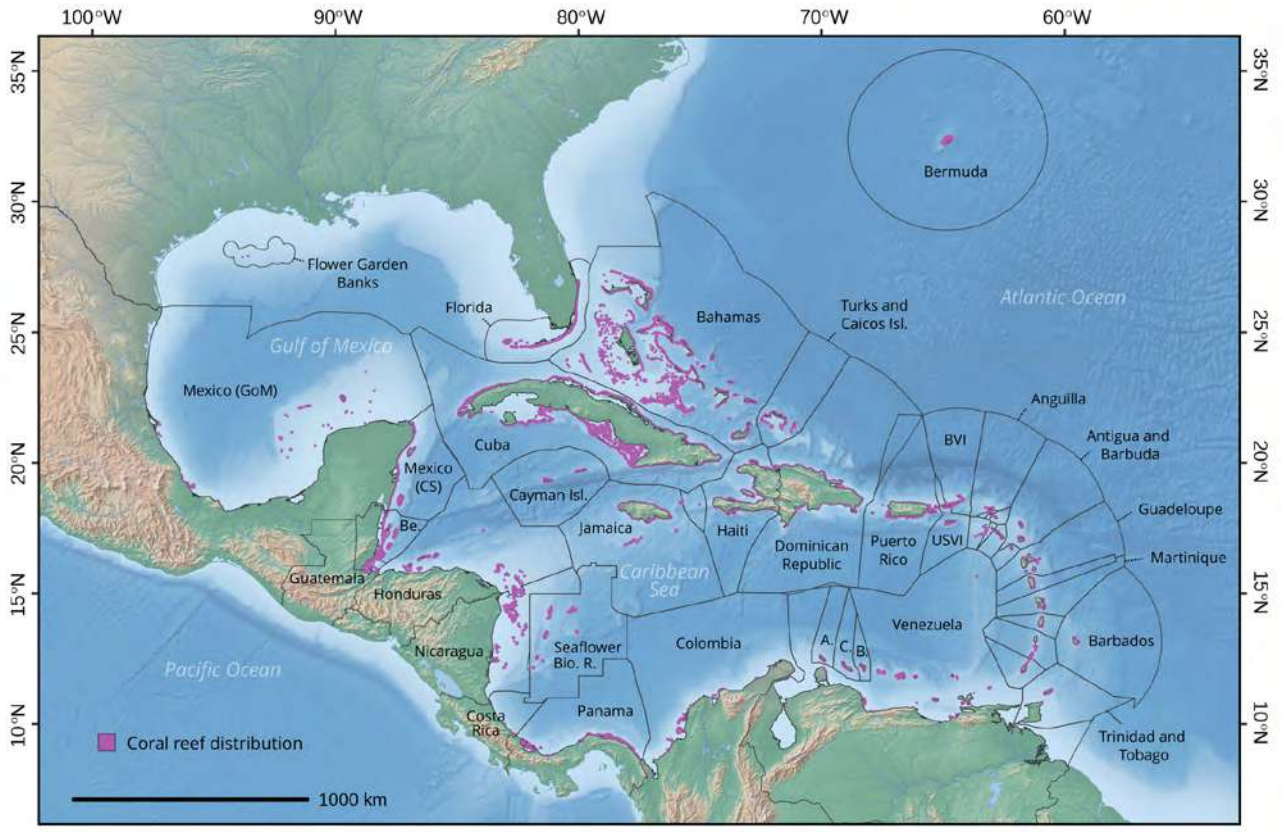
Acropora palmata © Michael Aw / Ocean Image Bank

1. Coral reefs in the Caribbean

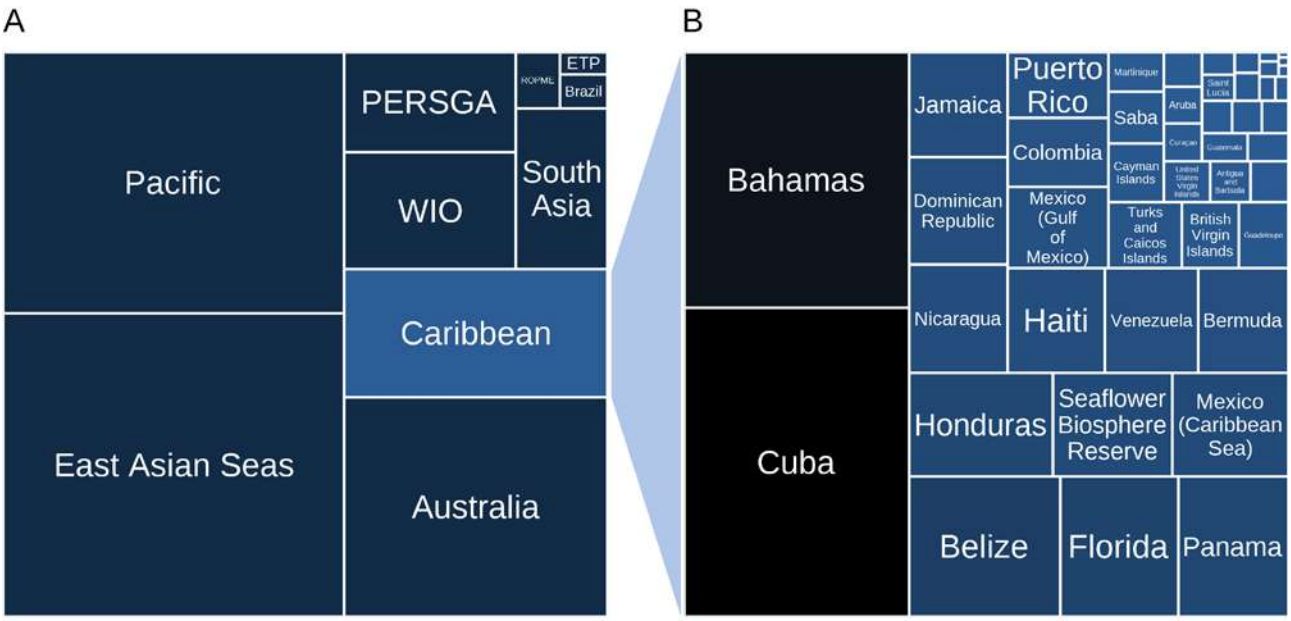
1.1. DISTRIBUTION AND EXTENT

The GCRMN-Caribbean region (hereafter the Caribbean) ranges from Mexico to Trinidad and Tobago on a longitudinal axis over 4,000 km, and from Bermuda to Panama on a latitudinal axis over 2,600 km (**Figure 1.1.1**). After the East Asian Seas, Pacific, and Australia GCRMN regions, the Caribbean has the fourth highest coral reef extent of the 10 GCRMN regions (**Figure 1.1.2.A**). Its 24,230 km² of coral reefs, which represent 9.7% of the world's coral reef extent, are hosted within 44 countries and territories (**Figure 1.1.1**).

While most countries in the region form a contiguous territory, several nations spread across several non-contiguous territories, including the United Kingdom (U.K.), the United States of America (U.S.A.), France, and the Netherlands (**Figure 1.1.1**). The U.K. overseas territories in the Caribbean comprise Anguilla, Bermuda, the British Virgin Islands, the Cayman Islands, Montserrat, and the Turks and Caicos Islands. U.S. Caribbean States and territories include Florida, Flower Garden Banks National Marine Sanctuary, the U.S. Virgin Islands, and Puerto Rico. France's overseas territories in the Caribbean consist of the Guadeloupe archipelago (Guadeloupe island, La Désirade, Marie-Galante, Les Saintes), Martinique, Saint-Martin, and Saint-Barthélemy. The Dutch Caribbean territories include Aruba, Bonaire, Curaçao, Saba, Sint Eustatius, and Sint Maarten. Colombia's Seaflower Biosphere Reserve, though part of the country, is geographically separate from its mainland reef systems. Mexico's coastline is influenced by two distinct bodies of water; the Gulf of Mexico, and the Caribbean Sea, so they are considered as two different territories.



↑ **Figure 1.1.1.** Map of countries and territories and distribution of coral reefs in the GCRMN Caribbean region. A. = Aruba, B. = Bonaire, BVI = British Virgin Islands, Be. = Belize, C. = Curaçao, CS = Caribbean Sea, GoM = Gulf of Mexico, Isl. = Islands, USVI = United States Virgin Islands. To facilitate interpretation, coral reef distribution is represented with a 5 km buffer. The borders and nomenclature represented on this map are not endorsed by the GCRMN and the co-authors of the report.



↑ **Figure 1.1.2.** Relative coral reef extent between GCRMN regions (A) and between countries and territories within the GCRMN Caribbean region (B). The area of each polygon is proportional to the relative coral reef extent of the associated region or country/territory. The absolute values of coral reef extent for countries and territories of the Caribbean are provided in **Table 1.1.1.** In subfigure A, ETP is the acronym for Eastern Tropical Pacific, and WIO for Western Indian Ocean. PERSGA (Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden) and ROPME (Regional Organization for the Protection of the Marine Environment) regions correspond to the Red Sea and Gulf of Aden, and the Arabian/Persian Gulf, respectively.

Coral reef extent varies greatly between countries and territories of the Caribbean. Cuba and The Bahamas host the largest coral reef areas in the region (4,931 and 4,081 km², respectively), accounting for 20.4% and 16.8% of the total Caribbean coral reef area, respectively (**Table 1.1.1**). These two countries, along with Belize, Panama, and the state of Florida (U.S.A) represent more than half of the region's coral reef extent, while Sint Maarten hosts the smallest coral reef area (~0.015% of the regional reef area).

↓ **Table 1.1.1.** Comparison of coral reef extent indicators between countries and territories of the GCRMN-Caribbean region. For countries and territories with coral reefs in both the Caribbean and Eastern Tropical Pacific regions (*e.g.,* Panama), only the extent of coral reefs located in the Caribbean region is provided. More precise estimates are sometimes available for certain countries and territories presented in the second part of the report. However, for reasons of methodological consistency, we report here the values obtained using the same approach. Refer to the **Materials and Methods** section for technical details regarding the estimation of these values.

Countries and territories	Absolute extent (km²)	Extent relative to the Caribbean (%)	Extent relative to the world (%)
Anguilla	52	0.216	0.021
Antigua and Barbuda	116	0.480	0.047
Aruba	99	0.410	0.040
The Bahamas	4,081	16.843	1.634
Barbados	62	0.256	0.025
Belize	1,506	6.215	0.603
Bermuda	672	2.772	0.269
Bonaire	68	0.282	0.027
British Virgin Islands	266	1.100	0.107
Cayman Islands	231	0.953	0.093
Colombia	494	2.038	0.198
Costa Rica	21	0.086	0.008
Cuba	4,931	20.351	1.975
Curaçao	103	0.424	0.041
Dominica	49	0.202	0.020
Dominican Republic	750	3.097	0.301
Florida (U.S.A)	1,175	4.850	0.471
Flower Garden Banks National Marine Sanctuary (U.S.A)	6	0.024	0.002
Grenada	110	0.455	0.044
Guadeloupe	233	0.962	0.093
Guatemala	92	0.378	0.037
Haiti	724	2.990	0.290
Honduras	1,060	4.376	0.425
Jamaica	734	3.029	0.294
Martinique	155	0.640	0.062
Mexico (Caribbean Sea)	858	3.541	0.344
Mexico (Gulf of Mexico)	582	2.403	0.233
Montserrat	8	0.032	0.003
Nicaragua	758	3.129	0.304
Panama	1,097	4.527	0.439
Puerto Rico	471	1.945	0.189

↓ **Table 1.1.1.** (*continuing*)

Countries and territories	Absolute extent (km²)	Extent relative to the Caribbean (%)	Extent relative to the world (%)
Saba	203	0.840	0.081
Saint Kitts and Nevis	82	0.338	0.033
Saint Lucia	61	0.250	0.024
Saint Vincent and the Grenadines	93	0.385	0.037
Saint-Barthélemy	27	0.113	0.011
Saint-Martin	13	0.053	0.005
Seaflower Biosphere Reserve	880	3.632	0.352
Sint-Eustatius	23	0.094	0.009
Sint-Maarten	4	0.015	0.001
Trinidad and Tobago	70	0.289	0.028
Turks and Caicos Islands	343	1.415	0.137
United States Virgin Islands	134	0.554	0.054
Venezuela	696	2.870	0.279
Entire Caribbean region	**24,230**	**100**	**9.703**

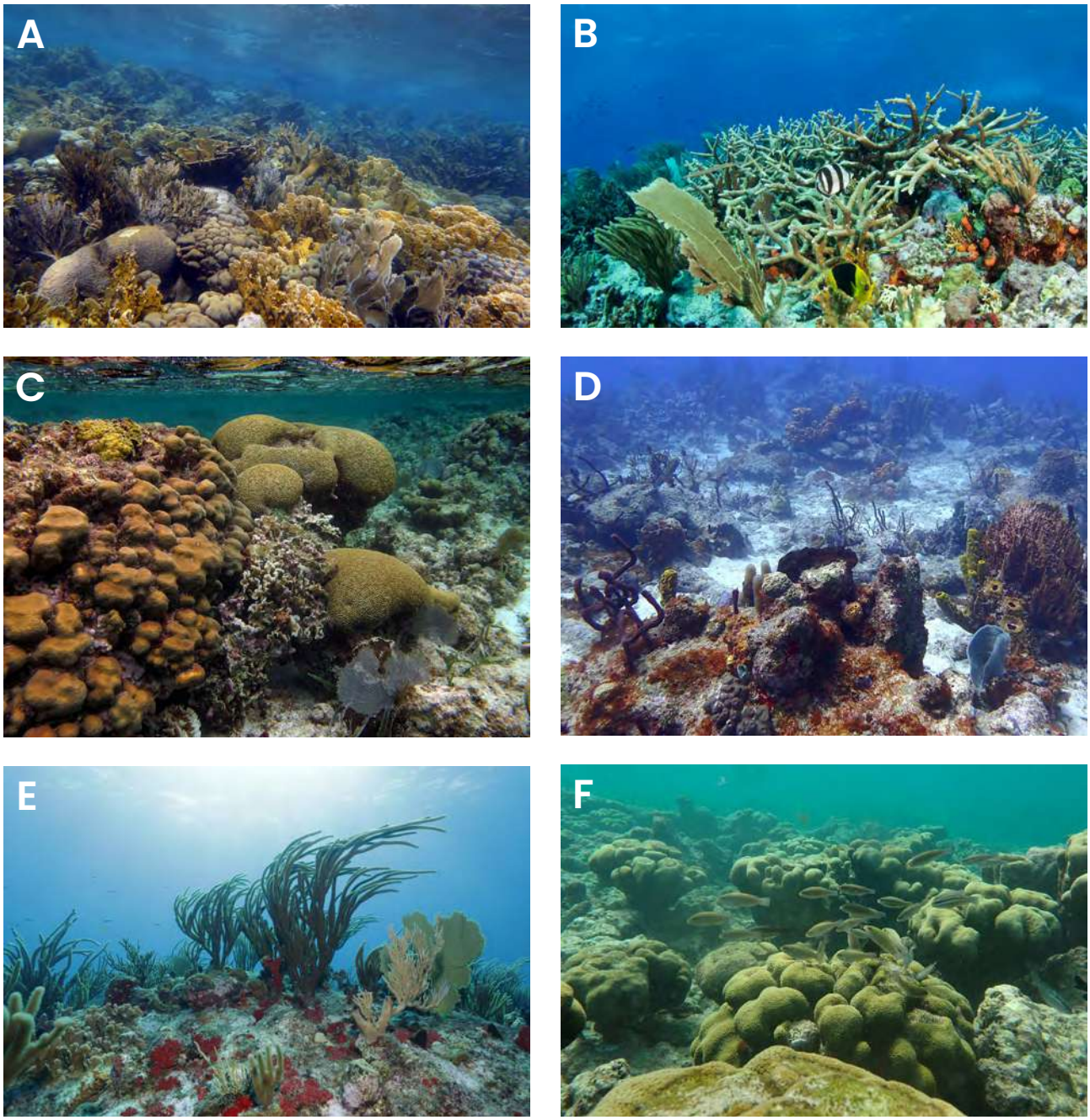
1.2. BIODIVERSITY

The Caribbean's geology and geography have strongly shaped the formation and distribution of its coral reefs. The region encompasses over 700 islands formed by volcanic activity, limestone deposition, and sea-level fluctuations (Cambers, 2018; Ortega-Ariza et al., 2021). The Bahamas, for instance, consists of carbonate platforms built from limestone over millions of years, creating extensive shallow environments favorable for reef growth. In contrast, volcanic islands like Dominica and St. Lucia provide hard substrates for coral colonization (Johnson and Kauffman, 2001). Unique cases include Flower Garden Banks in the Gulf of Mexico, where reefs thrive on salt domes rising from ~100 m depths to within 17 m of the surface (Scanlon et al., 2005). Coral reef distribution is further shaped by ocean temperature, wave energy, currents, and nutrient availability (Chollett et al., 2012a).

Regional variations drive differences in reef type, health, and biodiversity. The Southern Caribbean Upwelling System brings cool, nutrient-rich waters that fuel productivity and marine diversity (Rueda-Roa and Muller-Karger, 2013), whereas nutrient-poor but stable warm waters in The Bahamas support extensive, biodiverse reefs that are however highly vulnerable to bleaching. Currents aid larval dispersal and genetic connectivity (McManus et al., 2021), while wave energy influences resilience—moderate action boosts nutrient exchange, but hurricanes often cause destructive damage (Roberts et al., 1992).

Over 12,000 marine species are recorded in the Caribbean, with diversity hotspots in the Antillean Arc, Venezuelan coast, and the Mesoamerican Reef (Miloslavitch et al., 2010). Reefs occur in diverse forms such as fringing, patch, barrier, atoll, and bank barrier systems, each shaped by geomorphology, depth, and disturbance regimes (Medina-Valmaseda et al., 2022). Fringing and patch reefs, such as those near the Virgin Islands and Tobago Cays, support high coral and fish diversity (de Bakker et al., 2019), while barrier systems like the Mesoamerican Reef host over 70 hard coral, 36 soft coral, and 500 fish species, despite live coral cover often falling below 30% (Gibson, 2011). Biodiversity-partitioning studies reveal that regional gamma diversity (the total diversity across a landscape) is largely maintained by spatial beta diversity (differences in species composition among reef zones such as slopes vs. terraces), even as local alpha diversity (local species richness)

declines due to habitat degradation (Miyazawa et al., 2020). Structurally complex shallow zones experience intense bioerosion, sediment variation, and herbivore activity, which shape fine-scale community patterns (Reyes-Bonilla and Jordan-Dahlgren, 2017). Limited redundancy within functional groups heightens vulnerability, especially in Cuba and the Lesser Antilles, where species loss could undermine ecosystem processes. Overall, Caribbean reef biodiversity is marked by strong spatial heterogeneity, with reef type, health, and local pressures driving variation in richness, function, and resilience.



↑ **Figure 1.1.3.** Coral reefs in the Caribbean. Shallow reef community in Caracasbaai Peninsula in Curacao in 2020 (A, credit: M. Vermeij); Reef dominated by *Acropora cervicornis* in Saint-Barthélemy (B, credit: Agence Territoriale de l'Environnement); Reef in Mahahual, Mexico, in 2011 (C, credit: H. Bahena/ECOSUR); Guadeloupe, 2025 (D, credit: H. Bidenbach); Sint Maarten, 2024 (E, credit: H. Bidenbach); Reef dominated by *Orbicella annularis* in Union Island, Grenadines in 2024 (F, credit: S. Little).

1.3. ECOSYSTEM SERVICES

The Caribbean region has experienced a significant increase in population density of residents and visitors near coral reefs over recent decades (Sing Wong et al., 2022). The number of inhabitants living close to coral reefs is frequently used as a proxy for cumulative direct human stressors on these ecosystems (Andrello et al., 2022; Cinner et al., 2018). However, it can also be seen as a proxy for the dependence of human populations on coral reefs (Sing Wong et al., 2022). Indeed, coral reefs provide numerous ecosystem services to human populations. These services are defined as benefits that the human population derive from the presence of coral reefs, and are typically divided into provisioning, regulating, and cultural services (Woodhead et al., 2019).

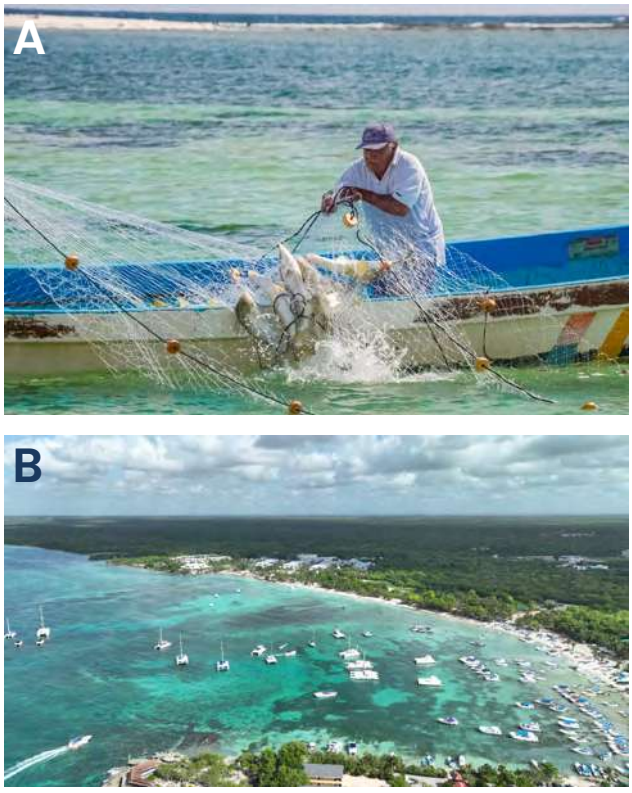
Coral reefs in the Caribbean provide vital provisioning ecosystem services that underpin food security, livelihoods, and economic resilience for millions of coastal residents. These reefs support diverse fisheries (*e.g.*, **Figure 1.1.4**)—including species like snappers, groupers, and spiny lobsters—that are essential sources of protein and micronutrients (such as vitamin B12, iron, and zinc), particularly in small island developing states where fish can account for up to 90% of dietary animal protein. In areas like the Seaflower Biosphere Reserve, reef-associated fisheries supply the primary protein source for local communities and support artisanal fishers who rely on traditional methods for their livelihoods (Prato et al., 2024).

Essential regulating ecosystem services are also provided by coral reefs in the Caribbean, by acting as natural barriers that mitigate the impacts of wave energy and storm surges, thereby protecting coastal communities and ecosystems. The structural complexity of healthy coral reefs plays a crucial role in dissipating wave energy. Studies have shown that reefs can reduce wave energy by an average of 97%, with the reef crest alone accounting for 86% of this attenuation (Ferrario et al., 2014; Carlot et al., 2023). This natural protection is especially significant for low-lying Caribbean islands, where coral reefs help prevent coastal erosion and flooding (Reguero et al., 2021).

Cultural ecosystem services of coral reefs, particularly through tourism, are a significant contributor to the region's economy. Coral reefs, along with mangroves and other coastal ecosystems in the Caribbean, provide an estimated value of nearly $15 billion annually in fisheries, tourism recreation and carbon sequestration, while coral reefs alone provide nearly $6.2 billion in benefits (Resilient Islands, 2019). Reef-associated tourism—including activities like snorkeling, scuba diving, and beach recreation—accounts for 23% of all tourism spending and more than 10% of the Caribbean's gross domestic product (Spalding et al., 2018). In countries such as Antigua and Barbuda, Bermuda, and Saint-Martin, reef-based tourism constitutes a substantial portion of national income, offering employment opportunities in sectors like hospitality, guiding services, and transportation (Pellerin et al., 2025).

Caribbean coral reefs are also a source of bioactive compounds with pharmaceutical potential. Marine organisms such as sponges, corals, and mollusks produce unique chemicals that have shown promise in treating diseases like cancer and infections (Phang et al., 2023).

However, the continued provision of these vital ecosystem services by Caribbean coral reefs is increasingly threatened by a combination of human-induced pressures and environmental threats, which contribute to reef degradation and reduced ecological functionality.



↑ **Figure 1.1.4.** Fisherman in the lagoon of Punta Herrero, Quintana Roo, Mexico in 2011. In the background, the role of protection of the barrier reef is visible, (A, credit: H. Bahena/ECOSUR). Boat excursions in the Bay of Bayahibe, Dominican Republic in 2025 (B, credit: Michael Del Rosario/FUNDEMAR).

2. Threats to coral reefs of the Caribbean

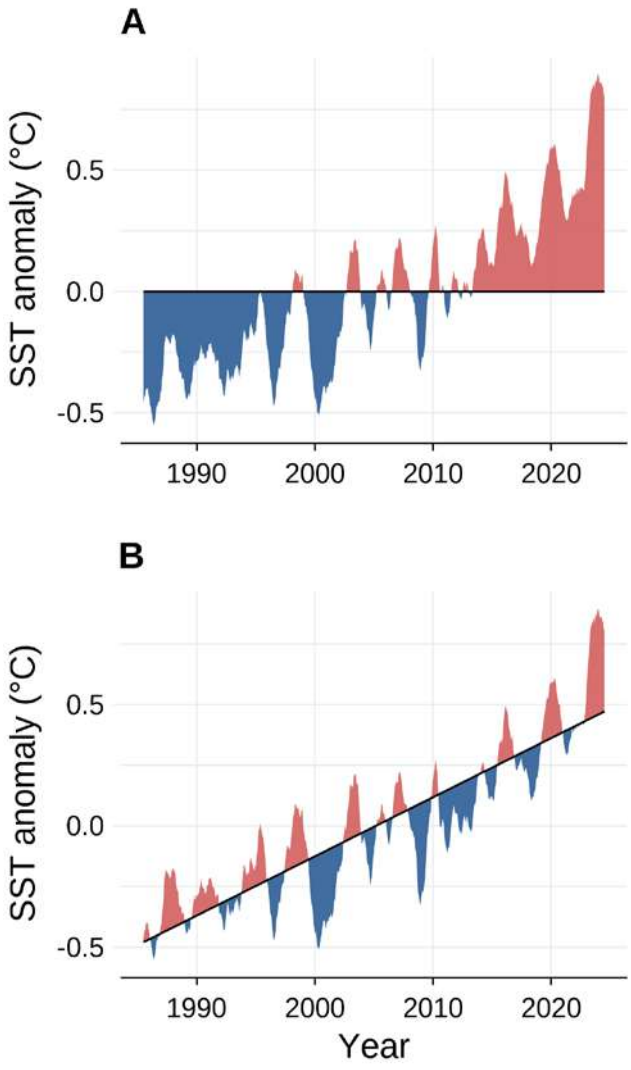
2.1. CLIMATE CHANGE AND CO_2

The concentration of greenhouse gases in the atmosphere has risen significantly over the past century, with a marked acceleration in recent decades due to human activities. Notably, since 1750, atmospheric levels of carbon dioxide (CO2), methane (CH4), and nitrous oxide (N2O) have increased by approximately 47%, 156%, and 23%, respectively (IPCC, 2023). These increases are primarily attributed to the burning of fossil fuels, industrial processes, and land-use changes (IPCC, 2023). The accumulation of greenhouse gases has caused an energy imbalance in the Earth's climate system by reducing the amount of energy lost to space through reflected solar radiation and thermal radiation (IPCC, 2021). This imbalance drives climate change, which has already resulted in a global surface temperature rise between 1.34 and 1.41°C in 2024 relative to the 1850–1900 baseline (WMO, 2025).

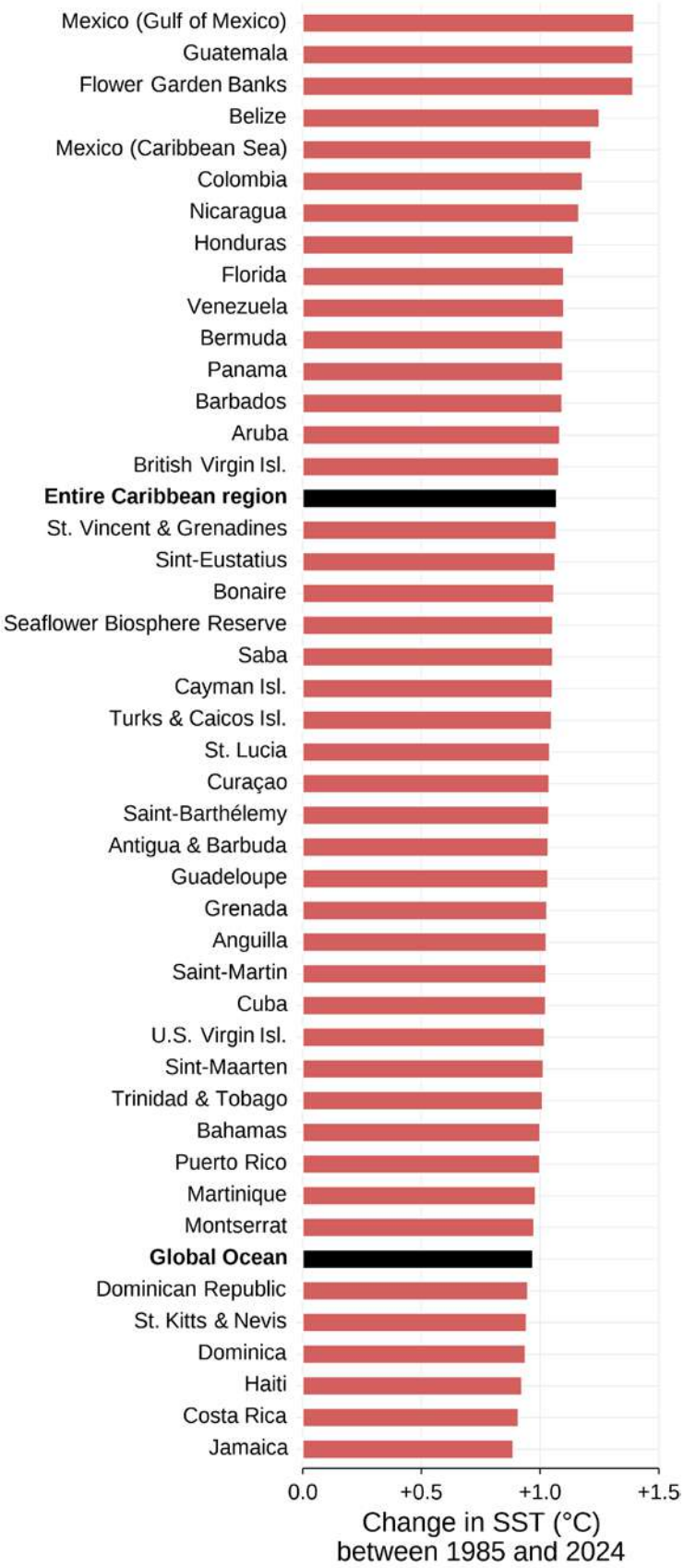
2.1.1. Sea Surface Temperature (SST)

2.1.1.1. Long-term trends in SST

Between 1985 and 2024, we estimated that Sea Surface Temperatures (SST) over Caribbean coral reefs have risen by an average of 1.07°C, corresponding to a warming rate of 0.27°C per decade (**Figure 1.2.1.B**). This rate is similar to the one reported by Chollet et al. (2012) for the period 1985–2009, but higher than the 0.18°C per decade found by Bove et al. (2022) for 1994–2019, and 0.17°C per decade reported by Heron et al. (2016) for 1985–2012. These differences could be attributed to differences in modeling methods to estimate the trends. According to our analysis, all Caribbean countries and territories experienced SST increases over coral reefs during the 1985–2023 period (**Figure 1.2.2**). The largest long-term SST increases were observed in Mexico (Gulf of Mexico), Guatemala, and Flower Garden Banks National Marine Sanctuary, with average rises of around 1.4°C (**Table 1.2.1**). Overall, SST of Caribbean coral reefs is warming at a faster rate than the global ocean (**Figure 1.2.2**). By extrapolating the estimated warming rate over the last decades, Bove et al. (2022) suggested that the SST over Caribbean coral reefs could warm by an additional 1.5°C by 2100. This escalation in chronic and acute heat stress is expected to further reduce the recovery windows between bleaching events, likely pushing many coral species beyond their thermal tolerance limits.



↑ **Figure 1.2.1.** Average Sea Surface Temperature (SST) anomaly from 1985 to 2024 over coral reefs of the Caribbean. Data used for the subplots A and B are the same, but the reference lines are different. In subfigure A, the black line is the long-term average in SST anomaly; values below this line are negative SST anomalies (*i.e.*, cooler than the long-term average), and values above this line are positive SST anomalies (*i.e.*, warmer than the long-term average). On subfigure B, the black line is the long-term trend in SST anomaly; values below this line are negative SST anomalies (*i.e.*, cooler than the long-term trend), and values above this line are positive SST anomalies (*i.e.*, warmer than the long-term trend).



↑ **Figure 1.2.2.** Comparison of average changes in daily SST of coral reefs between Caribbean countries and territories over the period 1985 to 2024. See **Table 1.2.1** for values. The value for the "Global Ocean" (*i.e.*, 0.97°C) was obtained from Forster et al. (2024)[1] and provided for comparison purposes. However, it must be noted that this value corresponds to the average SST change from 1850–1900 to 2014–2023 and not to the average SST change from 1985 to 2024.

[1] We preferred this value to the SST change value of 0.88°C between 1850–1900 and 2011–2020, provided by the IPCC AR 6 (IPCC, 2021; chapter 9, page 1214) because it coincided more closely with our study period.

↓ **Table 1.2.1.** Comparison of thermal regime indicators between countries and territories of the GCRMN Caribbean region. The column "Long-term SST average" corresponds to the long-term average of Sea Surface Temperature (SST) from 1985 to 2024 over coral reefs of the country or the territory. The column "Change in SST" corresponds to the change in SST from 1985 to 2024 over the coral reefs of the country or the territory. The column "Annual rate of change" corresponds to the annual rate of change in SST from 1985 to 2024, and was calculated as the long-term SST trend value divided by 39 (the number of years from 1985 to 2024). Refer to the **Materials and Methods** section for technical details regarding the estimation of these values.

Countries and territories	Long-term SST average (°C)	Change in SST (1985–2024, °C)	Annual rate of change (°C.year^{-1})
Anguilla	27.50	1.03	0.026
Antigua and Barbuda	27.49	1.03	0.027
Aruba	27.28	1.08	0.028
The Bahamas	26.91	1.00	0.026
Barbados	27.74	1.09	0.028
Belize	28.00	1.25	0.032
Bermuda	23.30	1.10	0.028
Bonaire	27.47	1.06	0.027
British Virgin Islands	27.49	1.08	0.028
Cayman Islands	28.17	1.05	0.027
Colombia	28.83	1.18	0.030
Costa Rica	28.11	0.91	0.023
Cuba	27.82	1.02	0.026
Curaçao	27.42	1.04	0.027
Dominica	27.70	0.94	0.024
Dominican Republic	27.53	0.95	0.024
Florida (U.S.A)	26.67	1.10	0.028
Flower Garden Banks National	25.27	1.39	0.036
Grenada	27.78	1.03	0.026
Guadeloupe	27.65	1.03	0.027
Guatemala	28.18	1.39	0.036
Haiti	28.18	0.92	0.024
Honduras	28.05	1.14	0.029
Jamaica	28.18	0.89	0.023
Martinique	27.71	0.98	0.025
Mexico (Caribbean Sea)	27.83	1.22	0.031
Mexico (Gulf of Mexico)	26.69	1.40	0.036
Montserrat	27.63	0.98	0.025
Nicaragua	28.09	1.16	0.030
Panama	28.16	1.10	0.028
Puerto Rico	27.72	1.00	0.026
Saba	27.73	1.05	0.027
Saint Kitts and Nevis	27.61	0.94	0.024
Saint Lucia	27.75	1.04	0.027
Saint Vincent and the Grenadines	27.79	1.07	0.027
Saint-Barthélemy	27.51	1.04	0.027
Saint-Martin	27.49	1.03	0.026
Seaflower Biosphere Reserve	27.99	1.05	0.027
Sint-Eustatius	27.76	1.06	0.027

↓ **Table 1.2.1.** (*continuing*)

Countries and territories	Long-term SST average (°C)	Change in SST (1985–2024, °C)	Annual rate of change (°C.year^{-1})
Sint-Maarten	27.52	1.01	0.026
Trinidad and Tobago	27.72	1.01	0.026
Turks and Caicos Islands	27.25	1.05	0.027
United States Virgin Islands	27.66	1.02	0.026
Venezuela	27.22	1.10	0.028
Entire Caribbean region	**27.48**	**1.07**	**0.027**

2.1.1.2. Marine heatwaves

Beyond the long-term increase in the daily average SST over coral reefs, climate change is also altering the characteristics of marine heatwaves—prolonged periods of unusually high SST lasting days to months and spanning vast areas (Frölicher et al., 2018; Laufkötter et al., 2020). These events exert substantial ecological pressure on marine ecosystems, with hard corals being among the most vulnerable (Smale et al., 2019). Prolonged heat stress can disrupt the coral–zooxanthellae symbiosis, often causing coral bleaching, where corals expel their symbionts and lose color (**Figure 1.2.2**, van Woesik et al., 2022). While factors such as depth may buffer heat stress, the loss of symbionts deprives corals of vital energy, potentially leading to starvation and death of the coral colony if new symbionts are not acquired quickly (Hughes et al., 2019). Even surviving colonies may face delayed effects, due to the disruptive effect of bleaching on the energetic balance, including reduced growth and reproduction (Levitan et al., 2014).

↑ **Figure 1.2.3.** Partially bleached coral colonies in Puerto Morelos, Mexico, in August 2023 (A, credit: V. Cucchiara); Reef experiencing partial bleaching in Oostpunt, Curaçao, in 2024 (B, credit: M. Vermeij); Reef experiencing partial bleaching in Martinique in 2023 (C, credit: Impact Mer).

In the Caribbean, major marine heatwaves were recorded in 1987, 1998, 2005, 2010, 2015–2017, 2020, and 2023–2024 (Glynn, 1993; Muñiz-Castillo et al., 2019; Cetina-Heredia and Allende-Arandía, 2023; Reimer et al., 2024). These events triggered bleaching of varying intensity, duration, and extent. For instance, the 1998 event caused near-total coral loss in two lagoonal reefs in Belize (Aronson et al., 2000), while the 2005 event led to bleaching in 25–95% of colonies region-wide and mortality up to 40% at many sites (Wilkinson and Souter, 2008; Eakin et al., 2010). Between 2015 and 2017, widespread bleaching occurred in the Mesoamerican Reef, with 2017 having the most severe effects (Muñiz-Castillo et al., 2024). Most recently, the fourth global bleaching event, which began in 2023, continues into 2025, has caused unprecedented widespread bleaching and mortality across the Caribbean, including the Cayman Islands, Martinique, the Dominican Republic, Cuba, and the Mesoamerican Reef (Reimer et al., 2024; Doherty et al., 2025; Bon et al., 2025; Croquer et al., 2024; Cobian-Rojas et al., 2024; McField et al., 2024).

The frequency and duration of marine heatwaves in the Caribbean have risen from about one per year in the early 1980s to around five per year in the 2020s (Bove et al., 2022), and are projected to increase further under continued greenhouse gas emissions (IPCC, 2021). While historically tied to El Niño-Southern Oscillation (ENSO) phases, heatwaves now appear to occur regardless of ENSO (Muñiz-Castillo et al., 2019). If these trends persist, ocean temperatures will regularly exceed coral tolerance limits, undermining the coral–zooxanthellae symbiosis and threatening the long-term survival of reef ecosystems.

2.1.2. Hurricanes

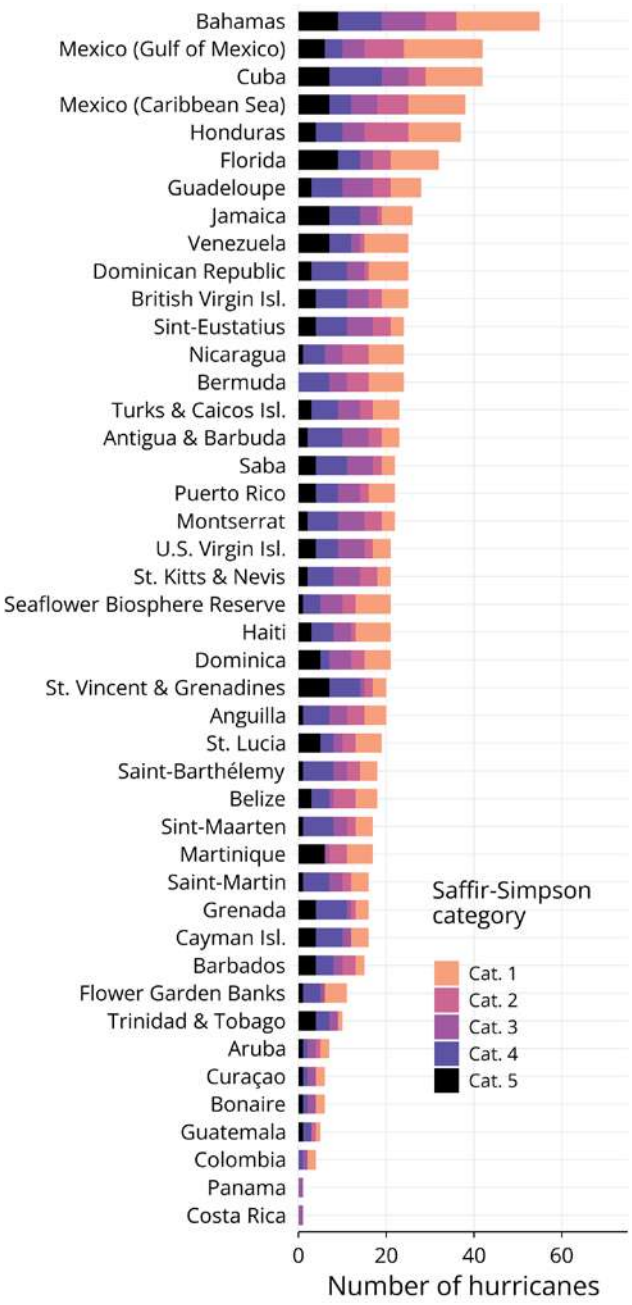
Hurricanes, also called cyclones or typhoons, are intense storms characterized by a central low-pressure system and circular winds exceeding 33 $m.s^{-1}$ (≈119 $km.h^{-1}$; Wehner and Kossin, 2024). They can generate waves over 10 m high and storm surges several meters above normal tides, leading to flooding, erosion, and infrastructure damage (Puotinen et al., 2020; Sammonds et al., 2023). Classified on the five-level Saffir-Simpson scale (see **Table 3.1** - Materials and Methods), hurricanes also bring heavy rainfall, which alters turbidity and salinity in coastal waters, with major consequences for coral reef ecosystems (Harmelin-Vivien, 1994).

The most immediate impact of hurricanes on coral reefs is physical damage to hard corals: corals may be broken, abraded, or destroyed. In Puerto Rico, Hurricanes Irma and Maria (2017) reduced live coral cover by 35% in *Acropora palmata*, 79% in *A. cervicornis*, and 96% in *Dendrogyra cylindrus* (Hernandez-Delgado et al., 2024). Branching corals are especially vulnerable due to their morphology, although acroporid fragments have shown a relatively high capacity for survival and reattachment to the substrate compared to other branching taxa (Madin et al., 2014; Kayal et al., 2015). In the event of a hurricane, there are often winners and losers, evidence following hurricanes Maria and Irma in 2017, where the proportion of upright sponges on coral reefs in the U.S. Virgin Islands declined by 37%, whilst encrusting sponges increased by 24% (Gochfeld et al., 2020). Similarly, the damages on dominant coral species create space and light for slower-growing organisms. Damage levels depend on wind speed, storm track, depth, and geomorphology, resulting in highly uneven spatial impacts. With the long-term decline of branching corals, modern hurricanes cause proportionally less total cover loss than in past decades (Gardner et al., 2005; Edmunds et al., 2019). However, impacts go beyond physical destruction—hurricanes may accelerate the spread of coral diseases (Dobbelaere et al., 2024) or invasive species (Johnston and Purkis, 2015). Conversely, they can temporarily reduce heat stress. Hurricane Wilma (2005) cooled sea surface temperatures near Florida reefs by 3°C, helping mitigate bleaching (Carrigan and Puotinen, 2014; Manzello et al., 2007).

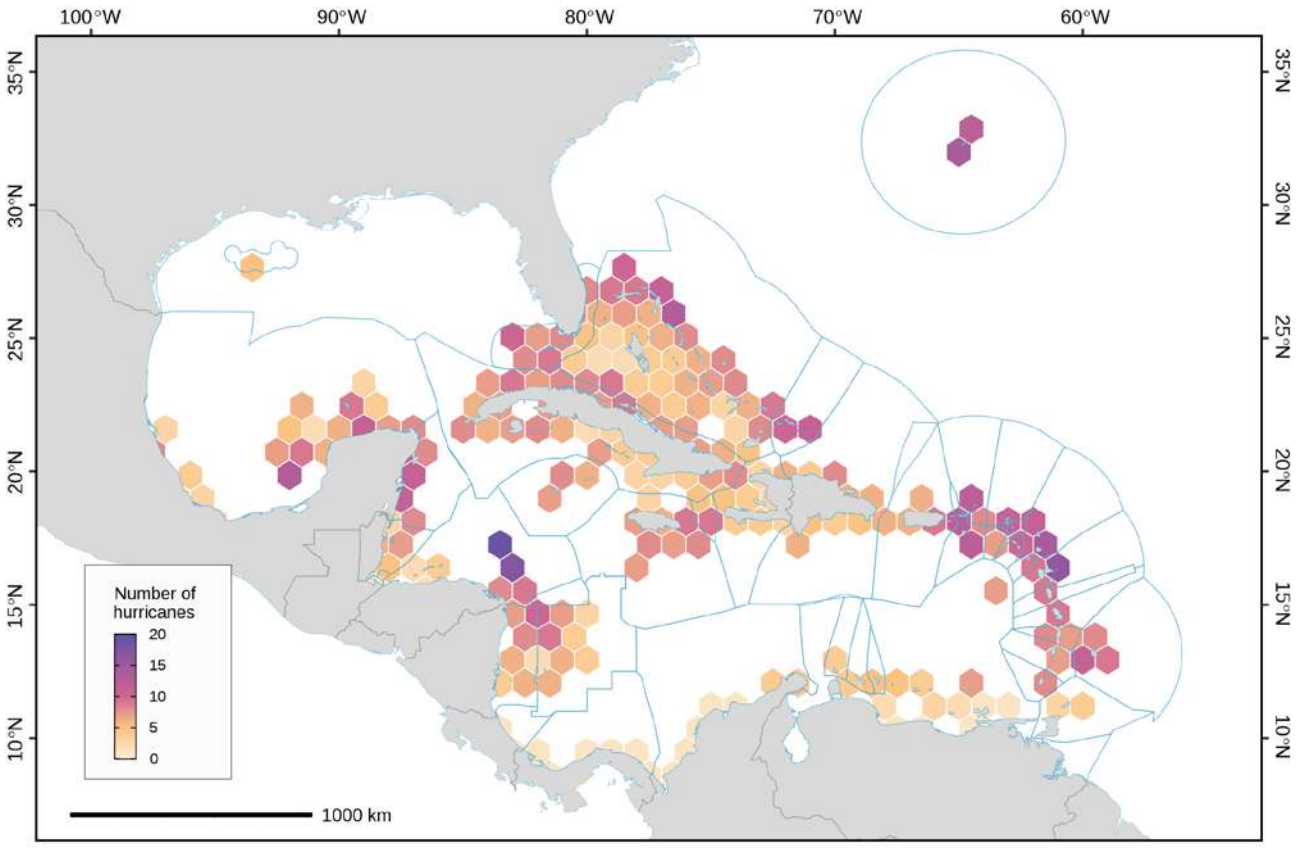
Between 1980 and 2024, 171 hurricanes were estimated to have passed within 100 km of Caribbean coral reefs, averaging 3.8 (±2.5) per year. Over this period, peak years included 1995 (11 hurricanes), 2005 (15), and 2017 (10) (Camp et al., 2018). As expected, the countries and territories with the most extensive coral reef extent (*e.g.*, The Bahamas, Cuba) or maritime areas (*e.g.*, Venezuela, Gulf of Mexico) intercepted the highest number of hurricanes (**Figure 1.2.4**). However, smaller territories like Guadeloupe were also frequently affected due to common hurricane tracks across the Caribbean arc and northern Caribbean Sea. In contrast, reefs in the southern Caribbean were rarely impacted (**Figure 1.2.5**). For instance, from 1980–2024 The Bahamas recorded 54 hurricanes, while Costa Rica and Panama recorded just one (**Figure 1.2.4**). The most intense storms near reefs were Hurricane Allen, which struck Mexico in 1980

with sustained wind speeds of 305 km.h^{-1}, and Hurricane Irma, impacting the northern Lesser Antilles in 2017 with sustained wind speeds of 287 km.h^{-1}.

Hurricane patterns have shifted with climate change and are projected to continue changing (IPCC, 2021). While overall frequency in the Atlantic may decline, storm intensity is expected to increase over the 21st century (Elsner et al., 2008; Knutson et al., 2010). Tracks have gradually shifted northward, a trend likely to persist (Baldini et al., 2016; Kossin et al., 2014; Studholme et al., 2022). The season remains May–November, peaking in September, but intense hurricanes are now occurring earlier, by ~3.7 days per decade since 1981 (Sobel et al., 2021; Shan et al., 2023). These changes are likely to have consequences for Caribbean reefs, which will face stronger storms with greater structural damage and reduced recovery potential, threatening reef resilience and the ecosystem services they provide (Cheal et al., 2017; Puotinen et al., 2020).



→ **Figure 1.2.4.** Comparison of the number of hurricanes that passed within 100 km of a coral reef between countries and territories of the Caribbean from 1980 to 2024. A single hurricane may be counted in multiple countries or territories if its track passed near coral reefs in more than one location.



↑ **Figure 1.2.5.** Occurrence of hurricanes that passed within 100 km of a coral reef of the Caribbean between 1980 and 2024.

2.1.3. Ocean acidification

Since the Industrial Revolution, over 500 billion metric tons of CO_2 have been released in the atmosphere from anthropogenic sources like fossil-fuel burning, land-use change, and industry (Boden et al., 2010). Oceans have absorbed about a third of this excess CO_2, slowing global warming but driving ocean acidification (Findlay and Turley, 2021). Dissolved CO_2 reacts with seawater molecules (H_2O) to form carbonic acid (H_2CO_3), which breaks down into hydrogen ions (H^+) and bicarbonate (HCO_3^-). This process increases seawater acidity and reduces carbonate ions (CO_3^{2-}), essential for buffering acidity, and has already lowered global average ocean pH by ~0.1 since pre-industrial times (Feely et al., 2009; Caldeira and Wickett, 2003).

A major consequence is reduced calcification in reef-building organisms. Corals and coralline algae depend on calcium (Ca^{2+}) and carbonate ions to build calcium carbonate ($CaCO_3$) skeletons. As carbonate availability falls, calcification slows while bioerosion and dissolution increase, shifting reefs from net growth to net erosion. Experiments show calcification declines of 5–60% under CO_2 concentration twice as high as pre-industrial levels (Feely et al., 2009), with Caribbean corals such as *Siderastrea siderea* and *Porites astreoides* showing steep nonlinear declines under combined warming and acidification (Bove et al., 2019; Cardoso et al., 2025). Similarly, crustose coralline algae, critical for coral settlement, experience growth declines of up to 40% and recruitment losses exceeding 70% (Guinotte and Fabry, 2008), further hindering reef recovery (Hill and Hoogenboom, 2022).

The current anthropogenic trend in ocean acidification already exceeds the level of natural variability by up to 30 times on regional scales (Friedrich et al., 2012). In the Caribbean, aragonite saturation state (Ωarg) has already dropped ~3% relative to pre-industrial levels, with declines continuing at ~-0.012 yr^{-1} (Gledhill et al., 2008). This trend, combined with local stressors such as pollution and runoff, further reduces calcification potential. Studies indicate that under projected end-of-century CO_2, key species like Acropora palmata and Porites astreoides may see fertilization, settlement, and growth rates fall by up to 70% (Albright et al., 2010; Albright and Langdon, 2011). The combined effects could leave Caribbean reefs structurally weaker and less able to protect coastlines from storms and erosion (Melendez and Salisbury, 2017).

Under scenarios of unmitigated CO2 emissions, atmospheric CO_2 could increase to ~800 ppm by 2100, leading to pH declines of up to 0.4 and carbonate ion reductions of nearly 50% (Findlay and Turley, 2021). By then, most coral reefs worldwide are expected to fall into net carbonate loss under scenarios RCP4.5 and 8.5 of increasing ocean acidification (Cornwall et al., 2021). In the Caribbean, pH is projected to decline steadily after 2040, with a ~0.2 unit reduction by 2100 (Lawman et al., 2022). These rapid changes in ocean chemistry—unmatched over millions of years—pose severe risks to reef survival, making it crucial to understand how corals respond to multiple stressors like warming and acidification in order to refine projections of reef futures.

2.1.4. Deoxygenation

Ocean deoxygenation, the decline of oxygen in marine waters, is driven mainly by global warming. Rising sea surface temperatures reduce oxygen solubility, increase stratification, and raise metabolic oxygen demand, explaining about 50% of oxygen loss in the upper 1,000 m of the ocean (Hughes et al., 2020). Additional drivers include nutrient runoff and atmospheric deposition, which cause eutrophication and algal blooms, fueling microbial respiration and creating coastal hypoxic "dead zones" (Levin, 2017). Over the past century, oxygen-limited zones have expanded across the Atlantic, with hypoxic (<2–3 mg/L O_2) and even anoxic (<0.5 mg/L O_2) waters now widespread in coastal areas and mid-water depths.

Deoxygenation disrupts ecosystems by reducing growth, reproduction, recruitment, and biodiversity, while increasing disease vulnerability and mortality (Laffoley and Baxter, 2019). Deoxygenation also directly impacts reef-building organisms like hard corals, altering habitat structure. In Panama in 2017, an acute event caused alteration of the benthic community composition, mass bleaching and mortality, reducing live coral cover by 50%, with no recovery after a year (Johnson et al., 2021a). Experiments show that prolonged deoxygenation greatly reduces larval survival in Caribbean corals, though species vary in sensitivity, with some displaying metabolic plasticity that provides limited tolerance (Mallon et al., 2025; Johnson et al., 2021b; Pontes et al., 2023).

Globally, ocean oxygen concentration has declined by ~2% over the past 60 years, with strong decreases recorded in the Gulf of Mexico and Caribbean Sea (-0.58 µmol kg^{-1} yr^{-1} since 1980) (Maishal, 2024). Broader losses have been observed in tropical and subtropical waters of the Pacific, Atlantic, and Indian Oceans, with trends persisting into recent decades (Bhuiyan et al., 2024).

Models project an additional 1–7% global oxygen decline by 2100 under high emissions (RCP8.5), concentrated in the upper 1,000 m of the water column, where biodiversity is highest (Laffoley and Baxter, 2019). Reduced ocean circulation may explain most deoxygenation in the deep ocean (>1,000 m). While the dynamics of coral reefs' response to periodic or persistent deoxygenation are difficult to predict, projections suggest that by 2100 ~94% of reefs will face weak to moderate hypoxia and ~31% severe hypoxia under SSP5-8.5 (Pezner et al., 2023). These changes will intensify ecosystem stress, reducing coral and fisheries productivity. Integrating climate dynamics, biogeochemistry, and physiology is essential for predicting the combined impacts of deoxygenation and coastal eutrophication on future marine ecosystems (Deutsch et al., 2023).

2.1.5. *Sargassum* accumulation

Brown algae of the genus *Sargassum* form dense floating mats that drift across the ocean's surface, creating complex habitats for a wide variety of marine species. These holopelagic algae, long present in the North Atlantic and the Gulf of Mexico, are vital ecological structures providing breeding, nursery, feeding, and sheltering areas for juvenile sea turtles, commercially important fish species, invertebrates, seabirds, and dolphins (Laffoley et al., 2011; Alleyne et al., 2023). However, since 2011, the massive and recurrent blooms of *Sargassum* in the Tropical North Atlantic have become a growing environmental and socio-economic concern. Once restricted to the North Atlantic Subtropical Gyre, these blooms now extend across tropical regions, exposing ecosystems and coastal populations to serious vulnerabilities related to water quality, beach access, and mortality of aquatic organisms (Rodriguez-Martinez et al., 2025).

When *Sargassum* accumulates on beaches, it decomposes and releases toxic gases such as hydrogen sulfide (H_2S), methane, and ammonia. They reduce light penetration, increase local water temperature, leading to acidification and oxygen depletion in the surrounding water (Jouanno et al., 2025a; Rodriguez-Martinez et al., 2025).

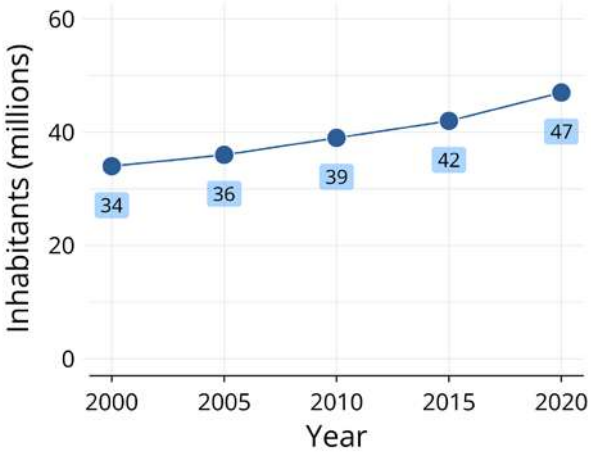
These anoxic conditions can cause mass die-offs of marine life and degradation of critical habitats like seagrass beds, coral reefs, and mangroves. In Mexico and Puerto Rico, seagrass die-offs have been directly linked to *Sargassum* accumulation (Black et al., 2025), while in Martinique, extensive mangrove mortality has been observed under similar conditions. The leachates from decomposing algae, rich in arsenic and other heavy metals, further contaminate soils, sediments, and groundwater, disrupt the successful settlement of coral larvae, hindering their ability to reproduce and grow, while posing threat for human health (Civallero et al., 2021; UNEP, 2021). In addition to ecological damage, these events have significant socio-economic repercussions, particularly for tourism and fisheries-dependent communities.

Several hypotheses have been proposed to explain the recent proliferation of *Sargassum*. These include increased nutrient input to the Atlantic Ocean from continental and atmospheric sources, and the long-distance transport of a seed population during the 2009/2010 North Atlantic Oscillation (NAO) event, which, combined with seasonal vertical mixing, enriched surface waters with nutrients and triggered large-scale blooms (Jouanno et al., 2025b). This regime shift in 2010, due to climate change, is thought to have initiated the recurrent blooms observed since 2011. Major brown tide events occurred in 2014, 2015, 2018, 2019, and most recently in 2023, prompting several Caribbean nations to declare national emergencies. It is estimated that roughly 10% of the offshore biomass—between 2 and 10 million tons annually over the past five years—eventually washes ashore (Jouanno et al., 2025a).

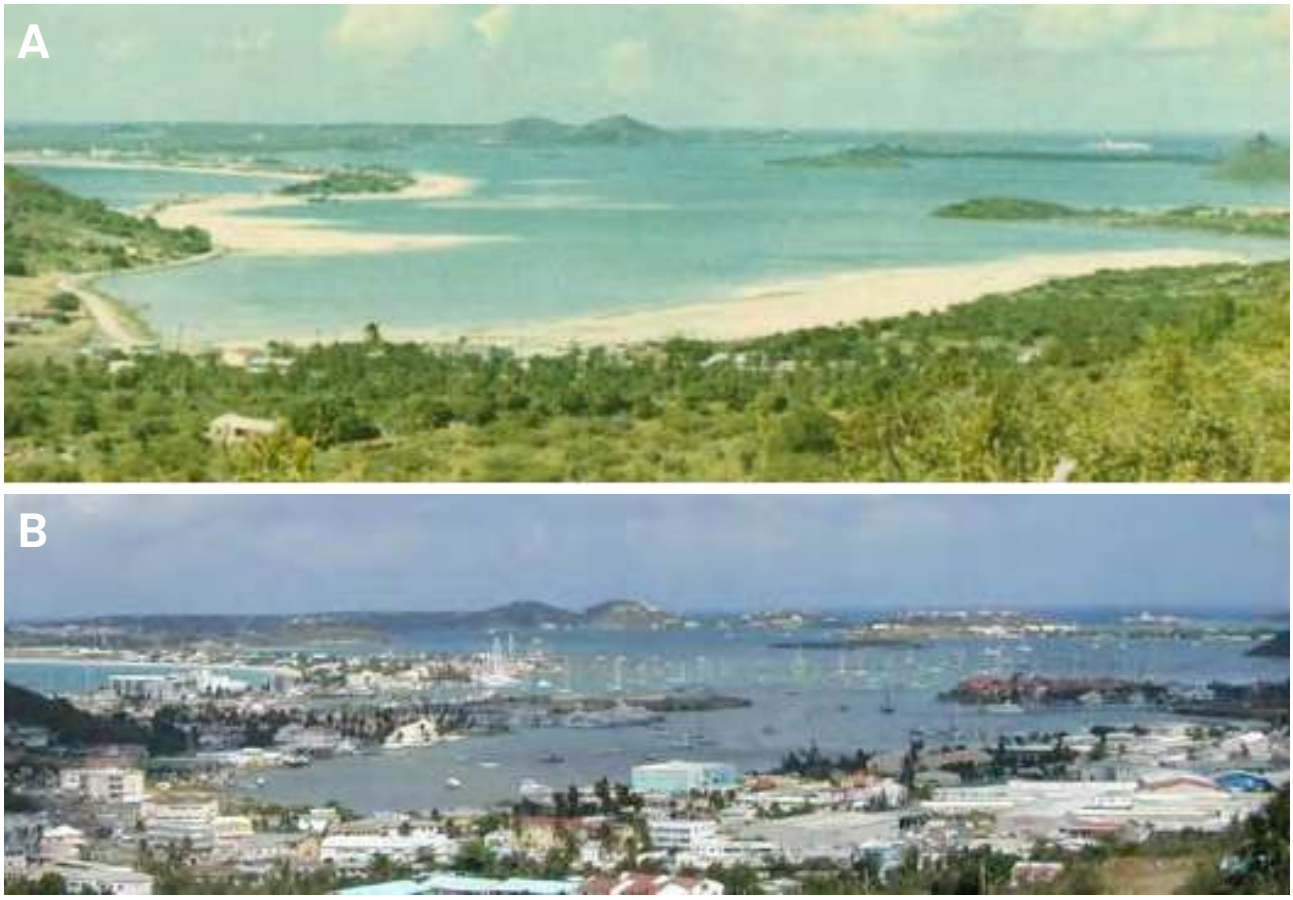
In response to this ongoing phenomenon, significant scientific, governmental, and private efforts have been mobilized to improve detection, mapping, monitoring, and forecasting of *Sargassum* influxes in the Caribbean region (UNEP, 2021). Remote sensing technologies—such as satellite imagery, drones, and automated cameras—now provide essential data for regional early warning systems (Marsh et al., 2023). Several organizations, including the GCFI, SPAW-RAC, DCNA, UNAM, UWI-CERMES, and ADEME, have developed best-practice guidelines for *Sargassum* collection, management, and utilization, aiming to minimize its environmental and economic impact (Irvine et al., 2023). Forecasts suggest that recurring blooms will likely persist in the coming years, underscoring the need for coordinated scientific research, international policy collaboration, and adaptive coastal management strategies across the Caribbean and Atlantic basins. Finally, a number of private sector enterprises, often supported by blended-finance programs like the Global Fund for Coral Reefs, are now developing commercial products from *Sargassum*.

2.2. COASTAL DEVELOPMENT AND LAND USE

Coral reefs across the Caribbean are facing escalating threats from a range of human-induced pressures, many of which have intensified significantly over the past few decades associated with high coastal human population, infrastructure, and industrial activity. The significant increase in coastal human population throughout the Caribbean region in the last decades led to greater pressure on coastal ecosystems. These densely populated regions play a critical role in shaping the demographic and environmental landscape of the wider Caribbean. However, there are stark differences in how populations are distributed in relation to coral reef ecosystems, which affects societal proximity and the prevalence of positive and negative interactions between humans and reefs (Pellerin et al., 2025). Many eastern Caribbean island nations—such as Barbados, Antigua and Barbuda, and Saint Lucia—have nearly 100% of their populations living within 20 km of coral reefs, while in continental countries like Mexico, Honduras, and Venezuela, only a small percentage of the population lives close to the sea and the reefs. The number of inhabitants living within 20 km of coral reefs in the GCRMN Caribbean region increased from 34 million in 2000 to 47 million in 2020, representing an increase of 13 million people, or 27.6% over two decades (**Figure 1.2.6**).



↑ **Figure 1.2.6.** Change in the human population living within 20 km of coral reefs in the GCRMN Caribbean region between 2000 and 2020.



↑ **Figure 1.2.7.** Coastal development illustrated by the Simpson Bay Lagoon, in Sint Maarten, in 1975 (A) and 2007 (B). Credits: A. Caballero.

Moreover, many Caribbean islands have pursued an all-inclusive model of tourism to develop the economy and local employment, leading to increased impacts on the coastal zone, by the construction of resorts, piers, etc. (**Figure 1.2.7**, Peterson et al., 2020; Velazquez-Ochoa, 2023). Rapid coastal development, expanding agriculture, deforestation, and inadequate wastewater treatment have led to increased sedimentation, nutrient enrichment, and pollution in coastal waters. Simultaneously, the rise in maritime traffic and expansion of port infrastructure has caused direct physical damage to reefs and heightened the risk of contamination. These compounded impacts are degrading coral health, reducing biodiversity, and jeopardizing the vital ecosystem services reefs provide—such as food security, tourism revenue, and shoreline protection.

2.2.1. Nutrients and sediments

Most rivers in the Wider Caribbean Region discharge large sediment loads that strain biodiversity and shallow coastal waters. The Mesoamerican region (Belize, Guatemala, Honduras, and Mexico's Yucatan Peninsula) alone contributes 374 million tons of sediment annually to the Caribbean Sea, while the Magdalena River in Colombia adds further inputs (Diez et al., 2019; Restrepo et al., 2016). Land use and land cover change have degraded water quality, delivering excess nutrients and sediments to coastal ecosystems (**Figure 1.2.8**, Clark et al., 2022). Coral reefs, which thrive in nutrient-poor waters, are particularly vulnerable: fertilizers, deforestation, wastewater, and river discharges introduce nitrogen and phosphorus that drive eutrophication, alter water quality, and weaken reef resilience (Carlson et al., 2019; Devlin and Brodie, 2023). The Amazon and Orinoco rivers also influence nutrient dynamics, particularly in the southern Caribbean (Chérubin and Richardson, 2007). Across the region, poor wastewater treatment, agriculture, urbanization, and deforestation are the main drivers of eutrophication (Rivas et al., 2020).

Anthropogenic land-use, such as urban development and agriculture, cause a decline in coral cover, diversity, and structural complexity. Nitrogen and phosphorus enrichment slows coral calcification, increases disease and oxidative stress, and promotes macroalgae growth, often shifting

reefs from coral to algae dominance (Zhao et al., 2021; Adam et al., 2020; Lesser, 2021). Sediments further damage reefs by smothering corals, lowering coral cover, and contribute to bleaching, reduced recruitment, habitat loss, and reduced fish diversity (Otaño-Cruz et al., 2019; Rogers and Ramos-Scharron, 2022). Coral species respond differently: *Acropora palmata* is highly sensitive to sedimentation, while *A. cervicornis*, *Pseudodiploria strigosa* and *Montastraea cavernosa* show greater tolerance and resilience (Rogers, 1983; Rushmore et al., 2021).

↑ **Figure 1.2.8.** Runoff in Cozumel in 2011. Credit: H. Bahena/ ECOSUR.

Nutrient and sediment inputs have increased sharply in recent decades due to agriculture and coastal development, emerging as one of the leading causes of water quality impairment, with tremendous losses for ecosystem services that sustain fisheries, aquaculture, recreation, and tourism (Horta et al., 2021). Fertilizer use rose by 30% between 1979–1989 in 17 Caribbean states, fueling runoff (Rawlins et al., 1998). Coral records from the Mesoamerican Reef System reveal a long-term rise in nutrient exposure, likely linked to land-use changes (Shaw et al., 2024). Sedimentation rates have doubled in places like Saint Lucia, and Colombian rivers show a 36% increase in suspended loads between 2000–2010 (Béguin, 2021; Lopez et al., 2017).

Regional water quality monitoring is urgently needed. In 2022, a collaborative initiative sampled 90 sites across four Mesoamerican countries. Results were concerning: most sites showed fair to poor water quality, and 74% contained human sewage pathogens, with 7%-17% exceeding Cartagena Convention limits (McField et al., 2024).

Future sedimentation and eutrophication remain difficult to predict, as they depend on climate change, precipitation, agriculture, and development trends. Models for Belize project runoff increases of 2.8–42% under different land-use scenarios (Martin-Arias et al., 2022). Without improved planning and management, current practices that drive nutrient enrichment are likely to persist across Latin America and the Caribbean (Drakes et al., 2017; Simpson et al., 2012).

2.2.2. Other pollution

Human activities are the main source of marine pollution, with 80% coming from land-based discharges and the rest from oil production, shipping, fishing gear, and atmospheric inputs. Coastal cities are major contributors due to inadequate wastewater treatment and waste management (Diez et al., 2019). Marine litter is accumulating in the Caribbean, both from local sources and distant regions via ocean currents. Plastics dominate, making up 88% of debris, and are found from beaches to deep reefs and even inside marine animals (Pinheiro et al., 2023; Aranda et al., 2022; Mesquita et al., 2022; Courtene-Jones et al., 2021). Globally, 4.8–12.7 million tons of plastic enter the oceans annually, and concentrations in the Caribbean can reach 200,000 pieces per km² (Diez et al., 2019). Their persistence poses long-term threats to marine ecosystems (Akhtar et al., 2022).

Chemical pollution from agricultural run-off or hydrocarbon production is also significant. Over 90% of hydrocarbon inputs come from land-based industry, with more than 100 oil refineries concentrated in the Gulf of Mexico (Diez et al., 2019). Polycyclic aromatic hydrocarbons (PAH) and heavy metals have been recorded in Caribbean MPAs (Nunes et al., 2021; Fuentes-Gandara et al., 2021).

Pollutants in coastal reef ecosystems pose a major threat to coral health, driving bleaching, disease, and biodiversity loss (Ajwang, 2024). Pesticides and herbicides disrupt photosynthesis, reproduction, and resilience of corals, while persistent chemicals such as chlordecone remain detectable decades after being banned (van Dam et al., 2011; Dromard et al., 2022). Recent surveys highlight how little is yet known: the first measurements of chemical pollutants in sediments around Bonaire, Saba, and St. Eustatius revealed alarming contamination levels, including dioxins, in some sites requiring urgent remediation. Similar patterns are likely to emerge as monitoring expands across other Caribbean islands (de Leijer et al., 2023). Marine debris, especially plastics, further weakens corals

by impairing feeding, causing oxidative stress, carrying pathogens, and increasing disease likelihood from 4% to 89%, with complex reef structures most at risk (**Figure 1.2.9**, Nama et al., 2023; Lamb et al., 2018). The common diseases of corals, white syndromes, skeletal eroding band, black-band, growth abnormalities, necrosis, and brown band, are blamed for being prevalent due to contamination by plastics (Akhtar et al., 2022; Lamb et al., 2018).

Since the industrial era, chemical accumulation in sediments and organisms has intensified. Plastic production rose from under 2 Mt in 1950 to over 390 Mt in 2021, with microplastic densities in oceans doubling each decade since 1970 (Xu et al., 2024; Goddijn-Murphy et al., 2025). Without urgent action, plastic leakage in the Caribbean could increase by 82% by 2040, further threatening reefs, fisheries, and coastal economies (Lebreton and Andrady, 2019; Mittempergher et al., 2022).

↑ **Figure 1.2.9.** Plastics in coral reefs of Xcalak, Mexico, in 2013. Credit: H. Bahena/ECOSUR.

2.2.3. Maritime traffic

Maritime transport has grown into a vital global industry, carrying over 90% of international trade (IMO, 2012). In the Caribbean, where island economies depend heavily on imports, 90–95% of goods are transported through maritime shipping (Singh and Mee, 2008). Cruise tourism, which began in the early 20th century as an activity for wealthy travelers, is now a global business and a central part of the Caribbean economy, requiring ports, terminals, and coastal infrastructure to serve increasingly large vessels (**Figure 1.2.10**, Pallis, 2015).

This expansion comes with significant ecological costs. Port construction often involves dredging, resuspension of buried contaminants and pathogens, burial, and land reclamation, all of which damage nearby reefs. For example, the creation of large shipping ports in Florida during the 20th century directly impacted more than 310 hectares of coral reef (Walker et al., 2012). Anchoring is another critical source of damage, due to the heavy anchors and the swinging of long chains: during the COVID-19 period, 28 cruise ships anchoring off Barbados caused structural harm to thousands of square meters of reef (Small and Oxenford, 2022). Even smaller vessels can be destructive—such as a recent incident in Guadeloupe, where a yacht dropping anchor destroyed part of a coral reef, underscoring how fragile these ecosystems remain. These damages can result in long-lasting impacts for the ecosystem, as seen in the U.S. Virgin Islands (Rogers and Garrison, 2001). The discharge of cruise ship's sewage close to the shore further impacts coastal ecosystems, with significant inputs of organic material and nutrients.

Accidents and pollution add further risks. Oil spills are particularly devastating: the 1986 spill off Panama released 8,000 tons of crude oil across 40 km² of mangroves, seagrass beds, and reefs, with impacts on coral cover and species composition lasting up to four years (Guzman et al., 2020). Smaller-scale incidents, like the 2024 spill from an overturned vessel near Trinidad and Tobago, are becoming more frequent, and models suggest that as much as 83% of the Caribbean Sea could be vulnerable to oil spill contamination (Singh et al., 2015). Global shipping also facilitates the spread of invasive species through ballast water and hull fouling, with at least 15 non-native marine species identified in Cuban waters linked to international trade (Costello et al., 2022; O'Brien et al., 2017).

↑ **Figure 1.2.10.** Cruise ships in Mahahual, Mexico, in 2019. Credit: H. Bahena/ECOSUR.

Maritime traffic has intensified further since the expansion of the Panama Canal in 2016, which allowed passage of Neopanamax ships carrying nearly triple the cargo of older vessels (Li et al., 2023). The cruise industry has also expanded rapidly since the 1960s, driven by major operators like Carnival, Royal Caribbean, and Norwegian cruise lines, with annual passenger numbers rising steadily before the pandemic (Honey, 2019; Coggins, 2014). COVID-19 temporarily halted both shipping and tourism, but trade has rebounded due to strong port connectivity, and many Caribbean islands have upgraded their facilities to host mega cruise ships (Ajagunna et al., 2022; Grant et al., 2024). While the cruise sector's long-term recovery remains uncertain, the overall intensification of maritime activity means continued and growing pressure on the Caribbean's already stressed coral reef ecosystems.

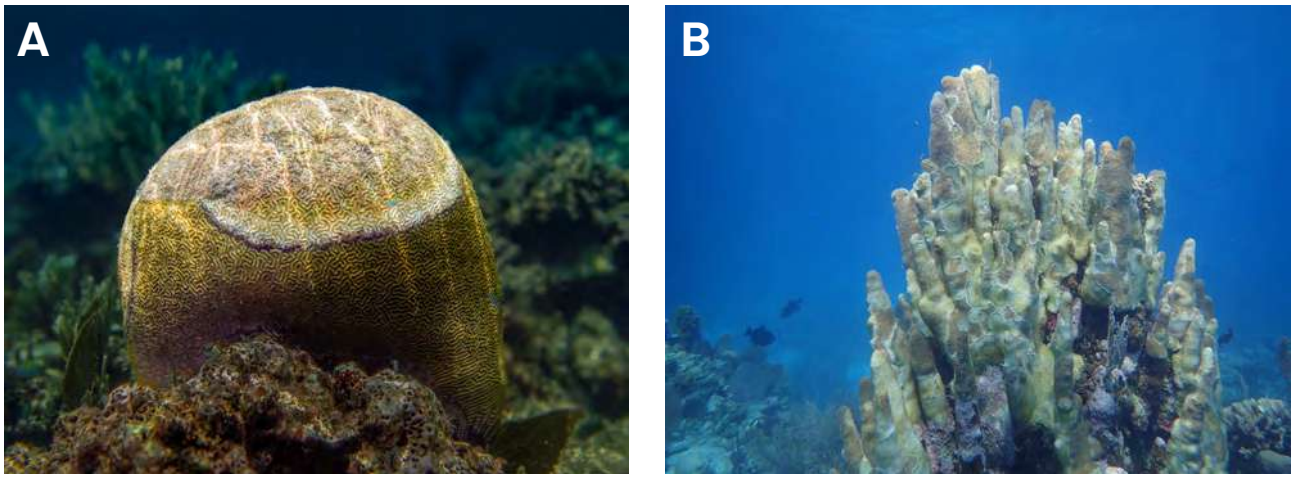
2.3. DISEASE OUTBREAKS

Caribbean coral reefs have faced increasing disease outbreaks in recent decades, driving widespread mortality among corals and reef organisms. Linked to rising sea temperatures, pollution, and other human impacts, these diseases are accelerating reef degradation and biodiversity loss. This section highlights the main diseases, their spread, and their ecological consequences.

2.3.1. SCTLD and other coral diseases

In recent decades, Caribbean scleractinian corals have suffered widespread disease outbreaks (Weil, 2004; Vega-Thurber et al., 2024). In some areas, dark-spot disease and yellow-band disease are distressing and killing coral populations to a significant degree. Dark-spot disease, first observed in the early 1990s, in the Colombian Caribbean, affects up to 16% of colonies in Colombia, especially *Siderastrea siderea* and *Orbicella* spp., with prevalence linked to warmer, shallow waters (Gil-Agudelo et al., 2004). Yellow-band disease, first reported in 1994 from corals in the lower Florida Keys, is observed in multiple species and is now common across most Caribbean reefs (Croquer and Weil, 2009). Black-band disease, driven and favored by cyanobacterial mats, is among the most widespread globally (**Figure 1.2.11**, Cissel et al., 2022). White-band disease targets branching *Acroporidae*, and leads to tissue destruction moving through coral colonies. White plague disease (WPD) spreads rapidly under warmer, turbid, low-oxygen conditions, leaving exposed skeletons (Chaves-Fonnegra et al., 2021; Vega Thurber et al., 2024). Stony Coral Tissue Loss Disease (SCTLD), first detected by the port of Miami, Florida in 2014, has since spread to at least 30 Caribbean countries, affecting over 30 coral species with extremely high mortality (Alvarez-Filip et al., 2022). SCTLD is considered the most devastating coral disease recorded, due to its overall prevalence and mortality, with even minimal infections often triggering outbreaks and threatening long-term coral recovery by affecting coral recruits (Papke et al., 2024; Croquer et al., 2022). Impacts of Type II WPD and SCTLD are similar, mainly due to the rapid tissue loss they produce in their hosts (Croquer et al., 2021).

Coral diseases in the Caribbean, by directly affecting the mortality of most sensitive species, led to a change in benthic community composition, fish assemblages and caused major



↑ **Figure 1.2.11.** Black band disease affecting *Pseudodiploria strigosa* in Xahuayxol, Quintana Roo, Mexico, in 2009 (A, credit: H. Bahena/ECOSUR); *Dendrogyra cylindrus* affected by stony coral tissue loss disease (SCTLD), in Chinchorro, Quintana Roo, Mexico, in 2023 (B, credit: E. Perez-Cervantes).

shifts from coral- to algae-dominated reefs, reducing structural complexity, biodiversity, and reef functionality. Stony coral tissue loss disease (SCTLD) is the most devastating, with prevalence reaching 16–57% in its first year at some sites and coral losses up to 62% (Papke et al., 2024; Hayes et al., 2022). Rapid declines in coral cover have been reported across the Caribbean, including in Turks and Caicos Islands, Florida, the U.S. Virgin Islands, the Mesoamerican Reef, and the Dominican Republic, with mortality rates exceeding 90% in some coral families and widespread increases in algae and cyanobacteria (Dahlgren et al., 2021; Alvarez-Filip et al., 2022; Heres et al., 2021). In the Mesoamerican Reef and Dominican Republic, the widespread coral die-off caused a significant loss of coral cover, followed by a rapid increase of algae cover across all surveyed reefs, reducing reef functionality, both in terms of functional diversity and calcium carbonate production (Alvarez-Filip et al., 2022; Estrada-Saldivar et al., 2021; Randazzo-Eisemann et al., 2022; Steneck and Torres, 2023; Croquer et al., 2024).

The Caribbean is a global hotspot for coral diseases due to their rapid emergence, virulence, and geographic spread (Green and Bruckner, 2002). Historical outbreaks of white band, dark-spot disease, yellow-band disease, and black-band disease have repeatedly affected major reef-building species. Their prevalence is strongly linked to warming sea surface temperatures and human-driven stressors (*e.g.*, pollution, maritime traffic, dredging operations), with events like Hurricane Irma accelerating long-distance transmission of pathogens (Miller and Richardson, 2014; Dobbelaere et al., 2024). These dynamics have positioned disease as a leading driver of coral decline, surpassing other disturbances in many areas (Weil, 2004).

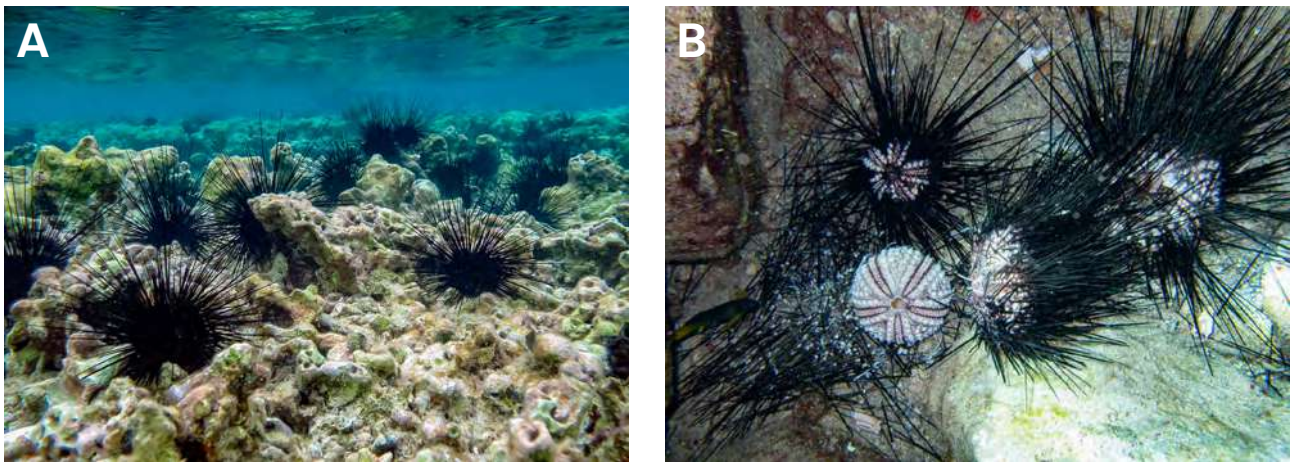
Although no large-scale cure exists, recent treatments provide hope for local mitigation. CoralCure rope treatments significantly reduced black-band disease incidence, while topical amoxicillin proved highly effective in halting SCTLD lesions. Probiotics are currently being developed for SCTLD treatment, showing promising results. Sustained application and follow-up treatments have allowed some colonies and reef sites to recover toward lower disease prevalence (Eaton et al., 2022; Neely et al., 2021). While these approaches cannot yet counteract regional-scale outbreaks, they highlight practical interventions that may enhance reef resilience and slow coral loss in particularly high-value affected reefs.

2.3.2. Urchin die-offs

Sea urchins are key herbivores on Caribbean reefs, with *Diadema antillarum* once the most abundant. *D. antillarum* is a large, black, mobile urchin whose grazing reduces turf and macroalgae while enhancing coral growth and bioerosion (Lessios, 1988; Cramer et al., 2017). Historically abundant, this benthic herbivore became especially important for maintaining coral-dominated reefs, after overfishing reduced herbivorous fish (Jackson et al., 2001; Pandolfi et al., 2003). Its populations collapsed regionally after mass die-offs in 1983–84 and subregionally in 2022 (Lessios et al., 1988; Hylkema et al., 2023).

The 1983–84 mortality event caused a 98% population decline across 3.5 million km² in the Caribbean (Lessios, 2016), and started on individuals next to the Atlantic entrance of the Panama Canal. Disease symptoms included loss of tube feet control, spine shedding, and death within days (Lessios et al., 1984; Bak et al., 1984). Within 13 months, mortality had spread to all parts of the Caribbean, and no *D. antillarum* population is known to have escaped this event (Lessios, 2016). The cause was never confirmed, but its spread and host-specificity suggested a water-borne pathogen (Lessios, 1988). The collapse led to macroalgae and turf dominance on several reef types, inhibiting coral recruitment and reef resilience (Carpenter, 1986; Mumby et al., 2007). Recovery of *D. antillarum* had been slow and patchy, limited by poor fertilization, low larval supply, predation, and competition (Rogers and Lorenzen, 2008; Harborne et al., 2009; Feehan et al., 2016; Hylkema et al., 2023). By 2015 densities were only ~12% of pre-die-off levels (Lessios, 2016). Where recovery occurred, macroalgae cover decreased and coral recruitment increased, illustrating the critical role this species has on Caribbean coral reefs (Idjadi et al., 2010).

A second mass die-off began in January 2022 in the U.S. Virgin Islands. A collaborative region wide effort, the Diadema Response Network, formed to track and understand the cause of this recent die-off (www.agrra.org/sea-urchin-die-off/). In a span of just a few months, over 750 reports were submitted to the AGRRA Diadema Tracking map and 25 countries/territories in the Caribbean had confirmed signs of the die-off. The 2022 mass mortality had some similarities to the 1983-84 event including that it was a water-borne outbreak, highly infectious to populations, highly lethal to individuals, showing similar signs of disease, and



↑ **Figure 1.2.12.** Healthy *Diadema antillarum* in coral reefs of Mahahual, in 2006 (A, credit: H. Bahena/ECOSUR). Diseased *D. antillarum* in Saint-Barthélemy (B, credit: Agence Territoriale de l'Environnement, Saint-Barthélemy).

rapid death of urchins affected. However, there were differences from the 1983-84 event including different originations and shorter duration (half as long). Mortality estimates reached 98–99% in several locations during both outbreaks (Levitan et al., 2023; Hylkema et al., 2023). Symptoms mirrored 1983–84 (**Figure 1.2.12**) and molecular analysis in 2022 from a large-scale collaborative effort of pathologists and resource managers linked the outbreak to the scuticociliate pathogen *Philaster apadigitiformis* (Hewson et al., 2023). Similar events were later recorded in the Mediterranean, Red Sea, and Indian Ocean (Zirler et al., 2023; Roth et al., 2024; Quod et al., 2024). While the origin and triggers of this pathogen remain unknown, the identification of the pathogen and the collaborative networks formed are helping to guide monitoring and management response efforts and prepare for potential disease outbreaks.

The two *D. antillarum* die-offs and poor recovery have driven a lasting shift from coral to algal dominance on Caribbean reefs (Manuel et al., 2021). In response, restoration initiatives have begun across the region. Restocking efforts in Puerto Rico and Curaçao reduced turf and macroalgae, promoted crustose coralline algae, and enhanced coral settlement (Williams et al., 2022; Latijnhouwers et al., 2024). Similar projects in Saba and the Dominican Republic found that artificial settlement substrates and urchin presence improved recruitment and coral survival (Hylkema et al., 2022; Cano et al., 2021).

Despite these successes, large-scale restoration faces challenges. Reliable culture methods were only recently established (Pilnick et al., 2021; Wijers et al., 2023), settlement and grow-out remain difficult to scale, and released urchins suffer high predation without prior adaptation (de Breuyn et al., 2023; Wijers et al., 2024). Advances in larval culture and in raising urchins in natural reef-like conditions are expected to make larger-scale restoration feasible within a few years, hopefully boosting coral recovery efforts across the Caribbean.

2.3.3. Fish diseases

Reef fish are vulnerable to parasitism, which affects growth, mortality, behavior, and community structure (Cook et al., 2015). Blood samples from 6 Caribbean islands revealed 8 parasite species infecting 14 fish species across 8 families, with damselfishes showing the highest incidence. Given their role in structuring benthic communities and serving as prey for piscivores, this has important ecological implications.

Black-spot syndrome (BSS), linked to the trematode *Scaphanocephalus expansus*, has recently increased among Caribbean surgeonfish, with prevalence rising from 39% in 2017 to 51% in 2018 (Dennis et al., 2019). Across 41 species studied, 29 were infected, with a prevalence of 62%, especially in large-bodied herbivores (Malawauw et al., 2024). Infections reduce body condition by triggering immune and oxidative stress responses (Cohen-Sanchez et al., 2023). Since herbivorous fishes maintain reef resilience by controlling algal cover (Tebbet et al., 2022; Faricha et al., 2023), diseases may compromise this role, accelerating coral-to-algae shifts.

Mass mortalities of reef fish were reported in Florida and the Caribbean in the 1990s, with causes uncertain but linked to parasites (*Brooklynella hostilis*, *Uronema marinum*, and amoebae), cold

upwelling, or toxins from macroalgae or benthic dinoflagellates (Landsberg, 1995). The black-spot syndrome was first documented in Bonaire in 1985, where prevalence was higher than in other regions (Elmer et al., 2019). However, museum specimens show *Scaphanocephalus* infections as early as 1948, though at much lower prevalence (<10%) compared to today (71%) (Johnson et al., 2024). This suggests a significant increase in parasite pressure over the past century, though the drivers remain poorly understood.

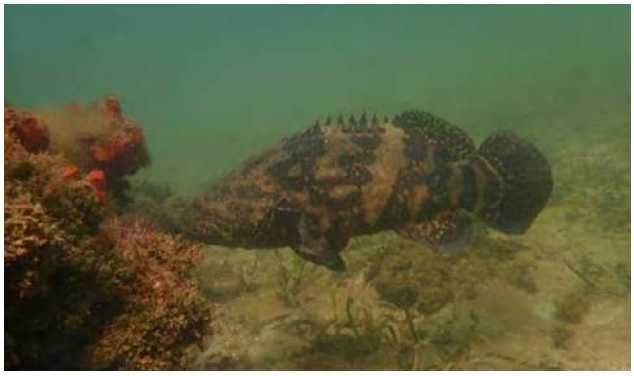
2.4. UNSUSTAINABLE FISHING

Coral reefs across the Caribbean are increasingly threatened by unsustainable fisheries targeting carnivorous fish, herbivores, and invertebrates. Overfishing of top predators like groupers, snappers, and sharks has disrupted food webs and reduced reef resilience, while the depletion of herbivores such as parrotfish has led to algal overgrowth and declining coral health. Key invertebrate species, including spiny lobsters and queen conch, also face steep declines due to intense fishing, habitat degradation, pollution and bycatch, and weak enforcement of conservation measures. These combined impacts threaten marine biodiversity, ecosystem functions, and the livelihoods of coastal communities throughout the region.

2.4.1. Carnivores

In Latin America and the Caribbean, small-scale fisheries are vital to culture, livelihoods, and food security, yet unsustainable targeting of predatory fish such as groupers, snappers, and sharks is causing serious ecological and economic impacts. Despite rising effort, catches of key species have dropped below historical levels, with overfishing pushing fishers toward smaller, lower-trophic species (Oribe-Perez et al., 2023; Vadziutsina and Riera, 2021).

In Antigua and Barbuda, centuries of overexploitation have severely reduced Nassau (*Epinephelus striatus*) and Goliath (*Epinephelus itajara,* **Figure 1.2.13**) grouper populations (Horsford, 2008). Similar trends are observed in the southeastern Caribbean, with a focus on yellowedge grouper (*Hyporthodus flavolimbatus*) and southern red snapper (*Lutjanus purpureus*) (Mendoza and Larez, 2004). While spawning aggregations are critical for the resilience of some coral reef fish populations and for the sustainability of many coastal fisheries, they are targeted by fisheries of groupers and snappers in Cuba, and black grouper and mutton snapper in Belize (de la Guardia et al., 2018; Babcock et al., 2018). Weak enforcement in Colombia has failed to curb the decline of snappers and groupers populations (Caceres et al., 2022). Shark populations, particularly the Caribbean reef shark (*Carcharhinus perezi*), also face heavy pressure in Colombia, Venezuela, and Belize, where protection within marine reserves has not prevented declines near fishing grounds (Tavares et al., 2024; Clementi et al., 2021; Flowers et al., 2022).



↑ **Figure 1.2.13.** Goliath grouper (*Epinephelus itajara*) in coral reefs of Martinique, in 2023. Credit: J. Letellier, Impact Mer

The depletion of top predators has triggered trophic cascades across Caribbean reefs. Predator biomass has fallen by 80–95%, even within reserves, reducing top-down control and allowing mesopredators and prey fish to proliferate (Valdivia et al., 2017; Sandin et al., 2022; Boaden and Kingsford, 2015). In Belize, declines of large groupers and snappers coincided with population booms of smaller species such as *Cephalopholis fulvus* and *C. cruentatus*. These shifts alter foraging dynamics, intensify predation on certain fish species, and contribute to ecosystem imbalances (Mumby et al., 2012). The cascading effects have been shown to extend to habitat conditions on heavily fished reefs in Jamaica, with coral cover and structural complexity declining, and algal cover increasing significantly (Hawkins and Roberts, 2004). Historically, Caribbean reefs had an inverted biomass pyramid dominated by carnivores, but overfishing has disrupted this structure (Valentine and Heck, 2005).

Predatory fish populations in the Caribbean have experienced major declines under heavy fishing pressure. In Belize, Nassau grouper spawning sites have been protected since 2003, yet numbers at Northeast Point dropped from over 3,000 individuals in 2001 to just 450 by 2015, likely due to illegal fishing (Perez and Tewfik, 2015). In Mexico,

a once-massive Nassau grouper aggregation at Mahahual disappeared by 1996 under intense exploitation (Aguilar-Perera, 2006), while goliath grouper collapsed in the 1970s–80s and now persist only in small numbers near oil platforms (Bravo-Calderon et al., 2021). Curaçao shows similar long-term declines, with large groupers nearly gone from artisanal catches by the mid-20th century (Vermeij et al., 2019). Snappers have also suffered: at Gladden Spit, Belize, mutton snapper catch per unit effort dropped by 59% between 2000–2002 (Graham et al., 2008). Sharks have also declined, with records of sustained overfishing since the 1960s in Mexico's Islas Marías (Shaff et al., 2023).

Overfishing and climate change further threaten predatory fish populations (Townhill et al., 2021). Warming seas, habitat degradation, and extreme weather reduce fish productivity, with projections of biomass declines over 10% by mid-century in many small island states and over 20% by 2100 in parts of Central America (Blanchard and Novaglio, 2024). Even with improved management, some countries such as Barbados, Cuba, and Jamaica may face seafood deficits by 2050 (Teneva et al., 2023). Despite these challenges, conservation success stories highlight the potential for recovery when strong protective measures are implemented. In the Cayman Islands, Nassau grouper biomass rebounded to pre-exploitation levels after 16 years of protection (Stock et al., 2021). In the U.S. Virgin Islands, the Red Hind Bank no-take zone has increased fish size and reproductive capacity after three decades of protection (Rosemund et al., 2022). These cases highlight the value of strong, long-term fisheries management for sustaining Caribbean fish populations.

2.4.2. Herbivores

Parrotfish (Scarinae) are crucial for coral reef health, controlling algal growth and promoting coral recruitment (Callwood, 2021; Harms-Tuohy, 2021). Unsustainable fishing of parrotfish and other herbivores threatens Caribbean reefs and fisheries. Despite regulations, over 70% of Caribbean nations maintain active parrotfish fisheries, often replacing snapper/grouper fisheries, with *Sparisoma viride* and *Scarus vetula* commonly caught via traps and spearfishing (Harms-Tuohy, 2021). In The Bahamas, half of fishers target parrotfish specifically (Callwood, 2021). Rising commercial demand and food security needs have driven biomass declines, especially in heavily fished areas like Haiti, the Dominican Republic, and Jamaica (Vallès et al., 2015; Harms-Tuohy, 2021; Croquer et al., 2022; Steneck and Torres, 2023). Fishing has caused size-dependent declines, reducing medium-bodied species biomass (Rivas et al., 2022).

↑ **Figure 1.2.14.** Parrotfish (*Scarus taeniopterus* and *S. vetula*) grazing on a reef in Westpunt, Curaçao, in 2024. Credit: M. Vermeij.

Herbivore fish also enhance reef resilience by removing sediment and epilithic algae, aiding coral survival (**Figure 1.2.14**, Akita et al., 2022). While functional groups such as scrapers, excavators, and browsers are important, their macroalgae control is limited (Dell et al., 2020; Molina-Hernandez and Alvarez-Filip, 2024). Only a few species, notably *Acanthurus coeruleus* and *Sparisoma aurofrenatum*, regularly consume dominant macroalgae, leaving reefs vulnerable when these species decline. Herbivore depletion can increase coral overgrowth by sponges, with overfished sites often showing up to three times more coral-sponge contact than less-fished areas (Loh et al., 2015). The impact of declining parrotfish populations is further complicated by changes in their size structure, which leads to reduced grazing efficiency, accelerating coral decline and macroalgal-sponges dominance (Edwards et al., 2014; Mumby, 2006; Adam et al., 2015; Shantz et al., 2019).

Declines in parrotfish population abundance and size structure reflect both historical and modern fishing pressures. In Jamaica, depletion began in the 1850s, accelerated by changes in trap designs, recreational fishing, and later government subsidies (Hardt, 2009). Similar trends occurred in St. Vincent and the Grenadines, Belize, and The Bahamas, with large and medium-bodied species disproportionately lost (Mohammed et al., 2003; Paddack et al., 2009; Mumby et al., 2012; Muraoka et al., 2022; Sherman et al., 2022). By the late 20th and early 21st centuries, widespread declines in reef fish, including parrotfish, became

evident across the Caribbean. Habitat degradation contributed to an overall loss in fish density, with declines between 2.7% and 6.0% per year across different trophic groups (Paddack et al., 2009).

The future of Caribbean parrotfish is uncertain as new fisheries emerge, threatening populations and reef health. As traditional fisheries for species like groupers and snappers decline, new fisheries for parrotfish have begun to emerge, as seen in The Bahamas (Sherman et al., 2018). However, effective management can drive recovery: in Belize, a 2009 regulation fully protecting parrotfish and surgeonfish nationwide allowed parrotfish biomass to surpass pre-fishery levels (Mumby et al., 2021). This was followed by similar full protections in the remainder of the Mesoamerican Reef from 2010 to 2025 (including Honduras, Guatemala and Mexico; Mcfield et al., 2024). In Bermuda, a fish trap ban in 1990 and listing parrotfish as protected species in 1993 resulted in a 30% increase in Acanthuridae biomass in four years and 30% increase in Scaridae biomass within six years (Luckhurst and O'Farrell, 2014). Well-enforced conservation can sustain both populations and the critical ecosystem services they provide.

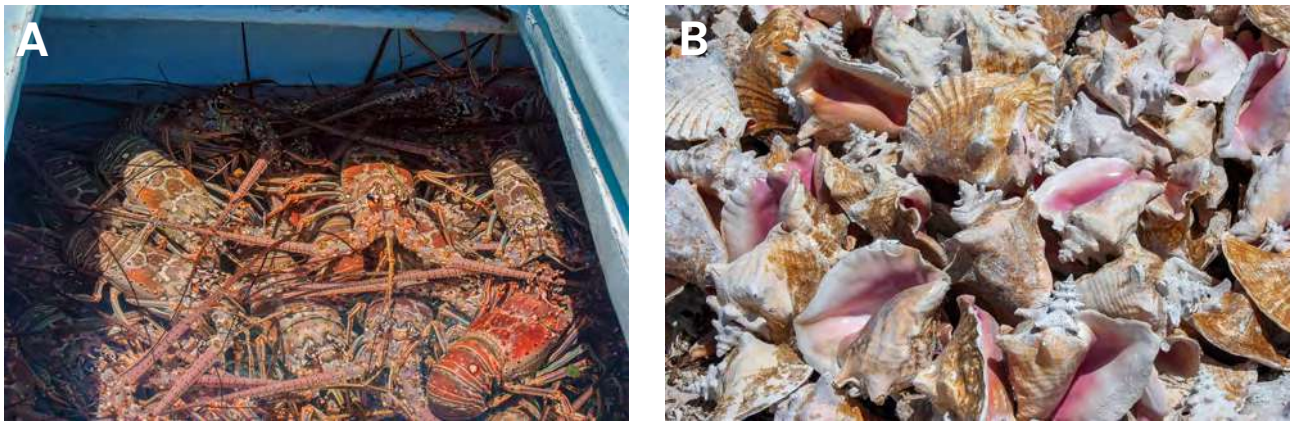
2.4.3. Invertebrates

The Caribbean spiny lobster (*Panulirus argus*) and queen conch (*Aliger gigas*) are among the region's most valuable fisheries (**Figure 1.2.15**). The Caribbean produces ~42,000 tons of lobster annually, but many populations are overexploited (Chavez, 2009). In The Bahamas, lobster accounted for 80–90% of fisheries value between 1995–2015, with 90% exported, representing 13% of U.S. imports (Sherman et al., 2018). However, illegal, unreported, and unregulated (IUU) fishing makes up ~36% of Bahamian landings. High fishing pressure, habitat loss, and removal of large lobsters also threaten stocks in Anguilla, Belize, and Cuba, with Belize exceeding sustainable harvests (Wynne and Coté, 2007; Tewfik, 2023; Muñoz-Nuñez, 2009).

Queen conch generates $3–5 million annually in The Bahamas but is vulnerable due to slow growth, late maturity, and density-dependent reproduction. Many areas have fallen below reproductive thresholds of 50–75 adults/ha (Garcia Rodriguez and Fanning, 2018). Conch densities have sharply declined in the Grenadines, deeper Bahamian waters, and Belize, with juveniles constituting 38% of the catch (Acosta, 2006; Stoner et al., 2012).

Declines of lobsters and conchs have major ecological impacts. Lobsters act as omnivorous mesopredators, controlling invertebrate populations, recycling nutrients, and serving as prey for predators. Juveniles shelter in seagrass and mangroves before moving to reefs, linking habitats and enhancing nutrient flow (Acosta and Robertson, 2003; Briones-Fourzan et al., 2019). Overharvesting reduces reproduction, connectivity, and survival, and disrupts benthic communities (Sherman et al., 2018; Buttler et al., 2022). Conchs maintain reef resilience by grazing algae; their collapse destabilizes seagrass and macroalgal systems, threatening biodiversity and ecosystem stability (Horn et al., 2022; Sherman et al., 2018).

Other invertebrates, including the white sea urchins (*Tripneustes ventricosus*) and cephalopods, have been traditionally harvested in several islands (Lawrence, 2007), and provide economic and ecological services but remain poorly monitored (Lawrence, 2007; Juhl, 1973). Sea cucumbers are increasingly targeted for export markets,



↑ **Figure 1.2.15.** Caribbean spiny lobster (*Panulirus argus*) in a fishing boat in Punta Allen, Mexico, in 2009 (A), and shells of queen conch (*Aliger gigas*) on a beach in Chinchorro, Mexico, in 2008 (B). Credit: H. Bahena/ECOSUR.

particularly to Asia, and have been overexploited in parts of the Caribbean, with fragile stocks and limited recovery potential (Toral-Granda et al., 2008). These fisheries highlight both the diversity of invertebrate resources and the urgent need for better monitoring, management and assessment beyond the region's flagship species.

Historically, major invertebrate fisheries have followed patterns of expansion and decline following increasing catch capacities before a collapse of targeted populations. Florida's lobster fishery peaked in 1972 (Labisky et al., 1980; Hutchinson et al., 2024); Belize fisheries grew in the 1960s but declined after the 2000s (Tewfik, 2023; Tewfik et al., 2020); Cuba fisheries collapsed in 1990; and Puerto Rico and St. Eustatius also experienced overfishing (Labisky et al., 1980; Brunel et al., 2021; Muñoz-Nuñez, 2009; Morris et al., 2004). Conch fisheries collapsed in Mexico and the Dominican Republic, while Bahamian stocks declined despite protected zones (Theile, 2001; Stoner et al., 2018).

Future sustainability depends on stronger management, regulations, enforcement, and habitat protection. Cuba's lobster fishery has quotas, size limits, and closed seasons, while Mexico's closures and reduced fishing mortality have shown stock recovery (Muñoz-Nuñez, 2009; Ley-Cooper and Chavez, 2009). However, many CARICOM states still face weak regulations, and even weaker enforcement and IUU fishing (Winterbottom et al., 2012). For conch, protecting deep-water nurseries and restoration projects, such as hatchery releases in Colombia, are critical (Cruz-Marrero et al., 2024; Sabido-Itza et al., 2021; Abello et al., 2021). Strengthened regulations and enforcement at local and regional scales are essential for maintaining these ecologically key species populations and associated fisheries.

2.5. INVASIVE SPECIES

Invasive species are non-native organisms that, once introduced—intentionally or accidentally—can establish, spread, and disrupt local ecosystems through competition, predation, habitat alteration, or disease. In the Caribbean, a growing number of marine species have become invasive in recent decades, including species of fish, soft corals, and encrusting algae. These species pose serious threats to native biodiversity, coral reef health, and fisheries, as they outcompete or prey on native organisms and alter the structure and function of reef ecosystems.

2.5.1. *Pterois volitans*

The invasion of lionfish (*Pterois volitans*) in the Caribbean has been rapid since their first record in Florida in 1985, spreading across the western Atlantic and Caribbean (**Figure 1.2.16**, Côté et al., 2013; Diaz-Ferguson and Hunter, 2019). In places like The Bahamas and Cuba, lionfish initially reached high densities due to year-round reproduction, high fecundity, and a broad diet (Green and Côté, 2010; del Rio et al., 2023). These populations reduced native reef fish abundance and altered community structures, sometimes exceeding the impact of native predators (Albins, 2013; Green et al., 2012). Yet effects vary across the region. Long-term monitoring in Guadeloupe and Saint-Barthélemy shows initial booms followed by declines, with limited long-term impact as lionfish occupied niches left by overfished predators and fisheries helped regulate populations (Cameron and Roopnarine, 2022). Elsewhere, impacts range from severe prey declines to minimal or context-dependent effects (Vallès et al., 2025; Samhouri and Stier, 2021). Overall, lionfish disruptions differ by location, shaped by fishing pressure, predator depletion, and habitat.

↑ **Figure 1.2.16.** Lionfish (*Pterois volitans*) in coral reefs of Xahuayxol, Mexico, in 2019. Credit: H. Bahena/ECOSUR.

After first detection in Florida, lionfish spread quickly: Bermuda by 2000, The Bahamas in 2004, Cuba and other islands by 2007–2008, and Central America and Colombia by 2009 (Schofield, 2009). Genetic evidence points to a small founding population in Florida, with ocean currents aiding larval dispersal and depth and temperature influencing settlement (Betancur-R et al., 2011; Johnston and Purkis, 2011). By 2010, lionfish populations were established in Cuba at depths up to 188 m (del Rio et al., 2023) and reached Barbados by 2011 (Vallès et al., 2023).

Continued future expansion of lionfish populations in the Caribbean remains a concern given their fecundity and adaptability, with potential spread further into the Atlantic and even through the Panama Canal to the Pacific (Diaz-Ferguson and Hunter, 2019). Climate change may also expand suitable habitat in the southeastern U.S.A. (Grieve et al., 2016). Still, targeted removal works: in Roatán, culling reduced biomass by 95% and boosted native prey densities by 30% (Peiffer et al., 2017). Programs in Bonaire and Curaçao have also reduced numbers, though recruitment from deep waters persists (de Leon et al., 2013; Andradi-Brown et al., 2017). On some islands, including Guadeloupe, artisanal fisheries incorporate lionfish into markets, providing further control. Thus, while lionfish will persist in Caribbean reefs, their impacts can be moderated through targeted fisheries and sustained efforts to develop their marketing as a valuable eco-friendly seafood product.

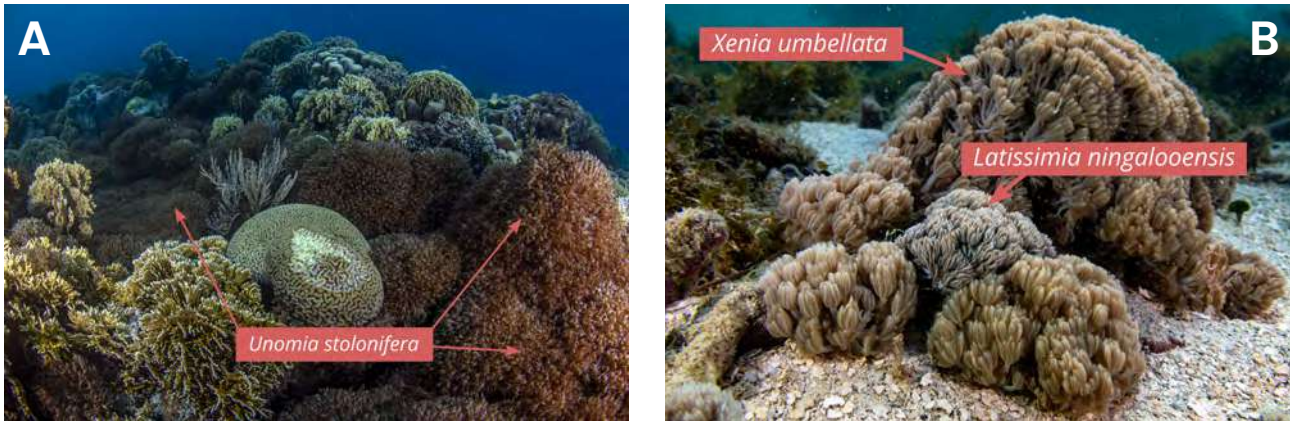
2.5.2. *Unomia stolonifera*, *Xenia umbellata*, and *Latissimia ningalooensis*

Xeniids, soft-bodied octocorals, which are native to the Indian and Pacific Oceans, have invaded several coastal Caribbean reef locations (**Figure 1.2.17**). *Unomia stolonifera* was released in North-East Venezuela in the early 2000s; by 2013, it covered 30–80% of reef area in some regions, reducing hard coral biodiversity (Aguilera-Perez and Gonzales-Diaz, 2024). An invasive xeniid that was first spotted in NW Cuba in 2022 (Espinosa at al., 2023) remained unidentified until it was observed off SW Puerto Rico in 2023 by Toledo-Rodriguez et al. (2025a) and was genetically identified as the distinct *Xenia umbellata* (Jimenez and Vega Rodriguez, 2025). *X. umbellata* is known to have colonised at least 14 shelf-edge, fringing reef sites in Puerto Rico, by overgrowing stony corals, sponges, macroalgae and abiotic substrata at depths extending to ~58m (Jimenez and Vega Rodríguez, 2025; Toledo-Rodriguez et al., 2025). Samples of two xeniid colonies that were growing among *X. umbellata* in a shallow (<3m) Puerto Rican fringing reef in 2024 have been identified by morphological and genetic barcoding analyses as *Latissimia ningalooensis* (Toledo-Rodriguez et al., 2025b), now bringing the total count of Caribbean soft coral invaders to three.

Soft coral invasions have caused major ecological disruptions on Caribbean reefs. *Unomia stolonifera* overgrows corals and seagrass, shifting benthic community structure (Ruiz-Allais et al., 2021). *Xenia umbellata* shows high resistance to warming and eutrophication, and its mixotrophic feeding strategy provides a further advantage under changing conditions and enhances its competitive advantage over native reef species (Simancas-Giraldo et al., 2021; Mezger et al., 2022).

Dispersal of *Unomia stolonifera* was attributed in part to asexual fragments that drifted, were rafted on detached seagrasses, or were entangled on fishing nets that rotated among sites by Ruiz-Allais et al. (2021), who also noted sexually mature colonies. When invasions of Indo-Pacific xeniids replace native stony corals and other sessile native benthos, habitat space for many of a reef's associated invertebrates and fishes may largely disappear.

Control efforts are expensive, given the rapid growth rates of invader species, which may already cover large areas when initially discovered and/or grow in cryptic habitats or deep water. Repeated



↑ **Figure 1.2.17.** (A) *Unomia stolonifera* overgrowing stony corals, fire corals, and octocorals at 8-10 m on a NE Venezuelan rocky reef in 2025 (Credit: H. Ramirez), (B) *Xenia umbellata* and *Latissimia ningalooensis* among corals and seagrasses at <3 m on a SW Puerto Rican fringing reef in 2024. (Credit: D. Toledo-Rodriquez).

culling or removal treatments are frequently needed when tiny fragments that survive the initial treatment remain on the substratum and generate new polyps (Espinosa et al., 2023). Physical control includes asphyxiation with black, weighted tarpaulins in Venezuela (Salazar et al., 2024), or with chlorine tablets under weighted, black plastic in Cuba (UKT, 2025), and entombment in a cement mixture in Puerto Rico (Jimenez and Vega Rodríguez, 2025). Methods of biological (biopesticides) and mechanical (ultrasound plus suction) control that are under investigation in Venezuela may provide additional solutions, but containing large invasions will be especially challenging and multiple approaches may be required (Villamizar, 2024).

2.5.3. Peyssonelids

Peyssonnelid algal crusts (PAC), particularly *Ramicrusta* spp. within the Peyssonneliaceae, have rapidly expanded across shallow Caribbean reefs since 2010, posing a major ecological threat (Stockton and Edmunds, 2021; Wilson et al., 2020). These red-brown encrusting algae replace crustose coralline algae (CCA), overgrow corals, and hinder coral recruitment (**Figure 1.2.18**, Cayemitte et al., 2023). In the U.S. Virgin Islands, PAC has reached up to 86% cover in shallow reefs (Stockton and Edmunds, 2021), and its presence across the Caribbean indicates a widespread invasion (Wilson et al., 2020).

↑ **Figure 1.2.18.** *Ramicrusta textilis* overgrowing a *Porites astreoides* coral colony on January 4, 2024, at Honeymoon Beach, St. John, U.S. Virgin Islands (depth ~ 3 m; credit: Abigail Gretta).

PAC proliferation has severely altered reef ecosystems. The algae deter scleractinian coral recruitment and outcompete reef builders, with experimental studies showing 96–100% mortality of *Porites astreoides* and *Favia fragum* larvae exposed to *Ramicrusta* spp., compared to minimal mortality with CCA (Wilson et al., 2020; Cayemitte et al., 2023). PAC overgrowth has been linked to the mortality of adult corals, with *Orbicella annularis* and *O. faveolata* showing low resistance, leading to reduced species diversity and structural complexity (Hollister et al., 2021). However, herbivores like the sea urchin *Diadema antillarum* can create PAC-free halos that enhance coral recruitment, suggesting a natural mitigation pathway (Stockton and Edmunds, 2021).

First reported in Lac Bay, Bonaire in 2010, PAC spread rapidly to Puerto Rico, the Virgin Islands, and Honduras (Edmunds et al., 2019b; Wilson et al., 2020). In St. John, U.S. Virgin Islands, PAC expanded from 0.2–20.6% cover in 2015 to 61.6% by 2017, and 31–86% by 2019 (Edmunds et al., 2019b; Stockton and Edmunds, 2021). In Puerto Rico, *Ramicrusta* spp. appeared suddenly, rising from absence before 2016 to presence at 76% of monitoring sites by 2018 (William and Garcia-Sais, 2020). This expansion has displaced ubiquitous algal forms like turf algae, accelerating reef shifts.

The continued spread of PAC threatens Caribbean reef resilience. Management should focus on enhancing herbivory, particularly from *D. antillarum*, while research on PAC-microbe interactions may reveal additional biological controls (Wilson et al., 2020). Effective conservation strategies and monitoring will be key to slowing the shift from coral- to algal-dominated reefs.

3. Temporal trends in benthic cover

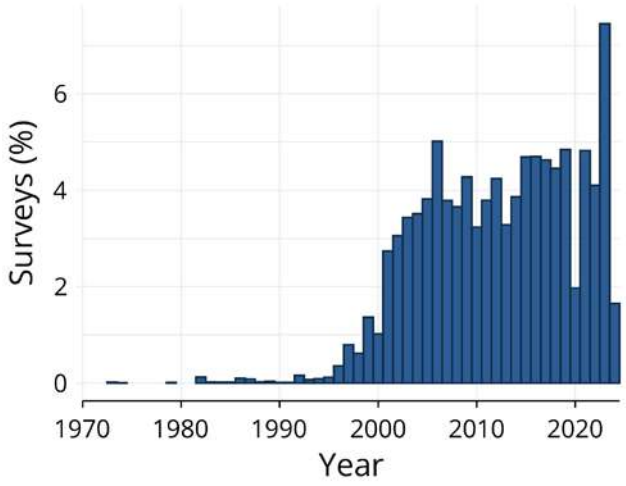
3.1. SPATIO-TEMPORAL DISTRIBUTION OF MONITORING

The estimation of temporal trends in benthic cover of five major benthic categories (hard coral, macroalgae, turf algae, coralline algae, and other fauna), and three hard coral genera (*Acropora*, *Orbicella*, *Porites*) were achieved through the integration of 72 datasets (**Supplementary Table 1**). Together, these datasets represent a total of 13,864 monitoring sites, on which 23,742 surveys were conducted between 1973 and 2024 (**Table 1.3.1**).

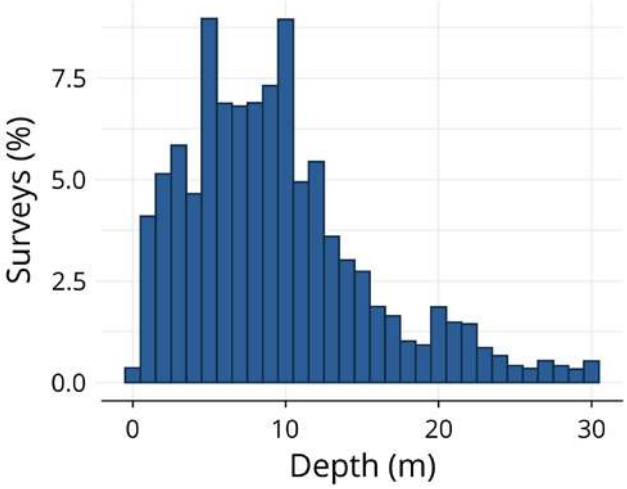
The distribution of monitoring efforts across years was not uniform between 1970 and 2024, as 95% of the total number of surveys were conducted between 2001 and 2024 (**Figure 1.3.1**). The increase in the number of surveys conducted annually between 1999 and 2003 was likely driven by greater funding for coral reef monitoring programs aimed at assessing the impacts of the 1998 mass bleaching event. The low number of surveys in 2020 was likely a consequence of the COVID-19 pandemic, which restricted coral monitoring activities and travel due to stay-at-home measures implemented in several countries and territories. The high proportion of surveys in 2023 likely reflects increased monitoring efforts aimed at assessing the impacts of the bleaching event that occurred that year. The low proportion of surveys in 2024 is attributable to limited data sharing by contributors, in part because processing photo-quadrat images collected that year requires additional time from monitoring teams.

Benthic cover surveys were conducted at depths ranging from 0 to 30 m (**Figure 1.3.2**), corresponding to the distribution of shallow coral reefs. The skew distribution toward shallower depths can be attributed to the increasing time and safety constraints associated with deeper dives. Consequently, the number of surveys that can be completed per dive decreases sharply with depth. In addition, some monitoring protocols in the region also recommend conducting benthic cover surveys at a standard depth of 10 m, an intermediate depth where coral cover and species richness are typically highest. Since the 2000s, citizen science initiatives have made reef monitoring more accessible to recreational divers; however, those with entry-level certifications are restricted to depths of 18 m or less, which may further accentuate the bias toward shallow sites. Finally, certain reef types, such as lagoons, are naturally shallow.

This report does not address the status and trends of mesophotic coral reefs, which occur between 30 and 150 m depth (Rocha et al., 2018). Because of their depth, mesophotic reefs are less accessible and therefore surveyed less frequently. Their depth also precludes the use of satellite-derived data. As a result, the distribution of mesophotic coral reefs remains poorly documented, and assessing the impacts they experience can be more challenging than for shallow coral reefs.



↑ **Figure 1.3.1.** Percentage of benthic cover surveys conducted annually in the GCRMN Caribbean region (1970–2024), relative to all surveys conducted during this period.



↑ **Figure 1.3.2.** Percentage of benthic cover surveys by depth in the GCRMN Caribbean region (1970–2024), relative to all surveys conducted between 0 and 30 m.

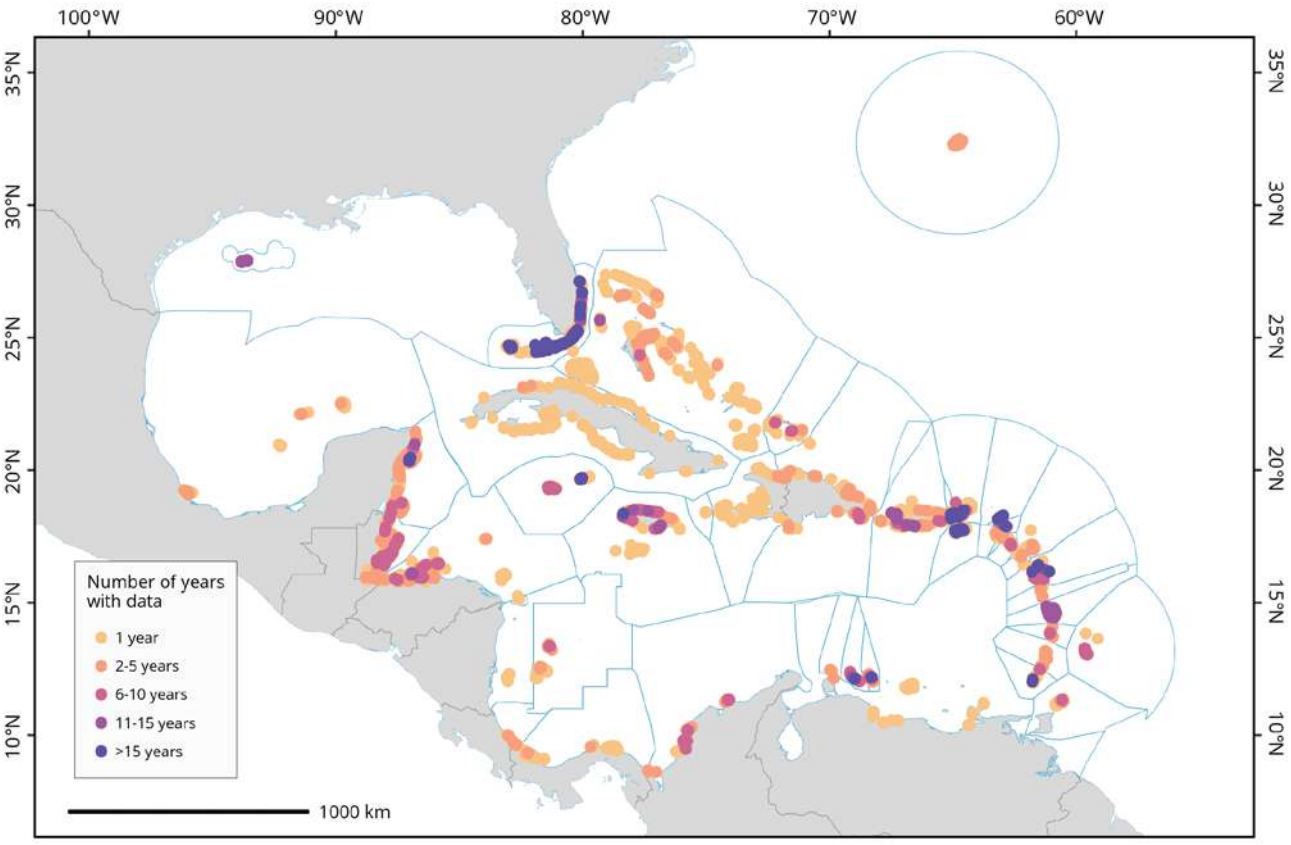
The countries and territories with the highest number of benthic cover monitoring sites and surveys were the U.S. Caribbean territories (U.S. Virgin Islands, Puerto Rico, and Florida; **Table 1.3.1**). In contrast, only a limited number of monitoring sites were available for some countries and territories, such as Saint-Barthélemy, Aruba, and Anguilla (**Table 1.3.1**). It should be noted, however, that this does not necessarily indicate an absence of benthic cover monitoring data for these countries and territories. Rather, some datasets may not have been available for inclusion in this report.

↓ **Table 1.3.1.** Comparison of benthic cover monitoring indicators between countries and territories of the GCRMN Caribbean region. Columns "Sites" and "Surveys" correspond to the number of unique monitoring sites and surveys assessed within the country or the territory. The column "Datasets" correspond to the number of individual datasets with at least one survey from the country or the territory. Columns "First year" and "Last year" are respectively the first and last year where a survey has been conducted within the country or territory. The values reported in the table are provided for all benthic categories combined. Navassa Island is not included in this table, which accounts for the difference between the column totals and the values reported in the last row, "Entire Caribbean region".

Countries and territories	Sites	Surveys	Datasets	First year	Last year
Anguilla	10	132	2	2002	2023
Antigua and Barbuda	41	50	2	2003	2022
Aruba	6	7	1	2003	2009
The Bahamas	542	808	3	1986	2024
Barbados	80	349	3	1982	2022
Belize	440	703	7	1985	2024
Bermuda	43	91	2	1982	2021
Bonaire	154	618	7	1973	2023
British Virgin Islands	34	328	3	1992	2024
Cayman Islands	70	227	5	1997	2024
Colombia	90	258	4	1997	2024
Costa Rica	30	39	2	1999	2023
Cuba	196	205	3	1999	2023
Curaçao	148	438	4	1973	2023
Dominica	31	46	2	2004	2018
Dominican Republic	178	516	8	1999	2024
Florida (U.S.A)	1,232	3,482	8	1986	2024
Flower Garden Banks National Marine Sanctuary (U.S.A)	325	1,297	6	1984	2024
Grenada	99	239	4	2004	2024
Guadeloupe	26	209	4	2002	2024
Guatemala	29	56	3	2006	2023
Haiti	96	109	2	2003	2018
Honduras	405	848	4	1997	2024
Jamaica	308	794	7	1986	2024
Martinique	41	322	3	2001	2024
Mexico (Caribbean Sea)	381	934	7	1997	2024
Mexico (Gulf of Mexico)	123	143	3	1999	2024
Montserrat	87	109	2	2005	2017
Nicaragua	14	14	1	2003	2003
Panama	92	184	4	1997	2024
Puerto Rico	2,825	3,130	7	1982	2024
Saba	30	68	3	1994	2024
Saint Kitts and Nevis	38	55	2	2004	2024

↓ **Table 1.3.1.** (*continuing*)

Countries and territories	Sites	Surveys	Datasets	First year	Last year
Saint Lucia	21	61	1	1999	2014
Saint Vincent and the Grenadines	55	74	4	1999	2024
Saint-Barthélemy	4	43	2	2002	2024
Saint-Martin	12	83	2	2007	2022
Seaflower Biosphere Reserve	90	137	4	1998	2024
Sint-Eustatius	37	72	3	1999	2023
Sint-Maarten	14	61	3	1999	2024
Trinidad and Tobago	52	115	3	2007	2023
Turks and Caicos Islands	87	145	4	1999	2024
United States Virgin Islands	5,183	6,071	10	1982	2024
Venezuela	50	57	3	1999	2018
Entire Caribbean region	**13,864**	**23,742**	**72**	**1973**	**2024**



↑ **Figure 1.3.3.** Spatio-temporal distribution of benthic cover monitoring sites across the GCRMN Caribbean region from 1970 to 2024. Sites that were monitored the largest number of times are represented on top of the sites monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications.

Notes about model result interpretation

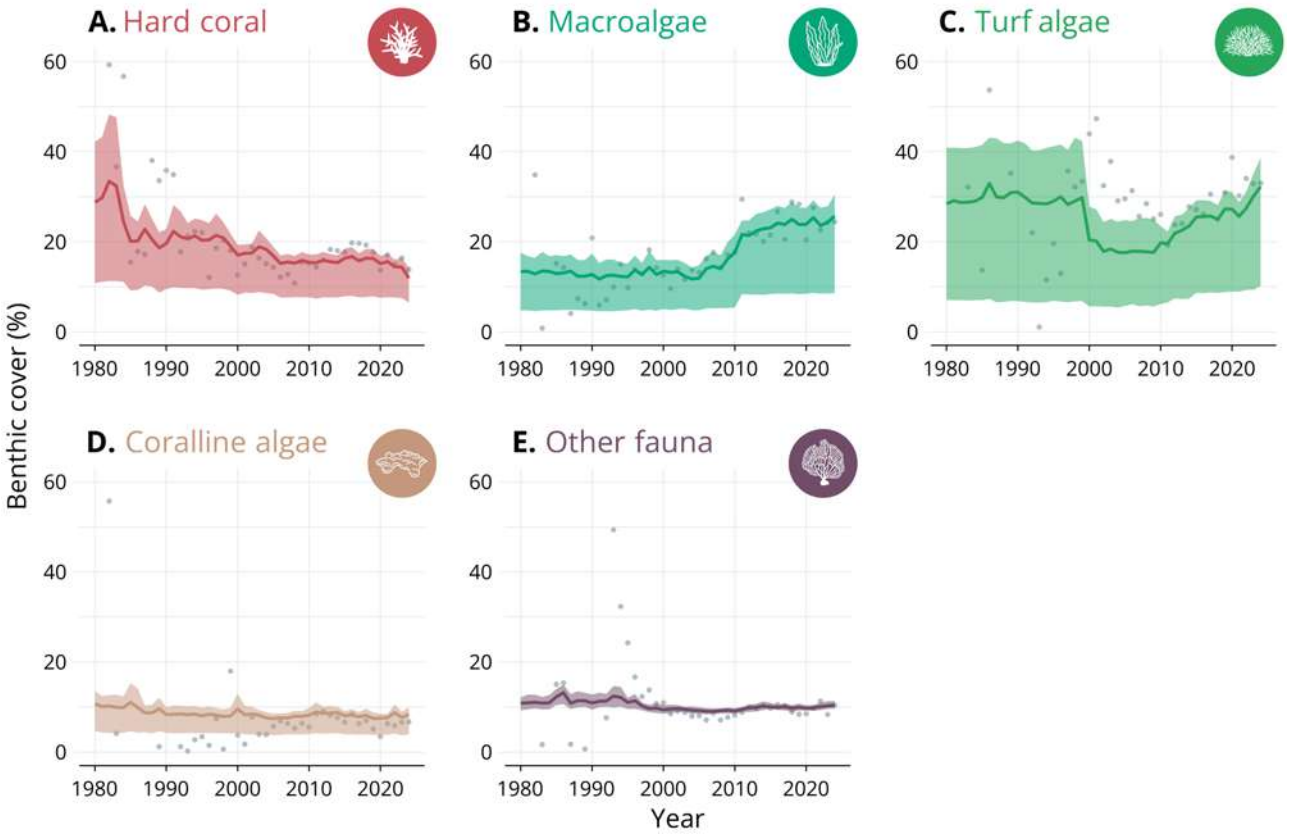
The distribution of coral reef benthic cover monitoring sites across the Caribbean is spatially and temporally uneven (**Figure 1.3.3**). Consequently, calculating the mean percentage cover of a given benthic category per year at the regional scale may yield biased estimates that primarily reflect changes at monitoring sites rather than trends representative of Caribbean coral reefs as a whole.

To address this limitation, we estimated temporal trends in the percentage cover of eight benthic categories using machine learning models. These models leverage relationships between observed percentage cover data and multiple predictors (*e.g.*, sea surface temperature) to predict percentage cover at a set of generated sites designed to be spatially and temporally representative of Caribbean coral reefs (see **Materials and Methods**).

Despite the advantages of this approach, certain sources of bias must be considered to correctly interpret the results presented in the two following sections (**Trends of major benthic groups** and **Trends in hard coral genera**).

1. Modelled values prior to 2001 should be interpreted with caution, as only a limited number of observations were available to train the models between 1980 and 2000 (**Figure 1.3.1**). The greater uncertainty of estimates for this period is partially, though not systematically, reflected in the wider confidence intervals (*e.g.* **Figure 1.3.5.C**).
2. The confidence intervals represent variance among model bootstrap iterations (see **Materials and Methods**). Consequently, the confidence intervals reported here are not directly comparable to sampling standard errors presented in earlier syntheses.
3. The yearly averages of the observed data are represented as grey points on the plots, alongside the modelled temporal trends. These points were included to visualize the raw observations (*i.e.*, the mean values measured in the field across the Caribbean for each year) in addition to the modelled estimates. It is therefore expected that the modelled trends do not perfectly "fit" these yearly averages. Indeed, in certain years, most monitoring sites were concentrated in limited areas of the region and were not spatially representative of Caribbean coral reefs as a whole, a bias explicitly accounted for by the modelling framework.
4. The modelled trends estimated in this study do not account for variations associated with different depths or reef types (*e.g.*, fringing reefs, barrier reefs).

3.2. TRENDS OF MAJOR BENTHIC GROUPS



↑ **Figure 1.3.4.** Modeled temporal trends of benthic percentage cover for hard coral (A), macroalgae (B), turf algae (C), coralline algae (D), and other fauna (E) in the Caribbean from 1980 to 2024. The bold lines indicate the modeled mean benthic cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

3.2.1. Hard coral

Definition. Hard corals are organisms belonging to the order *Scleractinia*, as well as some species of *Hydrozoa* from the family *Milleporidae*. Their hard, calcium carbonate skeletons create a rigid and structurally complex reef framework (Stoddart, 1969), which provides an essential ecological habitat for numerous species.

Results. From 1980 to 2024, the long-term average in the modeled benthic cover of hard coral across coral reefs of the Caribbean was 19% (95% Confidence Interval [8.8%–23.0%]; **Figure 1.3.4.A**). Over this period, the long-term trend in hard coral cover exhibited a significant decline (Mann-Kendall test, τ = -0.70, p-value = 1.38 x 10-11), decreasing from an average of 28.1% (95% CI [10.8%–39.9%]) in 1980–1985 to 14.6% (95% CI [7.45%–16.5%]) in 2019–2024. This represents an absolute decline of 13.5 percentage points, equivalent to a relative reduction of 48% in hard coral cover between the two periods. However, from 1980 to 2024, the trend was not steady. Indeed, hard coral cover marked steep declines in 1983 (-7.9 percentage points, -24.5%), 1998 (-1.9 percentage points, -9%), 2005 (-3.1 percentage points, -17.1%), and 2023 (-2.4 percentage points, -16.9%). Hard coral cover remained relatively stable from 1985 to 1997 at around 20.7%, and from 2006 to 2020 at around 15.8%.

Comparison with earlier research. Direct comparisons with previous studies are difficult because studies differ in site coverage, time window, depth range, spatial extent, and the definition/aggregation of benthic categories. Nevertheless, the average hard coral cover of 24.6% (95% CI [10.2% - 33.0%]) estimated for 1980–1990 aligns closely with the values of 26.2% reported by Gardner et al. (2003), 20.9% reported by Schutte et al. (2010), and 18% reported by Souter et al. (2021). For 2000–2010, the average hard coral cover of 16.6% (95% CI [8.15%–18.8%]) is also consistent with the values of 20.0%, 15.2%, and 16.4% reported by Bruno et al. (2009), Schutte et al.

(2010), and Jackson et al. (2014), respectively. Over the long term (1980–2024), our estimated absolute decline of 13.5 percentage points in hard coral cover is lower than previous estimates, including the ~20 percentage point decline reported by Schutte et al. (2010) for 1971–2006 and the 16.6 percentage point decline reported by Jackson et al. (2014) for 1970–2012. In contrast, Gardner et al. (2003) documented a substantially larger decline, estimating a reduction of approximately 40 percentage points between 1977 and 2001.

Interpretation of temporal trend. Although our modelled trend spans 1980–2024, the year 1980 should not be regarded as a baseline representing pristine Caribbean hard coral cover. For instance, Cramer et al. (2020) demonstrated that *Acropora palmata* and *A. cervicornis,* two species that historically contributed substantially to total hard coral cover, had already started to decline in the 1960s due to direct anthropogenic impacts. Jackson et al. (2014) estimated that hard coral cover from 1970 to 1983 was around 33% (95% CI [28.7%–37.6%]). The apparent increase in hard coral cover between 1980 and 1982 (**Figure 1.3.4.A**) is likely an artefact of the model, attributable to the limited availability of training data at the beginning of the study period (**Figure 1.3.1**), as indicated by the wide confidence intervals. Likewise, the sharp 7.9 percentage point decline in hard coral cover in 1983 seems unrealistically steep and has not been documented in previous studies (Gardner et al., 2003; Schutte et al., 2010), suggesting instead a more gradual decline between 1980 and 1984.

Several factors contributed to the decline in hard coral cover over the 1980s (**Figure 1.3.4.A**). The white-band disease from the late 1970s to early 1990s severely impacted *A. palmata* and *A. cervicornis* (Aronson and Precht, 2006), with occurrences reported across the northeastern Caribbean, Curaçao, Nicaragua, Panama, and Florida (Gladfelter, 1982). In Belize's Channel Cays, *A. cervicornis* cover collapsed from ~70% in 1986 to 0% by 1997 (Aronson and Precht, 2001). Additionally, the mass die-off of the herbivorous sea urchin *Diadema antillarum* in 1983–1984 (Lessios, 1988; **Section 2.3.2**) removed a key grazing pressure from the system. Combined with long-term declines in herbivorous fishes (McClenachan, 2009; Paddack et al., 2009) and nutrient enrichment, this may have facilitated algal proliferation, which likely inhibited coral recruitment and reduced the ability of hard coral cover to recover from other disturbances (Arnold et al., 2010; Speare et al., 2019).

Elevated sea surface temperatures (SST) in 1987 and 1995 (Cetina-Heredia and Allende-Arandía, 2023; **Figure 1.2.1.B**) triggered bleaching events, affecting ~20% of reefs in the region (Roberts, 1987; Gill et al., 2006), further exacerbating hard coral loss between 1985 and 1997. In 1998, the world experienced the first global coral bleaching event, triggered by elevated SST associated with a strong El Niño event (Cetina-Heredia and Allende-Arandía, 2023). The 1.9 percentage point decrease (representing a decline of 9.1%) in hard coral cover recorded that year (**Figure 1.3.4.A**) was very likely driven by this event. A second major bleaching episode occurred in 2005, coinciding with a 3.1 percentage point decrease (or a relative decline of 16.9%) in hard coral cover (**Figure 1.3.4.A**). This event was intense, with mean SST anomalies of ~0.8 °C relative to the 1901-2000 period (Eakin et al., 2010). Muñiz-Castillo et al. (2019) estimated that 42% of Caribbean coral reefs were at risk of mortality during this event. The 2005 coral bleaching event was exceptionally destructive in the Eastern Caribbean (Wilkinson and Souter, 2008). The exceptionally high number of hurricanes in 2005 may also have contributed to coral losses, although they could have simultaneously mitigated thermal stress through water column mixing (Eakin et al., 2010).

Between 2006 and 2019, regional hard coral cover remained relatively stable at approximately 15.8% (**Figure 1.3.4.A**). Accordingly, major marine heatwaves in 2010 and 2015–2017 (Cetina-Heredia and Allende-Arandía, 2023; Muñiz-Castillo et al., 2019) appear to have exerted only limited effects at the regional scale. However, the absence of a regional trend does not imply that local coral reefs were unaffected.

In contrast to this period of stability, a pronounced decline occurred between 2020 and 2024, with hard coral cover decreasing by 3.2 percentage points, corresponding to a 21.3% decline. The majority of this decline occurred in 2023 and 2024 and was likely driven by the fourth global bleaching event (Reimer et al., 2024). Indeed, following this event, widespread bleaching and mortality have been documented at several Caribbean sites, including the Cayman Islands (Doherty et al., 2025) and Martinique (Bon et al., 2025). We likely have underestimated the magnitude of this decline, as the bleaching event extended into 2024 and 2025 and the proportion of surveys included in our analysis for 2024 was limited (**Figure 1.3.1**). The 2020-2024 decline was also likely exacerbated by the Stony Coral Tissue Loss Disease (SCTLD).

First reported in Florida in 2014, SCTLD gradually spread across the region, reaching most Caribbean countries and territories by 2023 (Papke et al., 2024). High SCTLD prevalence has been observed throughout the Caribbean since 2019, with significant hard coral cover losses reported in Mexico (Estrada-Saldívar et al., 2021), U.S. Virgin Islands (Brandt et al., 2021), and Puerto Rico (Lucas et al., 2024), among others.

Model performance. The temporal trend of benthic cover of hard coral across Caribbean coral reefs from 1980 to 2024 (**Figure 1.3.4.A**) was estimated using data from 13,797 monitoring sites and 45,287 observations employed for model training. The model achieved a Root Mean Squared Error (RMSE) of 9.27, while accounting for 70% of the variance in the data (R^2 = 0.70).

3.2.2. Macroalgae

Definition. We defined macroalgae as erect algal forms, belonging to the classes Phaeophyceae, Florideophyceae (excluding the order Corallinales), and Ulvophyceae, and not identified by data contributors as turf algae. The dominant macroalgal genera in the Caribbean include *Dictyota*, *Lobophora*, and *Halimeda* (Williams and Polunin, 2001).

Results. From 1980 to 2024, the long-term average modeled benthic cover of macroalgae across coral reefs of the Caribbean was 16.5% (95% CI [6.0% - 20.0%]; **Figure 1.3.4.B**). Over this period, the long-term trend in macroalgae cover exhibited a significant increase (Mann-Kendall test, τ = 0.55, p-value = 1.19 x 10-7), rising from an average of 13.2% (95% CI [4.7% - 17.1%]) in 1980–1985 to 24.4% (95% CI [8.51% - 28.4%]) in 2019–2024. This represents an absolute increase of 11.2 percentage points, equivalent to a relative increase of 84.8% in macroalgae cover between the two periods. However, the trajectory of macroalgae cover over the study period was not monotonic, and three distinct phases can be identified. From 1980 to 2005, macroalgae cover remained relatively stable at approximately 12.8%. A sharp increase occurred between 2005 and 2011, with macroalgae cover rising by 9.7 percentage points, corresponding to a 81.5% relative increase. Thereafter, from 2011 onward, macroalgae cover continued to increase but at a slower rate, reaching 25.7% in 2024.

Comparison with earlier research. The average macroalgae cover of 13.2% (95% CI [4.7% - 17.1%]) estimated for 1980-1985 was above the values of 8% (1977-1983) from Schutte et al. (2010) and 7% (1970-1983) from Jackson et al. (2014). For the period 1995–2005, the average macroalgae cover of 12.9% (95% CI [4.9% - 15.8%]) was consistent with the value of 15.3% (2001-2005) estimated by Schutte et al. (2010), and the one of ~10% reported by Tebbett et al. (2023). However, it was lower than the estimates of 23.6% (1996-2006) and 23.2% (1999-2012) from Bruno et al. (2009) and Jackson et al. (2014), respectively. Over the long-term (1980-2024), the absolute increase of 11.2 percentage points in macroalgae cover reported here, is slightly lower than the ~12 percentage point increase estimated by Schutte et al. (2010), and lower than the 16.5 percentage point increase (1970-2011) estimated by Jackson et al. (2014).

Interpretation of temporal trend. Jackson et al. (2014) estimated that macroalgal cover in coral reefs of the Caribbean between 1970 and 1983 averaged approximately 7% (95% CI [3.6%–13.0%]). Schutte et al. (2010) reported a drastic rise in macroalgal cover in 1986, increased from 10% to more than 35% within a single year, followed by a subsequent decline. This rise has been attributed to the mass die-off of the sea urchin *Diadema antillarum* across the Caribbean in 1983–1984 (Lessios, 1988). Within just two years, population densities of *D. antillarum* declined by 87–100% at multiple sites (Hughes et al., 1987). In the absence of this key herbivore, algae proliferated rapidly, followed by a slight decline the following year, likely due to compensatory grazing by herbivorous fishes (Carpenter, 1990b; Schutte et al., 2010). For instance, in Discovery Bay (Jamaica) at 15 m depth, the benthic cover of non-coralline algae rose from 30.7% in August 1982 to 72.3% in December 1983, before decreasing to 64.7% in August 1984 (Liddell and Ohlhorst, 1986). Four decades after this mass mortality event, *D. antillarum* populations remained low across much of the Caribbean (Hughes et al., 2010; Tuohy et al., 2020). Contrary to Schutte et al. (2010), we did not detect such an abrupt increase in macroalgae cover in 1986 (**Figure 1.3.4.B**), which may reflect the limited amount of data available for the 1980–1990 decade in our analysis. However, the relative stability of macroalgae cover from 1990 to 2005, averaging around 12.7% (**Figure 1.3.4.B**), is consistent with the results of Schutte et al. (2010) and Tebbett et al. (2023).

From 2005 to 2011, macroalgae cover increased by 81.5%, coinciding with the decline in hard coral cover associated with the 2005 bleaching event. Indeed, the loss of hard coral provides available substrate for macroalgal colonization (Szmant,

2002). In addition to this disturbance-driven effect, several chronic drivers may have contributed to the observed increase in macroalgae cover. For example, the human population living within 20 km of coral reefs increased by approximately 40% between 2000 and 2020 (**Figure 1.2.6**), a factor shown to correlate positively with macroalgal abundance (Mora, 2008). This demographic expansion has likely led to declining coastal water quality, reflected in elevated nutrient loads, sedimentation, and turbidity (Berger et al., 2022; Jackson et al., 2014). Coastal development, agricultural land use, and inadequate wastewater management are among the primary drivers of this deterioration (Carlson et al., 2019; Wakwella et al., 2023). By increasing nitrogen and phosphorus concentrations in nearshore environments, these anthropogenic pressures have likely promoted the increase of macroalgae cover (McClanahan et al., 2003). Furthermore, unsustainable fishing pressure on herbivorous species, such as parrotfishes (Scaridae) and surgeonfishes (Acanthuridae), may have reduced grazing intensity, thereby facilitating further macroalgae proliferation (Paddack et al., 2009; McClanahan et al., 2003).

The reasons for the slowdown in macroalgal cover increase from 2011 to 2024 (**Figure 1.3.4.B**) remain unclear but may be linked to the limited availability of suitable substrate for colonization. In 2022, a new mass mortality of *D. antillarum* occurred in the Caribbean, with symptoms and mortality patterns strikingly similar to those documented during the 1983–1984 event (Hylkema et al., 2023). Nonetheless, the resulting macroalgae cover increase is expected to be less pronounced than in the 1980s, as the recent decline in *D. antillarum* populations appears, in absolute terms, less severe than that observed during the earlier event. Finally, macroalgae cover has increased more rapidly in 2023 and 2024, likely a result of the decline in hard coral cover (**Figure 1.3.4.A**).

Model performance. The temporal trend of benthic cover of macroalgae across Caribbean coral reefs from 1980 to 2024 (**Figure 1.3.4.B**) was estimated using data from 11,454 monitoring sites and 41,592 observations employed for model training. The model achieved a RMSE of 12.58, while accounting for 64% of the variance in the data (R^2 = 0.64).

3.2.3. Turf algae

Definition. Turf algae were defined as short (<2 cm) filamentous forms growing on either hard or soft substrates. It should be acknowledged that the term turf algae remains debated within the scientific community due to the varying definitions adopted across monitoring programs (Connell et al., 2014). Nevertheless, we considered it informative to present the temporal dynamics of turf algae, as this group is functionally distinct from both macroalgae and coralline algae in the context of coral reefs (Tebbett and Bellwood, 2019). Turf algae can overgrow hard corals (Vermeij et al., 2010) and may limit hard coral recruitment (Arnold et al., 2010; Speare et al., 2019).

Results. From 1980 to 2024, the time-averaged modeled benthic cover of turf algae across coral reefs of the Caribbean was 25.6% (95% CI [7.0% - 33.8%]; **Figure 1.3.4.C**). Between 1980 and 2000, turf algae cover fluctuated substantially around an average of 25.3% on an interannual basis (95% CI [6.0% - 20.0%]; **Figure 1.3.4.C**). From 2001 to 2009, turf algae cover remained stable at around 18.1% (95% CI [5.7% - 25.1%]). Then, turf algal cover increased from 2010 onward, reaching 32.2% in 2024. This represents a rise of 12.7 percentage points, corresponding to a 64.7% increase over the period.

Comparison with earlier research. To our knowledge, no studies have assessed the temporal dynamics of turf algae benthic cover in the Caribbean region. Consequently, direct comparisons with previous research were not possible.

Interpretation of temporal trend. Between 1980 and 2000, turf algae cover fluctuated considerably on an interannual basis, as indicated by the wide confidence intervals (**Figure 1.3.4.C**). Although part of this variability may reflect natural spatial and temporal fluctuations, data sparsity and inconsistencies in taxonomic categorization among monitoring programs likely also contributed. This high variability hinders the identification of a clear temporal pattern during the first two decades and makes the estimation of the long-term trend (1980–2024) unreliable. After a period of relative stability during the 2000s, turf algae cover began to increase gradually from 2010 onward. The drivers of this increase are likely similar to those identified for macroalgae cover. The decline of hard coral cover following bleaching events has created available substrate for colonization (Szmant, 2002). Concurrently, bottom-up processes,

such as nutrient enrichment (Berger et al., 2022; Jackson et al., 2014), may have promoted turf algae proliferation, whereas top-down controls may have weakened due to the decline of herbivorous fish populations (Paddack et al., 2009; McClanahan et al., 2003) and *D. antillarum* sea urchins (Tuohy et al., 2020), resulting in reduced grazing pressure.

Model performance. The temporal trend of benthic cover of turf algae across Caribbean coral reefs from 1980 to 2024 (**Figure 1.3.4.C**) was estimated using data from 11,623 monitoring sites and 30,831 observations employed for model training. The model achieved a RMSE of 14.46, while accounting for 61% of the variance in the data (R^2 = 0.61).

3.2.4. Coralline algae

Definition. We defined coralline algae as species belonging to the order Corallinales. These encrusting red algae produce calcium carbonate, thereby contributing to the cementation of reef frameworks and facilitating hard coral recruitment (Littler and Littler, 2013; Cornwall et al., 2023).

Results. From 1980 to 2024, the long-term average modeled benthic cover of coralline algae across coral reefs of the Caribbean was 8.5% (95% CI [4.1% - 10.4%]; **Figure 1.3.4.D**). Over this period, the long-term trend in coralline algae cover exhibited a significant decline (Mann-Kendall test, τ = -0.45, p-value = 1.23 x 10-5), decreasing from an average of 10.3% (95% CI [4.4% - 13.2%]) in 1980–1985 to 7.9% (95% CI [3.9% - 9.2%]) in 2019–2024. This corresponds to an absolute decline of 2.4 percentage points in coralline algae cover between the two periods, which represents a relative decrease of 23%.

Comparison with earlier research. To our knowledge, there are no studies that have estimated the temporal trend of coralline algae cover in the Caribbean region. We were therefore unable to compare our results with those of previous studies.

Interpretation of temporal trend. The apparent absolute decline of 2.4 percentage points in coralline algae cover from 1980 to 2024 was likely driven by elevated estimates during the first five years of the time series, which averaged 10.3% (95% CI [4.4% - 13.2%]). These early estimated values were strongly influenced by a particularly high annual mean in 1982 (**Figure 1.3.4.D**), derived from observations at only five sites. From 1985 onwards, coralline algae cover remained relatively stable at ~8.3% (95% CI [4.1% - 10.1%]). Thus, the long-term decline we report is probably an artefact of limited early data, and coralline algae cover has likely remained stable from 1980 to 2024. This stability suggests that coralline algae cover has been neither strongly favored nor disadvantaged by anthropogenic drivers, unlike macroalgae and turf algae (**Figures 1.3.4.B-C**) or hard coral covers (**Figure 1.3.4.A**). However, it should be emphasized that percent cover is not an ideal metric for coralline algae, as they are often cryptic and predominantly grow in crevices (Kornder et al., 2021). Consequently, the stability reported here must be interpreted with caution, and highlights the need for further research to better characterize temporal changes in coralline algae under intensifying anthropogenic pressures in the Caribbean. For instance, Quéré et al. (2015) highlighted the risk of diseases affecting coralline algae, two syndromes having been identified in Curaçao.

Model performance. The temporal trend of benthic cover of coralline algae across Caribbean coral reefs from 1980 to 2024 (**Figure 1.3.4.D**) was estimated using data from 8,152 monitoring sites and 26,386 observations employed for model training. The model achieved a RMSE of 6.35, while accounting for 54% of the variance in the data (R^2 = 0.54).

3.2.5. Other fauna

Definition. The "other fauna" category encompassed all benthic animals excluding hard corals. The majority of species within this group belong to the Octocorallia (*e.g.*, gorgonians), Porifera (sponges), Actiniaria (sea anemones), and Polychaeta (*e.g.*, feather duster and serpulid worms).

Results. From 1980 to 2024, the long-term average in the modeled benthic cover of other fauna across coral reefs of the Caribbean was 10.4% (95% CI [9.3% - 11.5%]; **Figure 1.3.4.E**). Over this period, the long-term trend in other fauna cover exhibited a significant decline (Mann-Kendall test, τ = -0.31, p-value = 2.85 x 10-3), decreasing from an average of 11.2% (95% CI [9.6% - 12.8%]) in 1980–1985 to 10.1% (95% CI [9.4% - 10.9%]) in 2019–2024. This corresponds to an absolute decline of 1.1 percentage points in other fauna cover between the two periods. However, the other fauna cover remained stable at around 9.7% from 1998 to 2024.

Comparison with earlier research. To our knowledge, there are no studies that have estimated the dynamics of benthic cover of "Other fauna" in the Caribbean region, which did not make it possible to compare our results with those of previous studies.

Interpretation of temporal trend. Excluding the first two decades, for which data availability was limited, the cover of other fauna remained stable from 1998 to 2024 at approximately 9.7% (**Figure 1.3.4.E**). However, given the taxonomic diversity of this group, this apparent stability may conceal increases in some taxa while others may have experienced declines over the study period. Even within a single taxonomic group, benthic cover may respond in opposite directions following disturbances. For example, Gochfeld et al. (2020) documented an increase in encrusting sponge cover following a hurricane in the U.S. Virgin Islands, whereas upright sponge species experienced declines. Moreover, benthic cover, measured as the two-dimensional (2D) proportion of substrate occupied by a given category, may fail to capture critical changes in certain taxa. Indeed, erect organisms such as gorgonians and some sponges (*e.g., Xestospongia muta*) contribute substantially to coral reef three-dimensional structure, yet their importance is underestimated by two-dimensional benthic cover estimates (Kornder et al., 2021). These limitations highlight the need for further research to better understand temporal changes in non-scleractinian benthic groups under increasing anthropogenic pressures in the Caribbean.

Model performance. The temporal trend of benthic cover of other fauna across Caribbean coral reefs from 1980 to 2024 (**Figure 1.3.4.E**) was estimated using data from 13,625 monitoring sites and 35,817 observations employed for model training. The model achieved a RMSE of 5.36, while accounting for 69% of the variance in the data (R^2 = 0.69).

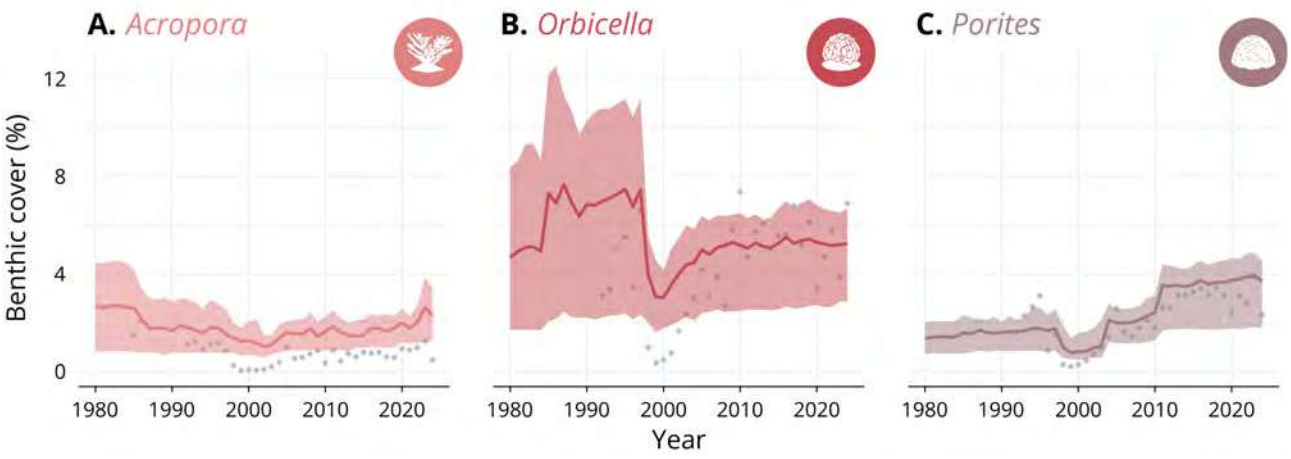
3.3. TRENDS IN HARD CORAL GENERA

3.3.1. *Acropora*

Definition. *Acropora* is a genus of Scleractinian hard corals represented in the Caribbean by *Acropora cervicornis* (staghorn coral), *A. palmata* (elkhorn coral), and *A. prolifera* (a stabilised hybrid of the first two) (Precht et al., 2019). *Acropora spp.* are the main branching hard corals species in the Caribbean, contributing disproportionately to reef structural complexity (McWilliam et al., 2018).

Results. From 1980 to 2024, the long-term average in the modeled benthic cover of *Acropora* across Caribbean coral reefs was 1.8% (95% CI [0.8% - 2.7%]; **Figure 1.3.5.A**). Over this period, *Acropora* cover showed no significant trend (Mann-Kendall test, τ = -0.164, p-value = 1.15 x 10-1), while two distinct periods can be distinguished. *Acropora* cover declined from 2.6% in 1980 to 1.0% in 2002, and then increased progressively in the following two decades to reach 2.3% in 2024.

Comparison with earlier research. The pattern of *Acropora* cover between 1980 and 2011 aligns with the trajectory documented by Jackson et al. (2014), though their estimates for 1980–1984 are somewhat higher (6% during that interval).



↑ **Figure 1.3.5.** Modeled temporal trends of benthic percentage cover for the hard coral genera *Acropora* (A), *Orbicella* (B), and *Porites* (C) in the Caribbean from 1980 to 2024. The bold lines indicate the modeled mean benthic cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

Interpretation of temporal trend. Previous studies have shown that *Acropora* cover had already declined prior to 1980, driven by anthropogenic pressures (Cramer et al., 2020). Jackson et al. (2014) estimated approximately 16% *Acropora* cover in 1975-1979 in the Caribbean. During the 1980s, *Acropora* cover decreased substantially, from 2.6% to 1.7% from 1980 to 1990 (**Figure 1.3.5.A**). The 1980s pronounced reduction has been attributed primarily to white-band disease, which spread extensively from the late 1970s through the early 1990s (Aronson and Precht, 2006; Gladfelter, 1982). For example, in Belize's Channel Cays, *A. cervicornis* cover collapsed from approximately 70% in 1986 to nearly 0% by 1997 (Aronson and Precht, 2001). The continued decline of *Acropora* cover until 2002 (**Figure 1.3.5.A**) was likely driven by the combined effects of the 1998 mass bleaching event and outbreaks of white pox disease, which exclusively affected *A. palmata* (Patterson et al., 2002). For instance, in Florida, *A. palmata* cover decreased by more than 70% between 1994 and 2000 as a result of white pox disease (Patterson et al., 2002). Several studies have emphasized the role of elevated SST in exacerbating coral disease prevalence across the region (Muller et al., 2008; Muller and van Woesik, 2014). Since 2003, *Acropora* cover has shown a progressive increase (**Figure 1.3.5.A**). The causes of this apparent recovery remain unclear and seem inconsistent with field observations. Indeed, recent surveys conducted across multiple Caribbean locations have reported the persistence of *A. palmata* and *A. cervicornis* at only a limited number of sites, generally at low benthic cover (Rodríguez-Martínez et al., 2014; Toth et al., 2022).

Model performance. The 1980-2024 temporal trend of *Acropora spp.* cover across the Caribbean (**Figure 1.3.5.A**) was estimated using data from 8,234 monitoring sites and 36,360 observations employed for model training. The model achieved a RMSE of 2.75, while accounting for 62% of the variance in the data (R^2 = 0.62).

3.3.2. *Orbicella*

Definition. The genus *Orbicella* (formerly the *Montastraea 'annularis'* species complex; Budd et al., 2012), comprises three Caribbean stony corals: *Orbicella annularis*, *O. faveolata*, and *O. franksi*. These massive species are among the primary Caribbean reef builders.

Results. From 1980 to 2024, the long-term average in the modeled benthic cover of *Orbicella* across coral reefs of the Caribbean was 5.5% (95% CI [2.3% - 7.9%]; **Figure 1.3.5.B**). Over this period, *Orbicella* cover showed no significant trend (Mann-Kendall test, τ = -0.0162, p-value = 8.83 x 10-1). Four distinct periods can nonetheless be identified. Between 1980 and 1997, *Orbicella* cover fluctuated considerably around an average of 6.5% (95% CI: [2.0%–10.4%]; **Figure 1.3.5.B**). A sharp decline occurred in 1998, with a loss of 7.4 percentage points, corresponding to a relative decrease of 58.9%. During the 2000s, *Orbicella* cover gradually recovered, before stabilizing from 2009 onward at around 5.2% (95% CI: [2.6%–6.6%]; **Figure 1.3.5.B**).

Comparison with earlier research. To our knowledge, no previous studies have quantified the temporal trend of *Orbicella* benthic cover at the scale of the Caribbean region. We were therefore unable to compare our results with those of previous studies.

Interpretation of temporal trend. The sharp decrease in *Orbicella* cover observed in 1998 coincides with the region-wide mass bleaching event that occurred that year (**Figure 1.3.5.B**). For instance, Roff et al. (2020) reported that *Orbicella annularis* suffered extensive mortality at Long Caye (Glovers Reef, Belize) following the 1998 bleaching, with benthic cover declining from 68% in June 1998 to 20% in June 2000. Similarly, in the Florida Keys, *Orbicella* cover decreased from approximately 4% to 3% after the 1998 event (Toth et al., 2022). Interestingly, our results did not indicate a decline following the 2005 bleaching event (Eakin et al., 2010) or the outbreak of white plague, contrary to what has been reported in the U.S. Virgin Islands (Miller et al., 2009) and Florida (Brandt and McManus, 2009). While our findings show a recovery in *Orbicella* benthic cover after 1998, pre-disturbance levels were never regained. Several studies have reported delayed reproductive impairment in *Orbicella* spp. following thermal stress (Levitan et al., 2014; Fisch et al., 2019), which may limit population recovery. Moreover, the sustained increase in turf algal cover since 2001 (**Figure 1.3.4.C**) raises concerns about ongoing competitive overgrowth, as turf algae can inhibit coral recruitment and tissue regeneration (Lirman, 2001; Vermeij et al., 2010).

Model performance. The temporal trend of benthic cover of *Orbicella spp.* across Caribbean coral reefs from 1980 to 2024 (**Figure 1.3.5.B**) was estimated using data from 8,290 monitoring sites and 37,075 observations employed for model training. The model achieved a RMSE of 4.93, while accounting for 77% of the variance in the data (R^2 = 0.77).

3.3.3. *Porites*

Definition. *Porites* is a genus of Scleractinian hard corals that includes two dominant species (*P. astreoides* and *P. porites*) and several species of lesser importance in the Caribbean, including *P. branneri, P. colonensis*, *P. divaricata,* and *P. furcata*.

Results. From 1980 to 2024, the long-term average in the modeled benthic cover of *Porites* across coral reefs of the Caribbean was 2.2% (95% CI [1.1% - 2.9%]; **Figure 1.3.5.C**). Over this period, the long-term trend in *Porites* cover exhibited a significant increase (Mann-Kendall test, τ = 0.69, p-value < 1.0 x 10-12), rising from an average of 1.4% (95% CI [0.7% - 2.1%]) in 1980–1985 to 3.7% (95% CI [1.7% - 4.7%]) in 2019–2024. This represents an absolute increase of 2.3 percentage points, equivalent to a relative increase of 164% in *Porites* cover between the two periods. However, this increase in the benthic cover of *Porites* was not continuous between 1980 and 2024, as it was punctuated by an abrupt decline in 1998 (-0.75 percentage points), followed by rapid increases in 2003 (1.1 percentage points) and 2010 (1.12 percentage points), interspersed with periods of relative stability.

Comparison with earlier research. To our knowledge, there are no studies that have estimated the temporal trend of benthic cover of *Porites* in the Caribbean region. We were therefore unable to compare our results with those of previous studies.

Interpretation of temporal trend. Our results reveal an increase in *Porites* cover on Caribbean coral reefs between 1980 and 2024. Within Caribbean hard corals, *Porites* has often been considered a potential "winner" under anthropogenic pressures (Green et al., 2008). However, more recent studies have nuanced this view, showing that reductions in average colony size and limited recruitment point to underlying demographic vulnerability (Edmunds et al., 2021).

Model performance. The temporal trend of benthic cover of *Porites spp.* across Caribbean coral reefs from 1980 to 2024 (**Figure 1.3.5.C**) was estimated using data from 8,290 monitoring sites and 37,075 observations employed for model training. The model achieved a RMSE of 3.06, while accounting for 54% of the variance in the data (R^2 = 0.54).

3.4. CONCLUSIONS

Our results indicate that Caribbean benthic coral reef communities have undergone profound changes between 1980 and 2024. Hard coral cover declined by half over this period, from 28.1% in 1980–1985 to 14.6% in 2019–2024 (**Figure 1.3.4.A**). Sharp decreases in hard coral cover coincided with region-wide bleaching events associated with thermal stress, particularly in 1998 (–9%), 2005 (–17.1%), and 2023–2024 (–16.9%). Given the limited amount of data available for 2024, the impact of the most recent event, the fourth global mass bleaching event (Reimer et al., 2024) that occurred in 2023–2024, may be underestimated in our analysis. Hard coral diseases such as white band disease and Stony Coral Tissue Loss Disease (SCTLD) have further contributed to the long-term decline in hard coral cover, although their effects have been more chronic and spatially variable.

The overall decline in hard coral cover observed over the study period conceals contrasting trajectories among the major hard coral genera in the region. Previous studies have shown that *Acropora*, which covered approximately 16% of the benthos in the 1970s, declined sharply before 1980 due to anthropogenic stressors and widespread outbreaks of white band disease (Cramer et al., 2020). Our results indicate that *Acropora* cover has remained low since 1980, averaging around 1.8%. *Orbicella* cover declined following the 1998 bleaching event and stabilized from 2009 onward at approximately 5%. In contrast, *Porites* cover increased by 164% between 1980 and 2024. These results reveal profound shifts in the composition of hard coral assemblages over the past four decades, characterized by the decline of branching species and the relative increase of more massive forms.

In contrast to hard corals, the benthic cover of macroalgae increased by 84.8% between 1980 and 2024 (**Figure 1.3.4.B**). Following a period of high variability during the first two decades of the study period, turf algal cover also increased, rising by 64.7% since 2001 (**Figure 1.3.4.C**). Two main drivers are likely responsible for these increases. First, the loss of herbivory, resulting from the decline of Scaridae and Acanthuridae fish populations

(Paddack et al., 2009), as well as the mass mortality of the sea urchin *Diadema antillarum* in 1983–1984 (Lessios, 1988) and again in 2022 (Hylkema et al., 2023). Second, the deterioration of water quality, manifested by elevated nutrient concentrations, sedimentation, and turbidity, has likely promoted the proliferation of both macroalgae and turf algae. Together, these drivers have simultaneously stimulated algal growth and reduced grazing pressure, reinforcing the dominance of macroalgae and turf algae on Caribbean coral reefs.

Finally, two additional benthic categories, coralline algae and other fauna, showed little change in their cover between 1980 and 2024 (**Figure 1.3.4.D-E**). However, benthic cover may not be the most appropriate metric for assessing these groups, as they include both cryptic species and species that develop primarily along a vertical axis. Moreover, the other fauna category encompasses organisms with highly diverse biological and ecological traits, which may have followed different trajectories over the study period. Therefore, further research is needed to better characterize long-term trends within these benthic groups and to understand the underlying drivers of any potential changes since 1980.

In summary, these results are consistent with earlier scientific studies reporting a shift from hard coral to macroalgal dominance on Caribbean coral reefs (Precht et al., 2020). They also align with observations of a decline in reef structural complexity, characterized by a transition from branching to more massive hard coral species (Alvarez-Filip et al., 2022). Such changes in the structure and composition of Caribbean coral reefs have likely already impacted, and will continue to affect, the ecosystem services they provide. For instance, the decline in hard coral cover and structural complexity will possibly reduce the abundance and diversity of reef-associated species, including fish and crustaceans that are important for food security and commercial fisheries. Furthermore, this degradation may compromise the capacity of Caribbean coral reefs to protect coastal areas from hurricanes, whose intensity is projected to increase under climate change (Knutson et al., 2010).

4. Recommendations

Recommendations from the previous report on the status of Caribbean coral reefs included the adoption of robust conservation and fisheries management strategies, simplification and standardization of monitoring protocols, fostering communication and exchange of information, and developing and implementing adaptive legislation and regulations. Despite some progress in protecting key species and reducing threats, significant priority actions are still required to secure a sustainable future for Caribbean reefs.

4.1. INTEGRATION OF CORAL REEFS INTO NATIONAL CLIMATE AND BIODIVERSITY STRATEGIES

Holistically addressing the triple planetary crisis of climate change, pollution, and biodiversity loss is essential. Global frameworks on Climate Change like the Paris Agreement (COP21) and the Aichi Biodiversity Target have established critical frameworks and targets for reduced greenhouse gas emissions and limits to global warming, but coral reef survival requires more ambitious targets, stronger ambition, more effective implementation strategies, and enhanced cooperation. The current trajectory of greenhouse gas concentrations needs a radical decrease for coral reefs to survive as we know them today, which is already a diminished version of the ecosystems prior to industrialization.

Coral reefs are among the ecosystems most vulnerable to climate impacts, as evidenced by the increased frequency and severity of mass bleaching events. The adverse effects of climate change are disproportionately born by Small Island Developing States (SIDS). SIDS have small but growing populations, limited resources, remoteness, increased vulnerability to natural disasters and other external shocks, excessive dependence on international trade, and fragile environments. Twenty nine out of 57 world SIDS countries/territories are located in the Caribbean, representing two-thirds of the countries/territories in the wider Caribbean region. Despite contributing less than 1% to global greenhouse gas emissions globally, SIDS face heightened vulnerability due to their limited resources and increased exposure to climate hazards, underscoring the inequitable nature of climate impacts (Mansoor and Inam, 2025). These impacts demonstrate that, despite the commitments of the Paris Agreement, global targets are not being met at the pace or scale required.

Translate Climate and Biodiversity frameworks into national plans

To safeguard reefs and their biodiversity, countries must not only reduce greenhouse gas emissions through global climate and biodiversity frameworks such as the United Nations Framework Convention on Climate Change (UNFCCC) and the Convention on Biological Diversity (CBD), but also ensure that coral reef protection is fully embedded into national strategies, including National Development Plans, Nationally Determined Contributions (NDCs), National Adaptation Plans (NAPs), National Biodiversity Strategies and Action Plans (NBSAPs), and National Biodiversity Finance Plans (BFPs). Coral reefs should be recognised as critical natural infrastructures and a nature-based solution-ecosystem that deliver climate mitigation, adaptation, and biodiversity benefits.

Coral reefs must be integrated into national and regional planning through measurable, science-based targets within NDCs, NAPs, and NBSAPs. This ensures countries commit to reef conservation while tracking progress and adapting to changing conditions. Stronger NDC implementation should link directly to coral reef outcomes, supported by climate finance for actions that deliver both mitigation and adaptation benefits. At the same time, biodiversity strategies should include updated management plans, monitoring systems, and clear restoration targets, aligned with ICRI's Key Policy Asks, to address accelerating pressures such as warming seas and ocean acidification.

However, given the relatively small emissions contribution by the Caribbean nations, real success in reducing greenhouse gas concentrations to levels that allow coral reefs to survive into the coming decades requires immediate changes and global action by the most developed nations.

Foster regional Blue Economy Strategies

Caribbean states require dedicated mechanisms to translate NDC, NAP, and NBSAP commitments into coral reef resilience projects. This includes developing regional blue economy strategies

that reduce drivers of coral reef degradation and provide economic valuations of ecosystem services to policy makers and investors. Such an enabling environment can include incentives for the financial sector to invest in the sustainable blue economy through de-risking, more resources to enforce existing regulation on poaching or pollution, and promotion of locally managed marine areas with engagement of fishery associations. Additional funding for coral reef resilience, restoration and monitoring could be unlocked through the development of reef insurance and biodiversity credits that channel resources to community-led restoration, pollution reduction, and sustainable blue economy ventures.

Strengthen regional cooperation for coordinated actions

Regional cooperation is also essential. Governance platforms such as the Caribbean Community (CARICOM), and the recently established Ocean Coordination Mechanism (OCM) provide opportunities for resource sharing, policy alignment, and coordinated action across the Wider Caribbean. Likewise, region-specific platforms including the Organisation of Eastern Caribbean States (OECS), the Central American Commission on Environment and Development (CCAD) and the Central American Organization for Fisheries and Aquaculture (OSPESCA), alongside funding mechanisms like the Caribbean Biodiversity Fund and Mesoamerican Reef Fund focus on specific countries for collaborative actions. By working collectively, states can optimize limited resources, avoid duplication, and enhance the overall effectiveness of climate and biodiversity strategies. Such coordinating bodies can be leveraged to address transboundary issues and help the observatory network related to pollution, migratory species, disease monitoring and mitigation, etc.

Ratify international conventions to reduce pollution and increase biodiversity protection

Regional environmental treaties remain critical tools for securing coral reef protection. The Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region (WCR) or Cartagena Convention is a regional legally binding agreement for the protection of the Caribbean Sea. Adopted in 1983, the Convention is supported by three technical agreements or Protocols on Oil Spills, Specially Protected Areas and Wildlife (SPAW) and Land Based Sources of Marine Pollution (LBS).

Other international initiatives such as ICRI, the Convention on Biological Diversity, the Ramsar Convention, and the Convention on Migratory Species, create collaborative frameworks that can drive greater ambition and accountability. Ensuring these commitments are fully implemented and adequately supported will be vital to maintaining coral reefs as a cornerstone of the climate–biodiversity nexus.

4.2. REDUCTION OF LOCAL THREATS

Reducing local pressures on biodiversity and coral reef ecosystems is crucial to ensuring their survival. While reducing local pressures on coral reefs is essential, many of the sectors responsible for these pressures, such as tourism, fisheries, agriculture, and waste management, face structural barriers that limit their ability to transition toward sustainability.

Reduce locally produced greenhouse gas emissions

The reduction of greenhouse gas emissions is urgent and essential for the preservation of coral reef ecosystems and their biodiversity. Countries and territories must transition to renewable energy sources, while stopping the use of fossil energy. The International Panel on Climate Change (IPCC, 2023) highlights mitigation options such as transitioning out of fossil fuels and investing in renewable energies. With abundant sun and wind, the Caribbean can reduce fossil fuel dependence through solar and wind energy, as Barbados plans with its 100% renewable electricity target by 2030 (Marshall and Koon, 2021). Geothermal projects, like Dominica's, offer additional clean power opportunities for some islands (Bhagaloo et al., 2023). Energy efficiency standards for buildings, appliances, and industries can further cut consumption. Climate-smart agriculture, agroforestry, and organic farming can help reduce emissions while boosting food security. Protecting carbon sinks such as seagrass, mangroves, and wetlands is also key as nature-based solutions for protecting or restoring ecosystems to mitigate the impact of climate change. In The Bahamas, mangrove restoration contributes to carbon storage and coastal protection (Hilmi et al., 2025).

Enhance biodiversity and enforce protection of vulnerable species

Protecting vulnerable species is critical to maintaining the ecological balance and resilience of Caribbean coral reefs. Herbivorous fishes such as parrotfish and surgeonfish, along with invertebrates like sea urchins, are essential for controlling algal overgrowth and sustaining reef health. Ecologically and economically important species such as queen conch and spiny lobster, as well as mesopredator fish like groupers, are also under significant pressure from overexploitation and habitat loss. A range of management measures can be applied to safeguard these species, including seasonal closures, catch size and quota limits, gear restrictions, and the creation of fully-protected replenishment zones. Targeted bans, such as the prohibition on parrotfish harvests in Belize, have demonstrated success in restoring fish populations and supporting reef recovery (Mumby et al., 2021). Critically endangered goliath groupers, nassau groupers, sawfish and a number of sharks are just a few of the species in need of regulatory protection, more aligned with their IUCN Red List status.

Complementary actions such as community-led monitoring, science-based harvest regulations, and stronger enforcement capacity are essential to ensure compliance and long-term effectiveness. The SPAW Protocol, endorsed by 19 countries, provides a regional legal framework to reinforce these efforts, requiring countries to regulate or prohibit activities affecting listed species. By aligning national policies with SPAW commitments, governments can strengthen the protection of vulnerable reef-associated species.

Ensure sustainable and regulated coastal developments

Unregulated coastal development threatens environmental sustainability, local communities, and economic resilience in the Caribbean. Governments should adopt Integrated Coastal Zone Management (ICZM) to balance economic growth with ecosystem protection and the equitable distribution of resources. Policies must regulate land use and minimize the negative impacts of development on fragile coastal ecosystems, requiring environmental impact assessments for new projects. Developers should apply mitigation measures to prevent habitat loss, pollution, and coastal erosion, and integrate climate-adaptive design (elevated structures, storm-resistant materials, and nature based solutions such as strict mangrove protection) to address hurricanes, flooding, and sea-level rise. Governments should enforce zoning and land-use planning regulations that limit high-density developments in ecologically sensitive areas.

Improve solid waste management and reduce plastic pollution

Effective waste management is vital for sustainable Caribbean coastal development, yet many nations face landfill overuse and methane emissions contributing to climate change. Solutions include investing in waste-to-energy projects, expanding recycling, and promoting circular economy initiatives, as seen in Trinidad and Tobago's landfill gas-to-energy efforts (Charles, 2022). Tackling plastic pollution is equally important. Governments should ban or restrict single-use plastics, incentivize biodegradable alternatives, and run public awareness campaigns. Strengthening waste collection and sorting systems and recycling infrastructure can prevent plastics from reaching the ocean (Diez et al., 2019), thus protecting marine ecosystems.

Improve wastewater treatment

Preventing chemical pollution and eutrophication in Caribbean coastal waters requires adequate wastewater treatment and management. Hotels, marinas, and coastal communities must not discharge poorly treated effluents, causing algal blooms, oxygen depletion, disease outbreaks, and biodiversity loss. Regulators should mandate modern treatment facilities that remove harmful chemicals, nutrients, sediments, and pathogens before discharge. Nature-based systems (constructed wetlands, biofilters), and water recycling reuse can further reduce loads while supporting biodiversity. The tourism sector should adopt advanced treatment systems and monitor effluents, while public awareness and incentives can encourage sustainable practices.

Reduce risk of invasions through strengthened aquarium trade regulations

Unregulated marine aquarium trade can introduce non-native species to Caribbean waters when unwanted pets or surplus stock are released, or when hitchhiking organisms and pathogens arrive hidden in shipments, sometimes leading to invasive populations that damage coral reefs. To reduce

this risk, regulations should tighten controls on the import and export of live coral reef organisms, require quarantine and health screening, and mandate best-practice handling to minimize escape or release. Clear "white lists" of low-risk, approved species, combined with licensing of traders and meaningful penalties for illegal releases, can further limit introduction pathways. Complemented by education campaigns for hobbyists and retailers about responsible pet ownership and the impacts of invasive species, strengthened aquarium trade governance can significantly lower invasion pressure on already stressed Caribbean coral reefs.

Promote sustainable fishing

Sustainably managed fisheries help preserve Caribbean coral reefs by maintaining herbivory and the viability of fish assemblages that support ecological functioning of the reef system. Local fisheries management on 12 islands increased juvenile coral density by 62% by protecting herbivorous fish that control algae (Steneck et al., 2018). Key strategies include seasonal closures (*i.e.* spiny lobster in Cuba and The Bahamas and queen conch in Antigua, Mexico, and Belize) to support reproduction and sustain catches (Muñoz-Nuñez, 2009; Sabido-Itza et al., 2021), size and catch limits to prevent overharvesting of juveniles, herbivores protection (*i.e.* parrotfish listed under SPAW Annex III are banned from fishing in the Dominican Republic, Bonaire and the entire Mesoamerican Reef) to maintain healthier reefs (Mumby et al., 2021). Mesopredator fish protection (groupers and snappers) is also essential to maintain ecological balance. Strongly enforced and long-term protection measures in the Cayman Islands and U.S. Virgin Islands led to increase in biomass, fish size and reproductive capacity of Nassau grouper and Red Hind, respectively (Stock et al., 2021; Rosemund et al., 2022). Reef fish consumption should be greatly reduced, especially of parrotfish, to maintain their grazing activity. Finally, fishers need transitional support to engage in alternative livelihoods while fishing pressure is reduced through a variety of regulatory mechanisms, including fully protected MPAs. Community engagement, participation, decision-making and outreach help build compliance and support for sustainable practices.

Promote sustainable tourism

As a key economic sector, tourism should shift toward low-impact models: ecotourism standards, green certification programs for accommodations, energy efficiency programs, and carbon offset strategies. In Saint Lucia, the government has supported initiatives that encourage sustainable tourism practices, including energy efficiency programs for resorts and the protection of natural ecosystems that attract visitors.

A transition to a more sustainable form of tourism can be accompanied by sustainable tourism global standards and certifications providers (*e.g.*, Global Sustainable Tourism Council, Blue Flag, Green Key, Green Globe, Green Destinations, Eco Certified Tourism, Travel Life, Earth Check, Reef Check, Green Fins, etc.) (Leotaud et al., 2024). An example of a responsible tourism model is being led by the Wave of Change, Iberostar Group in the Mexican Caribbean, Jamaica, and Dominican Republic which includes circular economy, nature-based solutions, decarbonization strategies, sustainable foods and destination stewardship (Calle-Triviño et al. 2025). This sector relies on coastal development planning, and links to improved wastewater treatment and solid waste management, and support for sustainable fisheries regulations.

Promote blended-finance mechanisms

Blended finance approaches can leverage public and private funds to identify and invest in solutions that can support coral reef resilience as well as create employment opportunities. For example, to address local stressors (mass tourism, intensive agriculture, sewage pollution), the Global Fund for Coral Reefs-funded MAR+Invest programme was launched in 2022 as a finance and development facility for the Mesoamerican Reef region. Led by MAR Fund, and with regional partners (Mexican Fund for the Conservation of Nature, Sureste Sostenible, Healthy Reefs for Healthy People, New Ventures, and VIWALA), MAR+Invest aims to foster the development and scaling of enterprises or projects with a positive impact on coral reefs. These initiatives support businesses that reduce drivers of degradation impacting coral reefs (*e.g.* investing in waste water treatment solutions). The GFCR Equity Fund has also invested in Carbonwave, the world's leading developer of ultra-

regenerative, plant-based, advanced biomaterials from *Sargassum*. The company transforms this seaweed (a resource long considered waste, which does not need land or other inputs to grow) into biomaterials that can replace petroleum products and other harmful industrial inputs.

4.3. IMPLEMENTATION AND ENFORCEMENT OF AREA-BASED MANAGEMENT TOOLS

Area-based Management Tools (ABMT), including Marine Protected Areas (MPAs), Locally Managed Marine Areas (LMMA), and Other Effective Area-Based Conservation Measures (OECMs), are well-established as tools for the management, protection, and conservation of marine and coastal resources, including coral reefs. Effectively managed ABMT, especially MPAs, with strong planning, participatory community engagement, equitable governance, and robust Monitoring, Control and Surveillance (MCS) measures, exhibit critical benefits for biodiversity and communities, by enabling stricter management of local pressures including marine tourism, anchoring, and extractive activities such as fisheries and marine dredging.

In 1966, 1,307 protected areas covering 1.16 million km² had been established in the Wider Caribbean Region, with 20% (~261) containing marine and coastal resources within their boundaries (Status of Protected Area Systems in the Wider Caribbean Region (1966)). By 2008, over 700 MPAs had been established, covering approximately 300,000 km², which is equivalent to 1.5% of the region's coastal and shelf waters. Of these established MPAs, 571 allowed for extractive activities (51,150 km²), 87 MPAs (protecting 236,853 km²) were zoned, enabling a mix of extractive and non-extractive activities, whilst 98 were considered no-take marine reserves, restricting all extractive activities throughout the area (Guarderas et al., 2008). In 2024, 7.85% (517,150 km²) of the total marine area (6,588,658 km²) of the Wider Caribbean Region (excluding Bermuda) was protected by 846 MPAs—a marked increase since 1996 (Gottlieb et al., 2024). Throughout this period, however, coral reef systems and associated biodiversity and fish populations have experienced rapid declines, with MPAs only partially effective in achieving their biodiversity and conservation objectives, due to weak enforcement and compliance (Pike et al., 2024; Gottlieb et al., 2024).

Strengthen regional efforts to achieve international targets in the Caribbean

In December 2022, Parties to the Convention on Biological Diversity (CBD), at the occasion of the 15th Conference of the Parties (COP15), adopted the Kunming-Montreal Global Biodiversity Framework (GBF, 2022). Among its 23 targets, Target 3, or the "30x30" target, calls to conserve 30% of terrestrial, freshwater, coastal, and marine areas by 2030. This target builds on Aichi Biodiversity Targets (2010-2020), specifically Target 11, to protect, conserve and effectively manage 10% of marine and coastal areas, as transcends regional initiatives including the Caribbean Challenge Initiative (CCI) 20x20 target to conserve and protect at least 20% of nearshore marine and coastal environments by 2020. The 30x30 framework aims to establish ecologically representative, well-connected, and effectively managed areas that safeguard biodiversity and provide essential services like carbon storage, water filtration, and coastal protection.

Although the number of MPAs, and the total area of protection has increased over time throughout the Caribbean, protected area coverage still needs to be expanded, consolidated, and further assessed to understand their effectiveness in conserving and enhancing biodiversity.

Other effective area-based Conservation Measures (OECMs) provide an important way for governments and non-state actors to contribute to the 30x30 goal by identifying and supporting sites and territories that deliver effective and equitable conservation outcomes (Jonas et al., 2024). However, their identification and designation across the region, alongside the establishment of MPAs, with a view to achieving Target 3 of the GBF, has yet to be realised.

Greater support is needed across the Caribbean to aid countries in embedding ABMTs, including MPAs and OECMs, into their national policies and conservation targets through updated NBSAPs. These instruments form the foundation for identifying priority coastal and marine areas, guiding biodiversity-inclusive spatial planning, and translating global commitments into actionable national strategies. By aligning national policies with global targets such as 30x30, NBSAPs and Sustainable Ocean Plans (SOPs), where developed, can enhance policy coherence and foster stronger collaboration among government agencies, scientific institutions, and stakeholders. Importantly, they promote an integrated approach

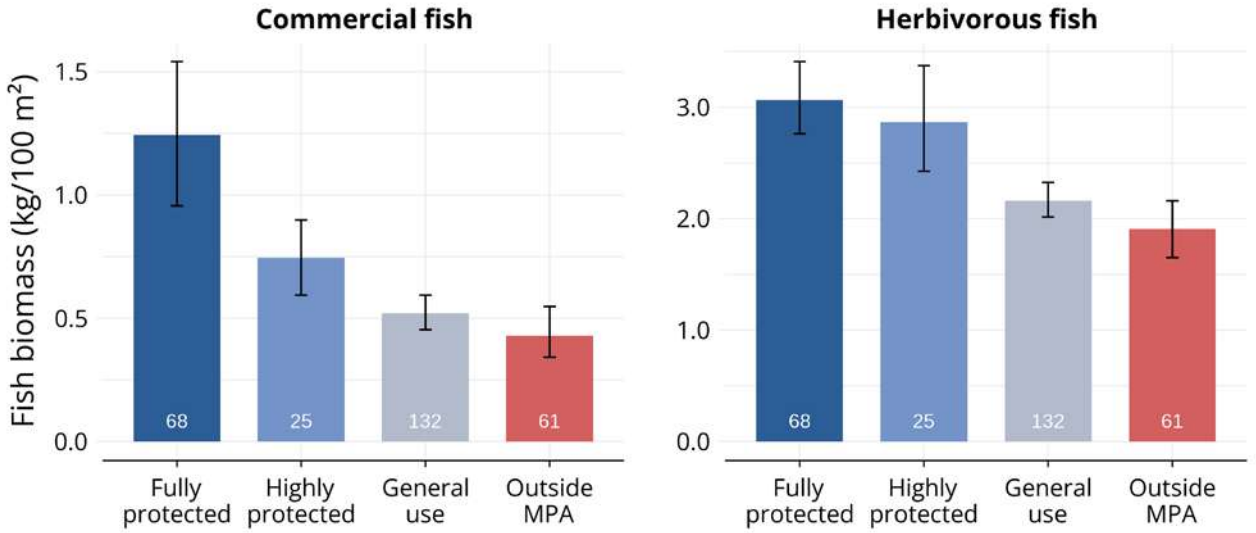
to conservation that improves coordination between land and ocean management and helps mobilise the financial and technical resources needed to deliver long-term biodiversity outcomes. In addition to strong and more representative national policies and targets, achieving Target 3 of the GBF requires strengthened support for protected areas through integrated management (strategic planning, budgeting), enforcement, community involvement, sustainable financing and capacity-building for marine protected area managers and practitioners.

Enhance fisheries and biodiversity through stronger protection levels

ABMTs, such as MPAs, enhance biodiversity and ecosystem function by safeguarding critical species, habitats, and ecological processes. Fully protected areas prohibit all fishing and extraction, whereas highly protected areas allow limited, regulated activities such as catch-and-release fishing, seasonal restrictions, or tightly controlled permits. General use areas are designated MPA where regulated extractive activities are allowed. These levels of protection strongly influence conservation outcomes—particularly for reef fish communities, as shown below with the fully protected zones having significantly higher biomass than general use areas or those outside MPAs, as seen in **Figure 1.4.1.**

By eliminating or reducing fishing pressure, MPAs increase fish diversity, abundance, biomass, and size structure, which in turn strengthens ecological roles such as successful reproduction, herbivory, and predation. Increases in herbivorous fish reduce macroalgal cover and improve coral recovery processes (Topor et al., 2019). In the U.S. Virgin Islands, for example, the Red Hind Bank Marine Conservation District—a permanent no-take reserve designated in 1999—has resulted in larger fish and healthier spawning aggregations (Rosemond et al., 2022). MPAs also promote population growth and fisheries spillover of commercially valuable species such as lobster and queen conch, as documented in Florida's Dry Tortugas National Park (Bertelsen and Cox, 2001; Spanier, 2024).

At broader scales, MPAs help maintain coral reef resilience. While global coral cover has declined, reefs inside MPAs—especially older, well-enforced no-take areas—have remained comparatively stable (Selig and Bruno, 2010). Long-term protection at sites such as Cuevones and Xcalak Reef National Park in Mexico has yielded higher hard coral cover, greater fish biomass and abundance, and reduced macroalgae relative to nearby unprotected reefs (Díaz-Osorio et al., 2022; Perera-Valderrama et al., 2016). Recovery after major disturbances, such as bleaching events and hurricanes, is also greater in fully protected zones.



↑ **Figure 1.4.1.** Comparison of commercial and herbivorous fish biomass in the Mesoamerican Reef (Mexico, Belize, Guatemala, and Honduras) across fully protected, highly protected, and general-use marine protected areas (MPAs), as well as in areas outside MPAs. Commercial fish are indicated by the grouper and snapper families, exhibiting the highest fishing pressure. Herbivorous fish (parrotfish and surgeonfish) are protected throughout the entire Mesoamerican Reef, so variations in zoning are mainly due to enhanced enforcement. Numbers displayed on the bars indicate the number of monitoring sites used to estimate fish biomass. Figure adapted with permission from the 2024 *Mesoamerican Reef Report Card* of the Healthy Reef Initiative (McField et al., 2024).

However, legal protection alone is insufficient. Effectiveness depends on enforcement strength, stakeholder participation, habitat quality, connectivity, and the proportion of fully-protected zones (Edgar et al., 2014; Di Cintio et al., 2023). Sustained compliance, supported through communication, monitoring, and local engagement, remains a critical determinant of success (Bergseth et al., 2024; Di Cintio et al., 2023).

Promote connectivity and integrated management

Adaptive MPA management plans are essential for translating protected-area designation into measurable conservation outcomes by defining clear objectives, zoning, threat-reduction actions, monitoring of ecological and social impacts and tracking of resource allocations and gaps (Leverington et al., 2010). In the Caribbean, where coral reefs, mangroves, and seagrasses are highly connected and climate-vulnerable, site-level plans that integrate adaptive management and connectivity are crucial to maintain ecological function and species movements (Pittman et al., 2014). Gottlieb et al (2024) found 89% of MPAs were well-connected across the Caribbean, with coral reefs being a highly represented habitat. However, relatively good ecological coherence was seen only in the Western and Southern Caribbean. Robust, science-based management plans enable better targeting of financing and monitoring, supporting adaptive management, and improving the long-term effectiveness and resilience of the marine protected areas (Coad et al., 2015). Finally, many Caribbean marine protected areas risk remaining "paper parks" without effective management plan implementation, adequate resources, and long-term planning (Bustamante et al., 2014).

Ensure effective enforcement

Effective enforcement is a cornerstone for ensuring that marine protected areas in the Caribbean fulfill their conservation objectives (Kaplan et al., 2015). Weak and under-resourced MCS, limited sanctions and unclear legal authority continue to undermine many MPAs in the region (Environmental Law Institute, 2015). Studies show that MPAs which incorporate stronger compliance-tools (incentives + penalties) are associated with higher biomass and densities of key species (Kaplan et al., 2015). Furthermore, robust enforcement frameworks enhance legitimacy among local stakeholders and strengthen governance, which in turn supports participation and long-term success. Without consistent patrolling, inter-agency coordination and legal backing, MPAs risk failing to protect biodiversity and coral reef ecosystems or support coastal livelihoods (Environmental Law Institute, 2015). In the Caribbean context, capacity-building in enforcement (trained personnel, equipment, legal infrastructure) is equally as important as good design in the delivery of real ecological and social benefits. However, even the best-designed management plans and enforcement frameworks cannot be sustained without reliable funding, as financial resources are essential to support staffing, equipment, monitoring, and long-term operations of Caribbean MPAs.

Enable sustainable funding for protected areas

While MPA areas are increasing, the majority are underfinanced and under-resourced, effectively making them "paper parks" with little to no activities around enforcement and compliance (Rife et al., 2013). Government funding is currently the main source of funding for ABMT management and should be secure and sustainable. However additional financial solutions can be implemented, including co-management by NGOs and sustainable blended finance that leverages public and private finance sources. For example, the Global Fund for Coral Reefs has supported blended finance approaches for coral reef resilience in the Mesoamerican Reef region, The Bahamas and Colombia, including facilitating finance to fishery associations in Colombia in exchange for agreements on fully-protected MPA zones, development of co-management models for MPAs, and reef insurance and blue carbon models that can provide funding for conservation and restoration work. Critical to well managed MPAs is having strong partnerships and collaboration across government, NGOs, the private sector and local communities.

Support community engagement and educational programs

Community engagement and education are key components for the long-term success of Caribbean marine protected areas, as they foster compliance, stewardship, and local support for conservation objectives (Gardner, 2009; Pollnac et al., 2010). Empowering local communities through capacity-strengthening initiatives—such as educational programs, economic

incentives, equity and inclusion measures, formal certifications, and local leadership—enhances support for conservation activities, including coral reef restoration, as demonstrated in the Seaflower MPA in Colombia (Uribe-Castañeda et al., 2024). Outreach programs, including school activities, public meetings, and media campaigns, increase awareness of MPA goals and regulations, while participatory approaches enable communities to contribute to decision-making, monitoring, and management (McConney et al., 2015; Christie et al., 2003). Studies indicate that MPAs with higher levels of local engagement and education achieve better ecological outcomes (Pollnac et al., 2010). Integrating public engagement strategies into management plans strengthens governance and ensures that conservation measures align with socio-economic realities, enhancing both legitimacy and resilience of Caribbean MPAs (Gardner, 2009; Christie et al., 2003).

To translate community engagement and educational initiatives into tangible conservation outcomes, MPA managers themselves require targeted capacity-building, including training in participatory approaches, conflict resolution, communication strategies, and adaptive management.

Strengthen capacity through cooperation networks

Strengthening capacity for ABMT managers is essential to ensure the effective and sustainable management of marine ecosystems in the Caribbean. Studies show that ongoing training and the development of skills in adaptive management, communication, and participatory governance strengthen the resilience of MPAs to environmental and socio-economic pressures (Gombos et al., 2021).

Strengthening cooperation networks among protected area managers is also a key lever of action: building their capacities, sharing best practices through peer-to-peer exchanges, and supporting initiatives will help increase the resilience of the marine socio-ecological system (Besançon and Mudford., 2023).

The Wider Caribbean Region is benefiting from a growing momentum in networking among protected area managers. These platforms promote experience sharing, conservation coordination, and capacity building. In 2024, the SPAW-RAC / UNEP-CEP launched a regional coordination process to strengthen synergies, improve technical coherence, and enhance multi-level governance among the main networks of protected area managers: the SPAW Protected area manager network, MPA Connect, NAMPAM, RedGolfo, DCNA, the Mesoamerican network of MarFund, SusGren, and RedParques. The CAMPAM network, although inactive since 2020, played a key role for two decades in establishing long-term training programs for marine protected area managers (Bustamante et al., 2018).

Caribbean nations are encouraged to expand ecologically connected ABMT with well-enforced fully-protected zones and seasonal closures, ensure inclusive stakeholder engagement, particularly with local fishers, tourism operators, and indigenous communities to secure sustainable financing, invest in monitoring and adaptive management, and actively support the development and strengthening of MPA networks to secure long-term ecological, social, and economic benefits.

4.4. DEVELOPMENT AND MAINTENANCE OF MONITORING PROGRAMS

Improve spatial representativeness

Coral reef benthic cover monitoring is unevenly distributed across the countries and territories of the Caribbean (**Figure 1.3.3**). United States territories, including Florida, the Flower Garden Banks, Puerto Rico, and the United States Virgin Islands, accounted for 58.9% of all surveys conducted in the region between 1970 and 2024, despite representing 7.4% of the total coral reef area in the Caribbean. In contrast, Cuba, which encompasses 20.3% of the region's coral reef extent (**Table 1.1.1**), contributed only 0.9% of the total number of surveys conducted during the same period (**Table 1.3.1**). These results indicate that coral reefs across Caribbean countries and territories are not monitored in proportion to their spatial extent. Instead, factors such as Gross Domestic Product (GDP), reef remoteness, and proximity to research facilities are likely to influence the intensity of benthic monitoring efforts. However, it is important to note that these figures may not fully reflect the actual data collected by countries and territories in the region. Some datasets may have limited visibility due to factors such as lack of indexing in online repositories or reduced

accessibility, for instance because of language barriers or the absence of open-access data, and were therefore not included in this analysis.

Several solutions can be considered to remedy, at least partially, this problem of spatial representativeness. Firstly, funding organizations must be encouraged to direct part of their funding towards implementation of monitoring programs in the accessible areas where little monitoring has been carried out so far. Secondly, in inhabited areas far from research centres, citizen science may contribute to alleviate the spatial gaps. Scientists or managers can be assisted by citizens with user-friendly protocols for taking photos or videos, which can then be analysed using artificial intelligence tools, such as ReefCloud, to extract the percentage cover of benthic categories. The inclusion of local organisations and community researchers in monitoring their reefs empowers the custodians of marine resources.

Maintain monitoring over time

Investing in long-term reef monitoring is crucial for the preservation of Caribbean reefs, as it provides continuous data on ecosystem structure, composition, biodiversity, and the impacts of climate change. However, monitoring must go beyond data collection to directly inform management decisions. Effective programs should be designed around clear management questions, linking indicators to defined thresholds or triggers that can prompt timely action. For example, monitoring coral cover, fish populations, or water quality should be tied to thresholds that, when crossed, signal the need for intervention, such as adjusting fishing regulations, strengthening MPA enforcement, or scaling up restoration measures. Decision-oriented monitoring should also rely on standardized methods to ensure comparability across sites, and include robust approaches such as power analysis to detect meaningful changes within relevant timeframes. Independent monitoring, particularly within MPAs, remains essential for ensuring reliable assessments of protection effectiveness. At present, high-quality and decision-relevant data on MPAs and reef health remain limited, and expanding such efforts is vital to improve adaptive management and safeguard the resilience of coral reef ecosystems. Collaborative efforts are underway, like the Healthy Reefs Initiative in the Mesoamerican Reef which has an almost 20 year program of standardized reef health monitoring linking data directly to management and policy through the >70 partner organizations and popular Mesoamerican Reef Report Cards produced every other year.

Monitor additional indicators

Long-term monitoring programs are increasingly vital in the Caribbean, not only to assess the effectiveness of conservation initiatives but also to guide timely responses to emerging threats. Monitoring should extend beyond hard coral and algal benthic cover to include additional indicators such as the presence and spread of invasive species, the prevalence and severity of coral diseases, coral recruitment, the abundance of key functional fish groups, and reef structural complexity (all included in the AGRRA methodology). Water quality is also needed throughout the region on a variety of spatial and temporal scales (remote sensing to laboratory analysis). These indicators are directly linked to management decisions. For example, tracking invasive species can inform the timing and scale of targeted removal efforts. A key challenge is not only to collect these data but to establish clear thresholds or triggers that signal when management action is necessary. Standardizing these approaches across the region and building capacity for trained observers are essential to ensure timely detection and effective responses. Leveraging new technologies such as remote sensing, automated imaging, or environmental DNA sampling can also help fill data gaps, particularly for remote or less accessible reefs, ensuring that managers have actionable information to protect reef health. Collecting socio-economic data is also important to study the link between resilient reefs and resilient human communities and provide the information to garner support from policy makers, investors and development actors.

Standardize fish monitoring

Coral reef health is intrinsically linked to the structure and function of fish communities, making standardized monitoring of key groups especially carnivores and herbivores is essential, such as in the AGRRA methodology. However, reef fish data have not been collected uniformly or consistently enough across the entire Caribbean to be implemented in the present report. It is crucial that fish populations are monitored with uniform protocols, to prevent data inconsistencies, and allow for robust regional and long-term comparisons. Standardized monitoring enables accurate assessment of population trends,

evaluation of MPAs effectiveness, fisheries regulations, and adaptive management to strengthen reef resilience. The GCRMN Caribbean Monitoring Guidelines (2018) detail recommended methods for three different capacity levels.

Make data findable, accessible, interoperable and reusable

Greater re-use of coral reef monitoring data will ensure robust knowledge to inform science-based policy and decision making for coral reefs now and in the future. In order to be able to better understand the status and trends of coral reefs, the coral reef conservation community needs to improve the way we integrate major heterogenous, individual datasets. This report sought to improve those processes as a result of lessons learned from Souter et al. (2021) and as part of the GCRMN Data Task Force initiative. We also recommend using approaches to enhance coral reef data management, attribution, sharing, access, and integration such as through the use of platforms like ReefCloud, MERMAID, AGRRA etc. Lastly, we recommend the datasets to be structured in accordance with existing global standards (*e.g.* Darwin Core), and the development of analytical tools to allow straightforward and comprehensive data analyses.

4.5. REEF RESTORATION

Reef restoration across the Caribbean has moved beyond small-scale pilot projects and is now demonstrating measurable ecological benefits and methodological innovation. Key restoration techniques include enhancing sexual reproduction, genetic diversity and thermotolerance, and improving key ecological processes that increase early-stage coral survival to strengthen ecosystem resilience, but also artificial reef deployment and herbivore enhancement. The Latin America/Caribbean regional group of the Coral Restoration Consortium aims to restore coral reefs, enhance biodiversity, and build local capacity for long-term reef conservation, through outplanting and propagation of coral fragments, regional restoration planning, engaging communities in conservation, conducting research and monitoring, advocating for supportive policies, and developing resilience strategies against climate change. While restoring reefs to identical functional status does not seem feasible, targeted actions can help restore specific functions, given appropriate mechanisms and budget to allow for multi-year monitoring, maintenance, and reporting.

Coral gardening and transplantation

Coral gardening has become the backbone of reef restoration in the Caribbean, with nurseries supporting both fast-growing *Acropora* spp. and, more recently, massive corals fragmented and cemented onto reefs (Hein et al., 2020). Larval propagation further diversifies restoration stock by increasing genetic variation. Building on these advances, we recommend expanding nursery programs, prioritizing multi-species and multi-genotype approaches, and integrating larval propagation to enhance the resilience of restored reefs. We recommend that techniques are validated locally before restoration work, especially for slow-growing massive coral species, and that corals are planted in healthy habitats with good water quality, abundant herbivore populations and ecologically balanced fish populations.

↑ **Figure 1.4.2.** Presentation of the restoration research programme, UNAM, Puerto Morelos Mexico, October 2024 (credit: A. Petit).

Artificial reefs

Artificial reefs, including shipwrecks, custom structures, and concrete modules, are widely used in the Caribbean for tourism, research, and restoration (Baine, 2001; Hylkema et al., 2021). While they provide habitat and substrate for coral recruitment, they also risk concentrating fishing pressure by acting as fish aggregation devices (Grossman et al., 1997). We recommend that artificial reef deployment be aligned with marine spatial planning, monitored for ecological outcomes, and preferentially placed in no-take zones to ensure they enhance, rather than undermine, restoration goals.

Herbivore enhancement

Herbivore restoration has advanced considerably, particularly for the sea urchin *Diadema antillarum*, whose populations collapsed in the 1980s (Lessios, 2016). Techniques now include larval collection with settlement collectors and head-starting in labs, as well as assisted natural recovery by leaving settlements collectors in-situ (Williams, 2016, 2022; Hylkema et. al. 2022a). Consistent small-scale methods have also been developed to culture *D. antillarum* from gametes (Pilnick et al. 2021, Wijers et al. 2023). Alternative invertebrate grazers, including the Caribbean king crab *Maguimithrax spinosissimus* and the West Indian sea egg *Tripneustes ventricosus* (Butler et al. 2024), have also shown strong potential for reducing algal cover and boosting coral recruitment (Spadaro and Butler, 2021, McField et al., 2024). We recommend investing in the upscaling of *Diadema* culture and post-release survival (De Breuyn et al. 2023, Wijers et al. 2024), while expanding trials with other grazers to diversify herbivore enhancement strategies (McField et al., 2024, McField et al., 2022).

Coral resilience enhancement

Recent advances in the Dominican Republic and Mexico highlight the integration of applied science and restoration to address compounding reef stressors. Standardized heat stress assays (CBASS) identify genotype-specific thermal thresholds, enabling selection of thermotolerant corals (Voolstra et al., 2020; Blanco-Pimentel et al., 2025, Miller et al., 2024). At the same time, ex-situ protocols have achieved >90% success in halting lesions caused by Stony Coral Tissue Loss Disease (SCTLD) (Pelose et al., 2024). We recommend integrating thermal tolerance screening with disease mitigation into restoration planning to safeguard genetic diversity and build more resilient coral populations across the region.

Engage with the tourism sector in reef restoration

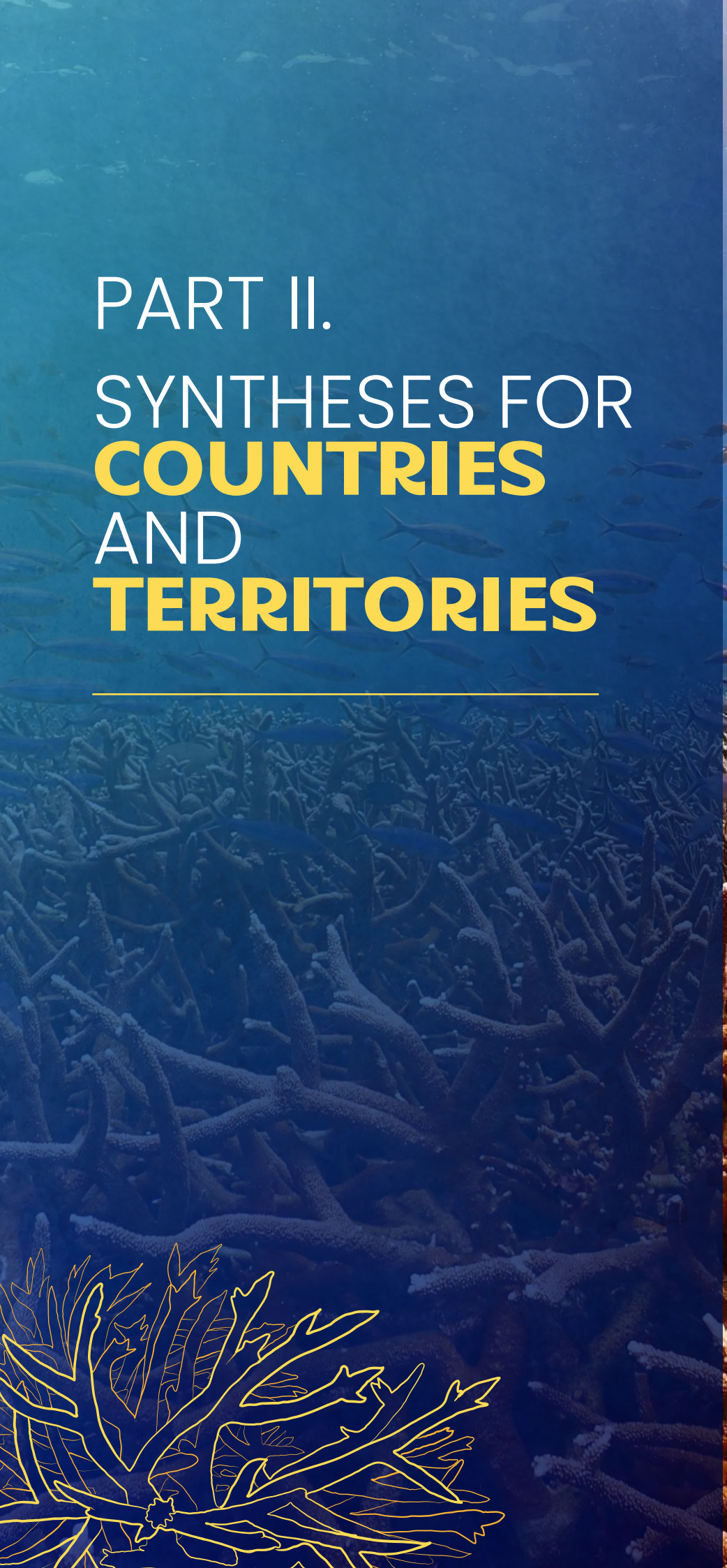
Coral reef conservation can be further enhanced through the engagement of the tourism sector. Both the tourism economy and marine ecosystems can benefit from active partnerships between governments, NGOs and the tourism sector. Tourism staff capacity should be improved on the technical aspects of reef restoration, and targeted communication campaigns created (Escovar-Fadul et al., 2022). Further, Restoration as a Services (RaaS) is a developing product being piloted by restoration companies like Coral Vita in The Bahamas. The concept is that restoration provides value in the form of enhanced coastal protection, tourism through a more vibrant habitat, and more resilient fisheries - and thus, there is value in the private and public sector paying for restoration.

Restoration benefits and pathways forward

Caribbean restoration initiatives are already showing measurable increases in hard coral cover, fish biomass, and ecosystem functionality, while also supporting tourism, education, and community livelihoods (Hein et al., 2019, 2020; Calle-Triviño et al., 2025). Partnerships between NGOs, governments, and the tourism sector, especially in Mexico and the Dominican Republic, are catalyzing these successes (Blanco-Pimentel et al., 2022; Uribe-Castañeda et al., 2024). We recommend scaling such partnerships, standardizing monitoring protocols, investing in scalable cost-effective solutions and ensuring long-term financing, like the parametric reef insurance. Strengthening the communication between coral restoration practitioners, scientists and local communities can help overcome these barriers by promoting knowledge sharing, standardization of methods, and building on the collective experience of all involved (Boström-Einarsson et al., 2020). Coral restoration projects need to be climate smart and take climate change into account (*e.g.* Fragments of Hope, Belize) and be part of a continuum of management strategies such as reduction of local stressors (Hein et al., 2020). A more coordinated approach could offer a more promising path for the role of restoration in coral reef conservation in the future.

PART II.

SYNTHESES FOR **COUNTRIES** AND **TERRITORIES**



Healthy *Acropora cervicornis* community © Mark Vermeij (CARMABI)

CHAPTER 1

Anguilla

Authors: Jasmin Ruan, Laëtitia Mathon.
Data contributors: Jasmin Ruan.

INTRODUCTION

Anguilla, a United Kingdom Overseas territory, is the most northerly island of the Lesser Antilles Arc at 18°35'N, and is an uplifted fossil coral reef surrounded by fringing and patch reefs as well as offshore cays (Bouchon et al., 2008). The Territory comprises the main island (25km-long, and 5km wide), along with several smaller islets including Dog, Scrub, Sombrero, and the Prickly Pear Cays (Gore et al., 2019). In December 2024, Anguilla's population was 16,010, compared to 6,500 in 1980. Despite its modest land area of 91 km² and 61 km of coastline, it maintains an exclusive economic zone (EEZ) of more than 92,000 km².

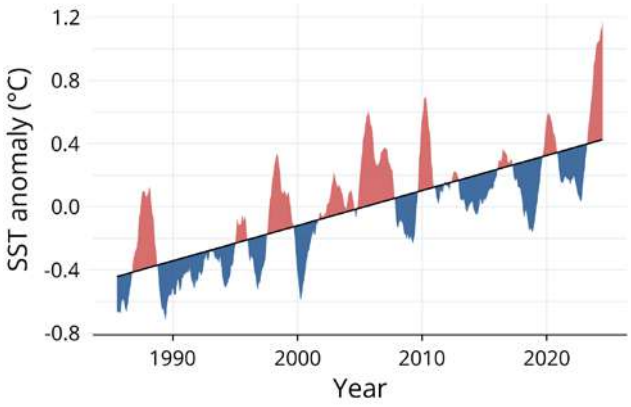
The island hosts 52 km² of coral reefs, representing 0.216% of the Caribbean's total reef extent. Vital to both fisheries and tourism, these reefs underpin local livelihoods. Tourism, once accounting for 56.6% of GDP, suffered a severe decline following the 2017 hurricanes, with an estimated 400% drop (Government of Anguilla, 2017).

To protect its reefs, Anguilla has established eight marine protected areas, including seven marine parks designated under the 2010 Marine Parks Regulation (Gore et al., 2019), and developed a long-term monitoring program.

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Anguilla was 27.50 °C. Over this period, SST increased by 1.03 °C, representing a warming rate of 0.026 °C per year (**Figure 2.1.1**). During the marine heatwaves of 2005, Anguilla was in the hotspot from June to October, and bleaching was observed in several locations. However, due to the absence of monitoring that year, the extent of coral bleaching is unknown (Bouchon et al., 2008).

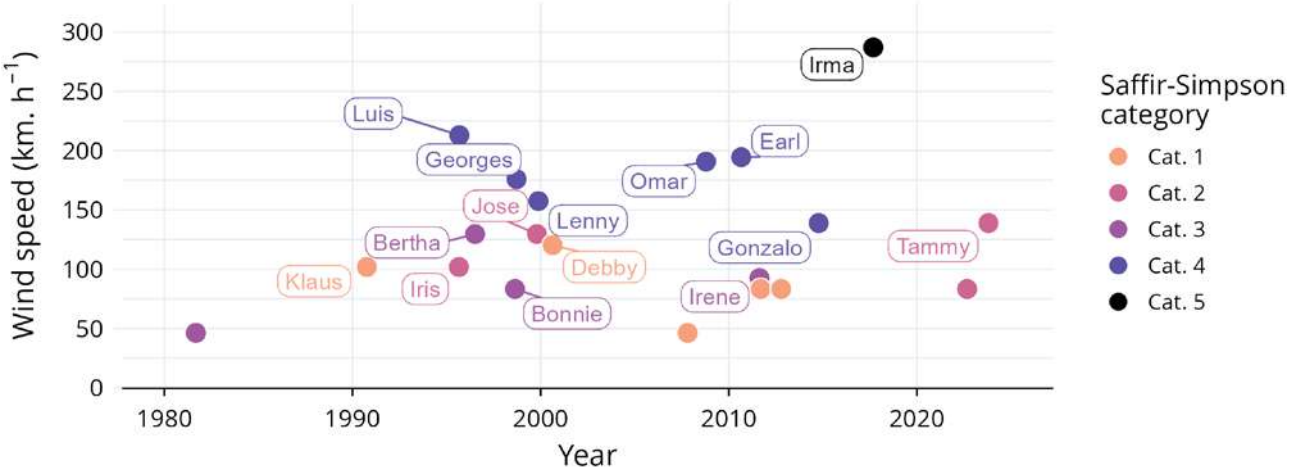


↑ **Figure 2.1.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Anguilla. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 28 tropical storms passed within 100 km distance from a coral reef of Anguilla, and of these 14 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.1.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed 6 km from a coral reef with sustained wind speed of 287 km.h^{-1}.

Repeated hurricanes have had strong impacts on Anguilla's coastline and coastal ecosystems, driving significant shoreline retreat and habitat loss. Between 1992 and 2014, beach profile data from 14 sites on the mainland and offshore cays of Prickly Pear and Sandy Island recorded an overall erosion rate of −0.51 m per year (Cambers and Wynne, 2019).



↑ **Figure 2.1.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Anguilla. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Anguilla are not represented, although they can have had an impact.

Other threats

Anguilla's reefs additionally face multiple human-driven and ecological pressures.

Although fishing is a relatively small industry, contributing about 2.26% of GDP in 2012 and supporting over 200 residents (Gore et al., 2019; Gumbs, Wynne, and Johnson, 2015), fishing effort has intensified. Trap use per vessel has risen nearly 700% in 30 years, from 25 to over 200, increasing pressure on reef species and altering ecological balances.

Lionfish (*Pterois volitans*), first reported in 2010, are now widespread. Populations are relatively low on most shallow reef areas, but some hotspots have been identified at Crocus Bay, Meads Bay, Anguillita, and West Cay, where lionfish repopulate soon after eradication efforts (Wynne, 2016).

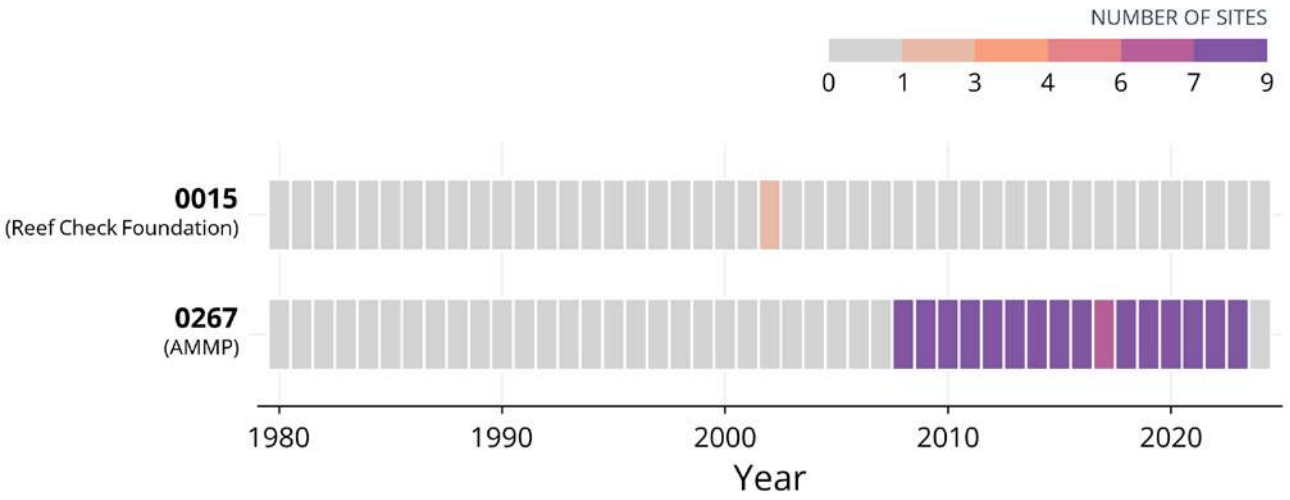
While demand from restaurants has supported removal, populations remain high in offshore areas, threatening native reef fish.

The loss of the grazing sea urchin *Diadema antillarum* has reduced algae control, and coral diseases continue to undermine reef resilience (Bouchon et al., 2008).

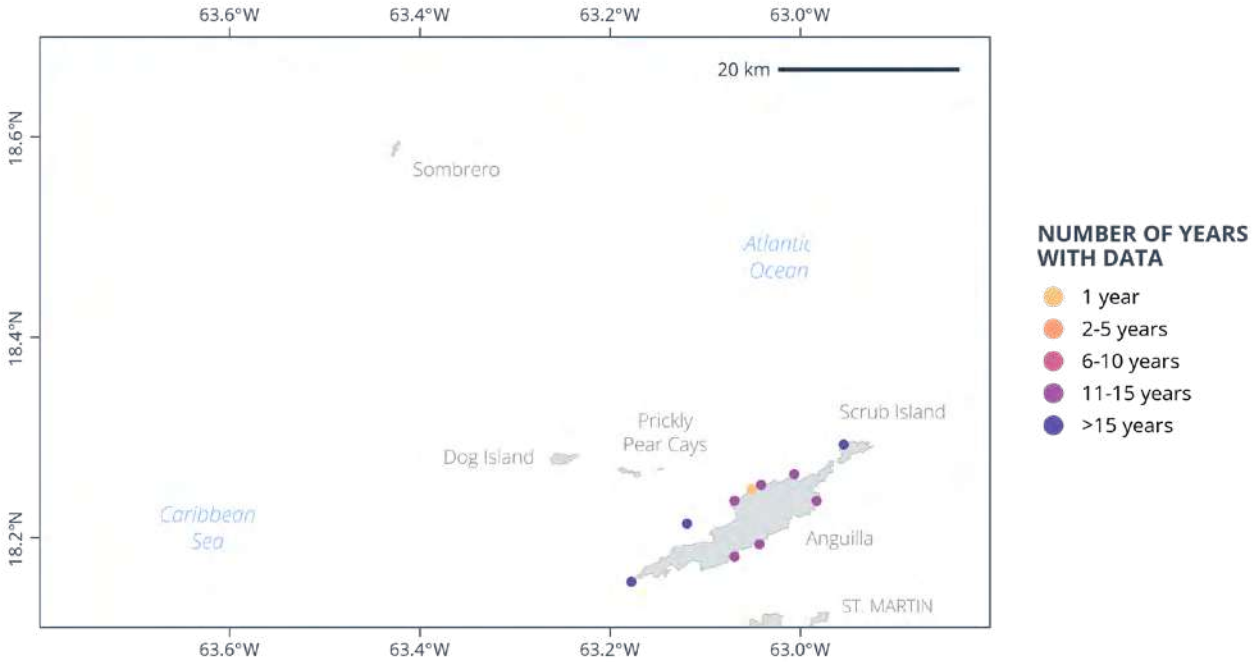
Like other islands in the region, Anguilla identified the presence of Stoney Coral Tissue Loss Disease (SCTLD) within reef ecosystems, both near the mainland and offshore cays. Coral mortality and extent of damage caused by SCTLD is not known.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.1.3**) encompassing 10 monitoring sites (**Figure 2.1.4**). These data were collected through 132 surveys conducted from 2002 to 2023. A previous study was conducted by the Bellairs Institute of Barbados in 1990, and improved data collection began in 2007 as the Anguilla Marine Monitoring Programme (AMMP) led by the Department of Fisheries and Marine Resources, in partial collaboration with the Anguilla National Trust.

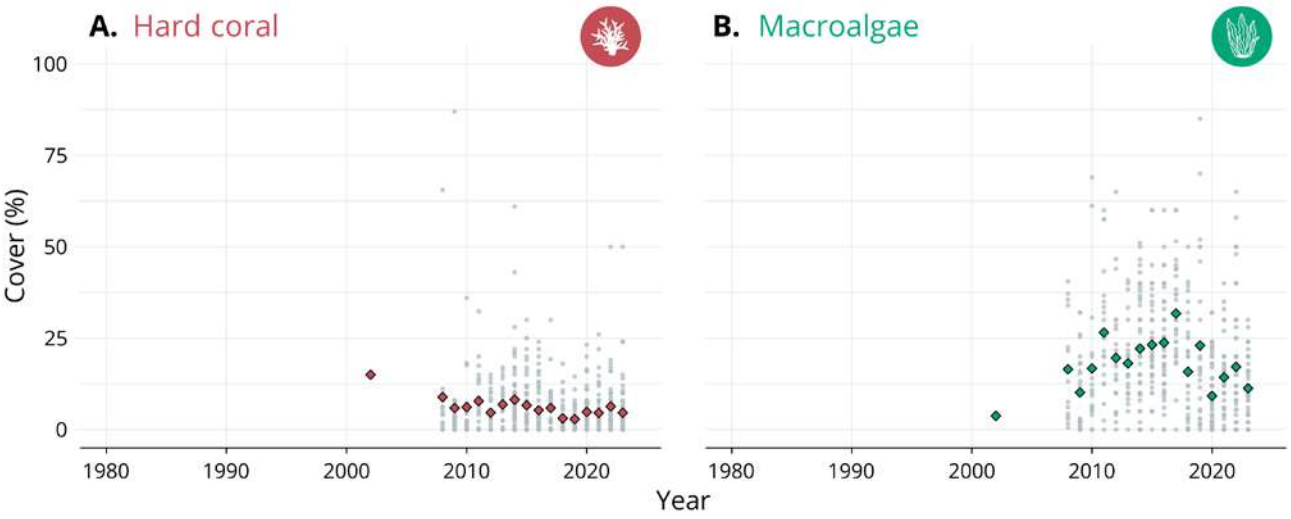


↑ **Figure 2.1.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.1.4.** Spatio-temporal distribution of benthic cover monitoring sites across Anguilla. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

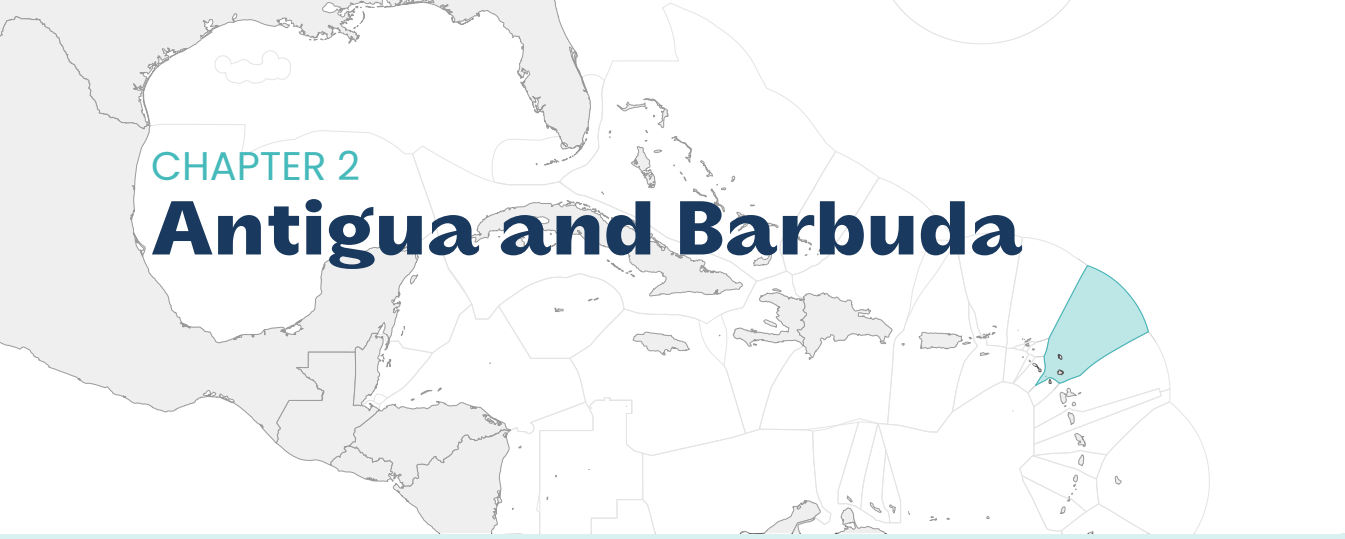
Anguilla's coral reefs, once described as largely pristine before the 1980s, have undergone a steady decline due to a combination of disease, hurricanes, and chronic anthropogenic pressures. The mass mortality of *Acropora palmata* in the 1980s, attributed to white-band disease, marked a turning point in reef health (Bouchon et al., 2008). By 1990, coral cover ranged from 4% to 23%, with the highest values recorded on offshore cays. However, Hurricane Luis in 1995 devastated Anguilla's marine ecosystems, severely damaging corals, with a loss of octocoral cover by 83%, *Montastraea* spp. by 95%, and other corals by nearly 90% (Bouchon et al., 2008). Hurricane Lenny in 1999 inflicted further damage, and subsequent surveys revealed only partial survival of offshore coral communities.



↑ **Figure 2.1.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Anguilla from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Long-term monitoring between 2007 and 2017 has provided critical insight into ongoing trends. Over this period, coral cover declined by 11%, from a mean of 5.6% to 5.1%, while algal cover increased by nearly 10%, from 14.2% to 15.0% (Wynne, 2017; Cambers and Wynne, 2019). Increase of macroalgae cover is linked to herbivore overfishing, the decline of *Diadema antillarum*, and nutrient enrichment from coastal development (Bouchon et al., 2008). *Porites astreoides* emerged as the most common coral species, though its frequency dropped by 45% over the decade. The massive reef-building *Acropora palmata*, once dominant, now accounts for only about 2% of recorded colonies (Wynne, 2017). Concurrently, fish communities have shown marked declines: species richness fell by 21%, abundance by 27%, and mean body size by 24%. Reductions were particularly pronounced among key ecological and commercial families, with average sizes of surgeonfish (*Acanthuridae*), parrotfish (*Scaridae*), and groupers (*Serranidae*) decreasing by 31%, 9%, and 54% respectively (Wynne, 2017).

While offshore sites show slightly greater resilience, most reefs remain in decline, with little evidence of significant recovery (Gore et al., 2019).



CHAPTER 2

Antigua and Barbuda

Authors: Ruleo Camacho, Margaret Wilson, Laëtitia Mathon.

Data contributors: Marilyn Brandt*, Ruleo Camacho*, Amit Hazra*, Alison Moulding*, Jasiel Murphy*, Suzanne Palmer*, Sophia Punnett*, Margaret Wilson*, Aletta Yniguez*.

*AGRRA data contributor(s).

INTRODUCTION

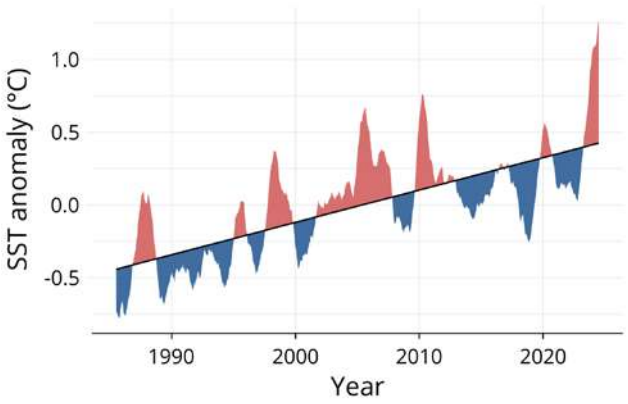
Antigua and Barbuda, a twin-island nation in the Lesser Antilles, is home to nearly 94,000 inhabitants in 2024 with a population density of 224 people per km², compared to 64,000 in 1980. The islands differ geologically and socioeconomically: Antigua, composed of volcanic rock, clay, and limestone, is more developed with a stronger tourism sector and higher population density, while Barbuda, largely limestone with notable cave systems, remains sparsely populated and less developed (Potter et al., 2017). Together, the islands host 116 km² of coral reefs, representing about 0.48% of the Caribbean's total reef extent. These reefs are vital to the country's economic sustainability, particularly given its reliance on fishing and tourism, yet many activities continue to place unsustainable pressures on the coastal environment (Ramsey et al., 2015). To safeguard these critical ecosystems, several marine protected areas (MPAs) have been established, including Diamond Reef and Palaster Reef Marine Parks (1973), Cades Bay Marine Park (1999), and both Codrington Lagoon and the North Sound (2005), although the level of implementation of the latter is unknown (Bouchon et al., 2008). Additional management of marine areas are executed within the Nelson Dockyard National Park under the management of the National Park Authority (Camacho et al., 2020) and the Redonda Ecosystem Reserve under the management of the Environmental Awareness Group.

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Antigua and Barbuda was 27.49 °C. Over this period, SST increased by 1.03 °C, representing a warming rate of 0.027 °C per year (**Figure 2.2.1**).

Marine heatwaves in 2005 and 2023 led to bleaching episodes, with the bleaching events of 2023 resulting in over 90% tissue loss among nursery populations (NPA Coral Bleaching Report, 2023).

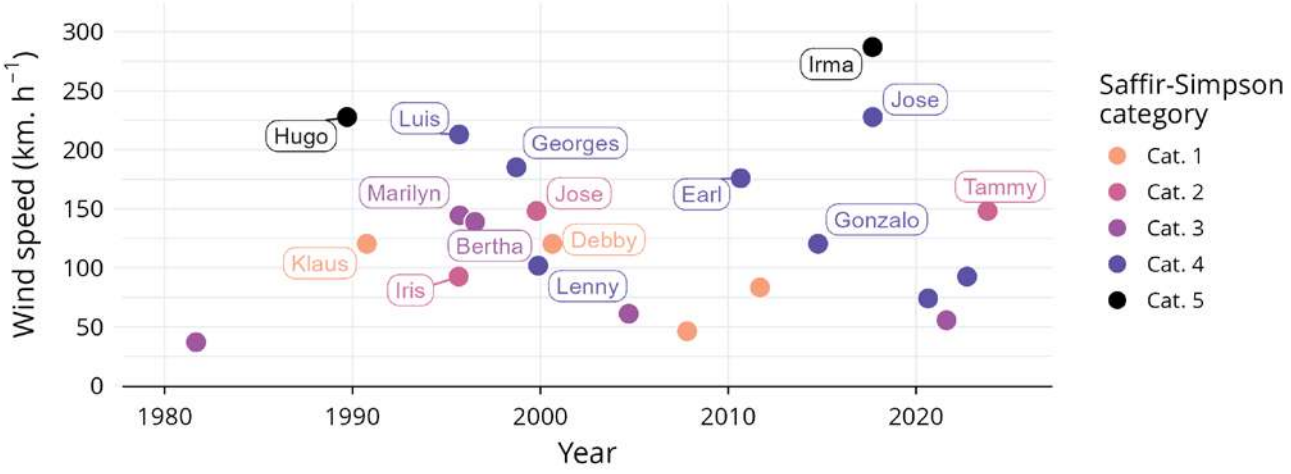


↑ **Figure 2.2.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Antigua and Barbuda. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 36 tropical storms passed within 100 km distance from a coral reef of Antigua and Barbuda, and of these 14 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.2.2**). The hurricane with the highest sustained wind speed over the studied period was Hurricane Irma in 2017, which passed directly over coral reefs with sustained wind speed of 287 km.h^{-1}.

↓ **Figure 2.2.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Antigua and Barbuda. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Antigua and Barbuda are not represented, although they can have had an impact.

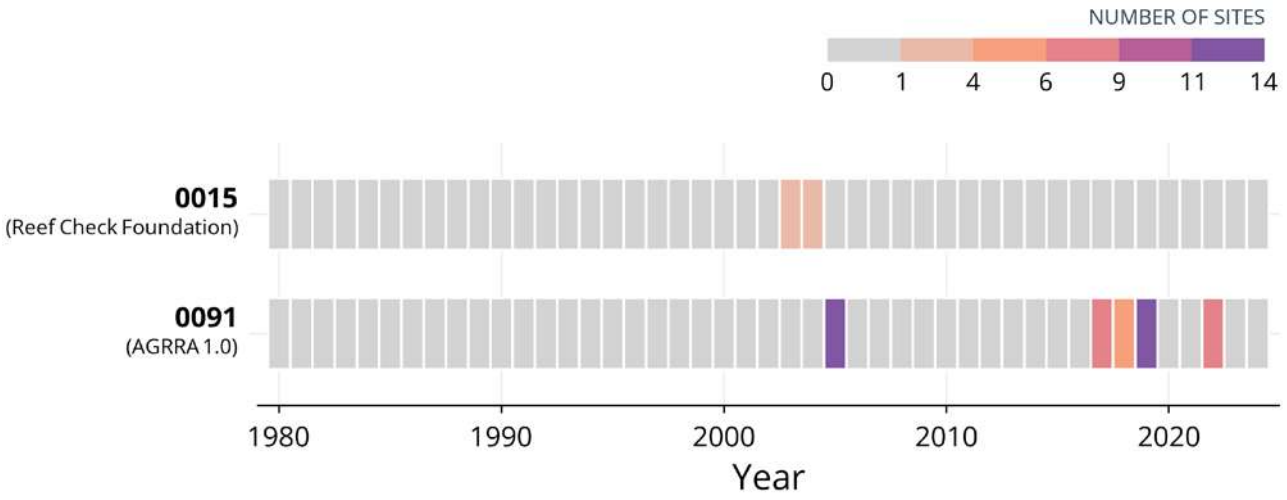


Other threats

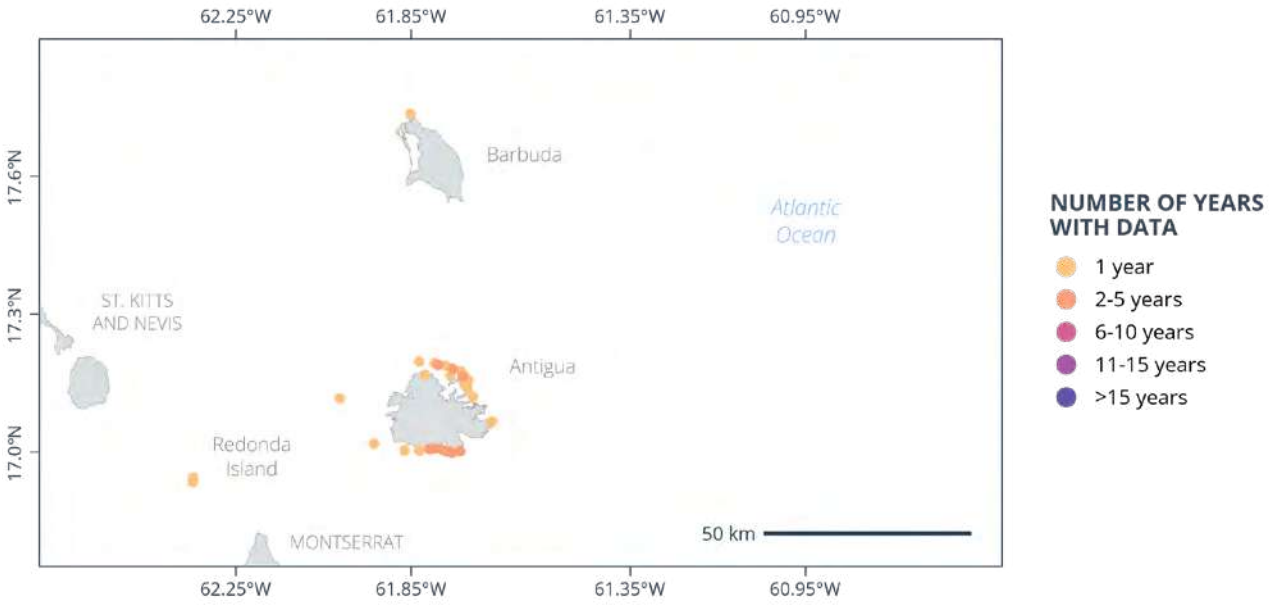
Coral reefs in Antigua and Barbuda face a range of ecological and human-induced threats that compromise their health and resilience. Biological stressors such as competition from macroalgae (Camacho and Steneck, 2017), cyanobacteria, and sponges, alongside coral diseases like yellow blotch and white plague, have significantly affected key species, particularly following marine heatwave and bleaching events. Physical stressors, including increased sedimentation—especially near dredged areas—have smothered shallow corals, while signs of recent death of *Acropora palmata* colonies highlight ongoing ecosystem stress (Bouchon et al. 2008). Loss of marine herbivores due to threats from development, overexploitation (Camacho and Steneck, 2017, Camacho et. al. 2020) and diseases (Hewson et al., 2023), also further facilitate increased growth of macroalgae and its associated threats to live coral. Human pressures exacerbate these natural threats: overfishing and excessive exploitation of marine resources strain the ecological foundation of economic growth (Carr and Heyman, 2009). Hurricanes can further threaten fisheries and reef communities, by damaging fishing gears and releasing ghost nets, as seen with the loss of 44% of fish traps after Hurricane Irma (Horsford, 2017).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.2.3**) encompassing 41 monitoring sites (**Figure 2.2.4**). These data were collected through 50 surveys conducted between 2003 and 2022. Reef monitoring has been implemented since 2016 using the AGRRA protocol (Camacho et al., 2020).



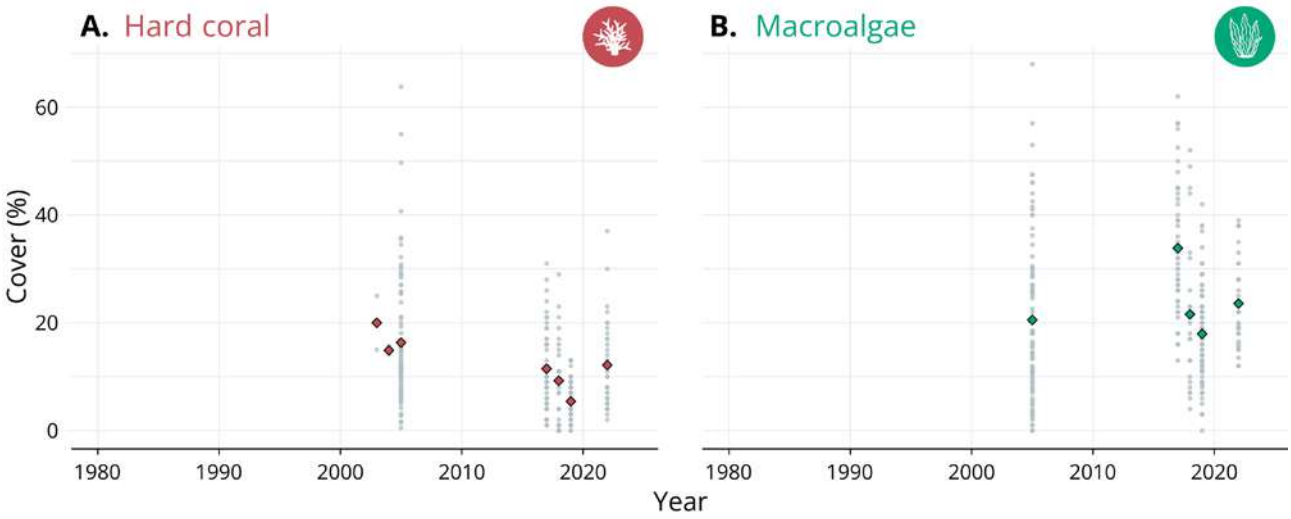
↑ **Figure 2.2.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.2.4.** Spatio-temporal distribution of benthic cover monitoring sites across Antigua and Barbuda. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

The coral reefs of Antigua and Barbuda have shown limited recovery over recent decades, with much of the reef structure dominated by large, interlocking branches of dead elkhorn coral (*Acropora palmata*), a remnant of previously luxuriant growth (Bouchon et al., 2008). Surveys in 2008 recorded 32 species of stony corals covering 2–19% of the benthos (mean 6.7%), with *Porites astreoides*, *Agaricia* spp., and *Montastraea* spp. being the most abundant, while *Acropora* species accounted for only 1.5% of cover and were sparsely distributed. Juvenile coral recruitment was very low (<0.9 recruits/m²), suggesting limited natural replenishment. Macroalgae, turf algae, and cyanobacteria were widespread, often smothering coral colonies and indicating competitive pressures on reef recovery (Bouchon et al., 2008).

More recent surveys (2017–2019) report an average live coral cover of roughly 9%, with macroalgal and turf algae cover averaging 23% each, showing little change at a national scale since 2016 (Camacho et al., 2020). This stability may reflect the slow growth of dominant brain corals and the absence of recent severe bleaching or Stony Coral Tissue Loss Disease events, yet it also underscores the persistent low abundance of key coral species and the decline of herbivorous fish, which exacerbates macroalgal dominance. Overall, coral cover and fish biomass remain similar to or below regional Caribbean levels, highlighting that Antigua and Barbuda's reefs continue to face structural degradation, limited recovery, and ecological pressures that constrain their resilience (Ruttenberg et al., 2018).



↑ **Figure 2.2.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Antigua and Barbuda from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 3

Aruba

Authors: Laëtitia Mathon.

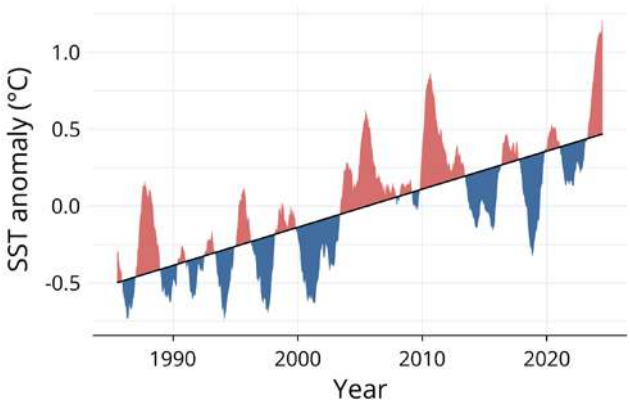
INTRODUCTION

Aruba, the westernmost of the ABC islands, covers 179 km² and is home to 107,624 inhabitants in 2024, compared to 60,000 in 1980 (van Duyl et al., 2024). Its southwestern coast is characterized by sandy beaches and a lagoon system, while the Caribbean Current, driven by trade winds and enriched by freshwater from the Amazon and Orinoco rivers, shapes local oceanography. The island hosts 99 km² of coral reefs, representing 0.410% of the Caribbean total. These reefs support over 240 species, including endangered sea turtles, Queen Conch, and Caribbean Spiny Lobster, while providing critical services such as coastal protection, nutrient cycling, and carbon sequestration (ACF, 2025). The combined value of these ecosystems through tourism, culture, and fisheries exceeds US$ 287.3 million annually, with reef-associated fisheries landing about 749 tons per year (Polaszek et al., 2018). Tourism peaked at ~2 million visitors in 2019 before declining to 940,000 in 2021 due to COVID-19. Aruba has taken conservation steps, including the creation of four marine protected areas (MacRae and Meyer, 2020).

THREATS

Thermal regime

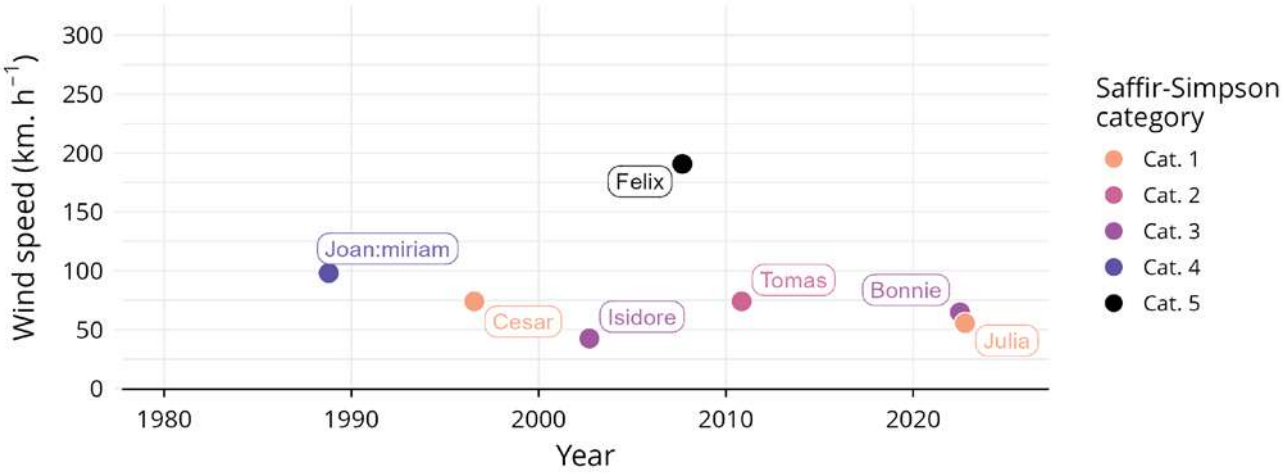
The long-term average in SST from 1985 to 2024 over coral reefs of Aruba was 27.28 °C. Over this period, SST increased by 1.08 °C, representing a warming rate of 0.028 °C per year (**Figure 2.3.1**).



↑ **Figure 2.3.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Aruba. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 10 tropical storms passed within 100 km distance from a coral reef of Aruba, and of these 1 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.3.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Felix, in 2007, which passed 57 km from a coral reef with sustained wind speed of 191 km.h^{-1}.



↑ **Figure 2.3.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Aruba. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Aruba are not represented, although they can have had an impact.

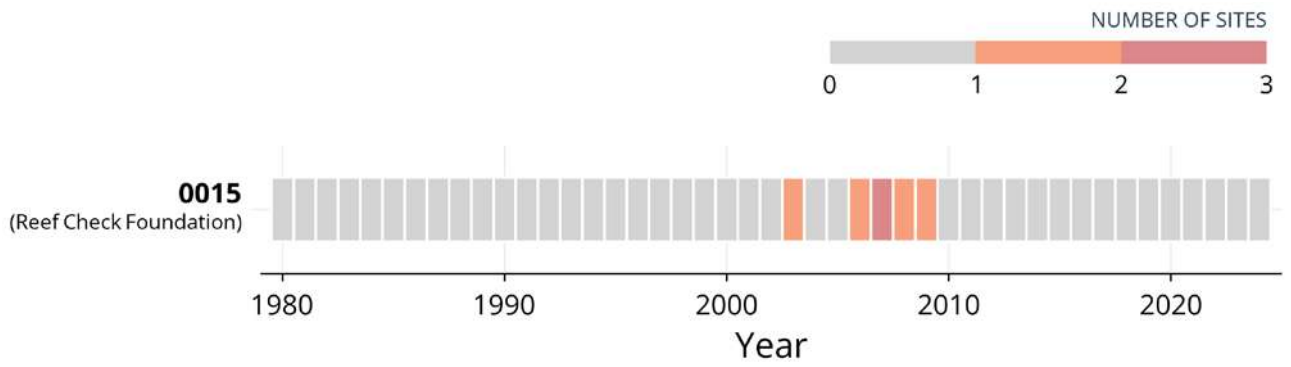
Other threats

Aruba's coral reefs are increasingly threatened by a combination of historic and ongoing human impacts. Industrial activity from the Lago Oil and Transport Co. refinery (1925–1985) caused localized reef degradation through oil contamination, dredging, and shipping disturbances, leaving adjacent reefs with low coral diversity and abundant dead rubble compared to unaffected control sites (Eakin et al., 1993). Today, pollution and eutrophication represent major stressors. Nutrient concentrations (NH_4, NO_3, NO_2) and chlorophyll-a levels are higher around Aruba than on Curaçao or Bonaire, reflecting higher eutrophication, particularly near urban areas and former oil terminals (van Duyl et al., 2024). This nutrient enrichment, linked to untreated sewage inputs and land-based runoff, fosters phytoplankton blooms and has been confirmed by isotopic and metabolomic evidence of widespread sewage and synthetic contaminants in coastal waters (Vermeij et al., 2019).

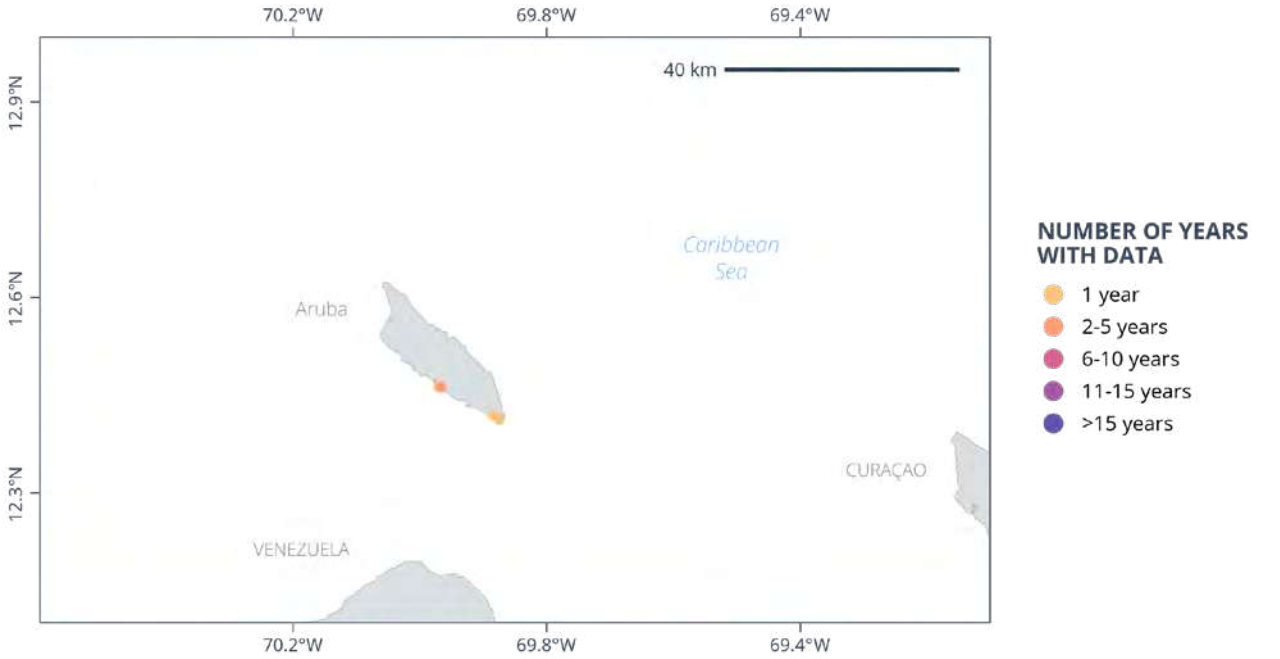
Unregulated coastal development further exacerbates reef decline. Construction of piers, ranchos, and artificial beaches has led to mangrove removal, sedimentation, and sewage discharge, while indirect effects such as eutrophication, trash, and illegal harvesting of protected species push ecosystems toward a critical tipping point (ACF, 2025). Additional pollution arises from poorly managed dumpsites, where leachate containing heavy metals contaminates surrounding waters (Beroske and Timpen, 2018).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 1 datasets (**Figure 2.3.3**) encompassing 6 monitoring sites (**Figure 2.3.4**). These data were collected through 7 surveys conducted between 2003 and 2009.



↑ **Figure 2.3.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.3.4.** Spatio-temporal distribution of benthic cover monitoring sites across Aruba. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Coral reefs in Aruba have undergone a marked long-term decline shaped by both chronic stressors and acute disturbances. In the mid-1970s, coral cover on the lower reef terrace and drop-off zones ranged between 30–40% (Bak, 1977). By the late 1980s, major structural shifts were evident: shallow reefs once dominated by *Montastraea annularis* were largely replaced by smaller brain corals (*Diploria* spp.) (Bak, 1987). Monitoring of comparable sites revealed an average decline in coral cover from 22.2% in 1986/88 to just 8.8% by 2019—a ~60% reduction over three decades (Vermeij et al., 2019). While a few sites with historically low coral cover (<4%) showed modest recovery, reefs along the northwestern coast, especially near tourism centers, became dominated by turf algae, cyanobacteria, and sponges, with corals, gorgonians, and crustose coralline algae (CCA) largely absent (Vermeij et al., 2019).

Recent surveys indicate stabilization, albeit at low coral cover: 9.26% in 2019, 8.85% in 2021, and 9.35% in 2023. This apparent stability masks important ecological shifts. Nearly half of surveyed sites showed increasing macroalgal cover at the expense of corals, CCA declined steadily, and herbivorous fish biomass dropped by more than half between 2021 and 2023, reducing grazing pressure critical for algal control (Eicher, 2024). Cyanobacteria presence increased sharply, from 14.7% of photographic records in 2021 to nearly 60% in 2023.

In December 2022, the outbreak of stony coral tissue loss disease (SCTLD) added a new acute stressor. Within six months, 13% of corals showed signs of infection, affecting nine species, including key reef builders such as *Montastraea cavernosa* and *Orbicella annularis* (Eicher, 2024).

Overall, temporal trends highlight a long-term trajectory of coral decline from historically healthy reefs to modern, algae- and cyanobacteria-dominated states, now further compounded by emerging coral diseases and human-driven pressures.

CHAPTER 4

The Bahamas

Authors: Jérémy Wicquart.

Data contributors: Elizabeth Abney*, Shelby Allen*, Fred Arnett*, Christian Avila*, Kenneth Banks*, Tamanji Bethel*, Greg Braun*, Dan Brumbaugh*, Felicity Burrows*, Lindy C. Knowles*, Shelley Cant*, Shelley Cant-Woodside*, Hayley Carr*, Kimberly Cartwright*, Shaun Christean*, Allison Corwin*, Craig Dahlgren*, Ancilleno Davis*, Giselle Deane*, Alexandra Dempsey*, Jared Dillet*, Beth F. Moses*, Nancy Gasman*, Brooke Gintert*, Will Greene*, Eric Grunthaner*, Thomas Guryan*, Lily Haines*, Vanessa Haley-Benjamin*, Alex Henderson*, James Herlan*, Drew Hitchner*, Tanya Kamerman*, Jeroen Kenter*, Luke Kikukawa*, Kristin Kimble*, Vladimir Kosmynin*, Philip Kramer*, Patricia Kramer*, Judith Lang*, Jillian Lessing*, Kemit-Amon Lewis*, Agnessa Lundy*, Christy Macdonald*, Sheldon Mackey*, Ken Marks*, Scott Martin*, Casuarina Mckinney*, Sarah Meyers*, Chris Moses*, Rochelle Newbold*, Valeria Pizarro*, Kate Porterfield*, Elyssa Quinton*, Mallory Raphael*, Bernhard Riegl*, Lakeshia S. Anderson*, Joshua S. Feingold*, Blake Schembri*, Krista Sherman*, Nikita Shiel-Rolle*, Andy Stamper*, Kristine Stump*, Tonika Sweeting-Stubbs*, Collin Tennant*, Susan Thornton*, Kathryn Van Artsdalen*, Alannah Vellacott*, Sandy Voegeli*, Andrew W. Bruckner*, Brad Weiler*, Dave Wert*, Amanda Williams*, Olivia Williamson*, Anna Zuke*.

*AGRRA data contributor(s).

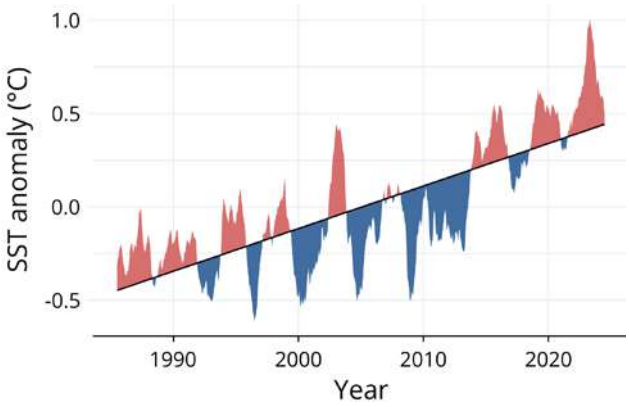
INTRODUCTION

The Bahamas, an independent island nation in the Atlantic Ocean, lies to the north of Cuba and Hispaniola and southeast of Florida. The archipelago consists of more than 700 islands and cays, of which about 30 are inhabited, the largest being Andros, Grand Bahama, Eleuthera, and New Providence (**Figure 2.4.4**). Coral reef extent is about 4,081 km², which represents nearly 17% of the Caribbean's total coral reef extent.

THREATS

Thermal regime

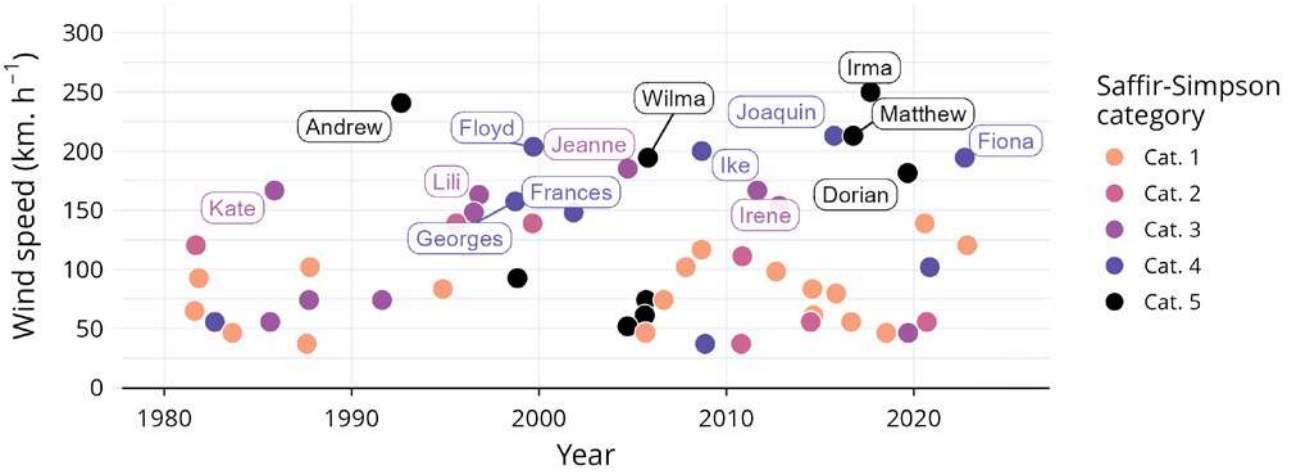
The long-term average in SST from 1985 to 2024 over coral reefs of The Bahamas was 26.91 °C. Over this period, SST increased by 1.00 °C, representing a warming rate of 0.026 °C per year (**Figure 2.4.1**).



↑ **Figure 2.4.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of The Bahamas. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 88 tropical storms passed within 100 km distance from a coral reef of The Bahamas, and of these 28 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.4.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed 0 km from a coral reef with sustained wind speed of 250 km.h^{-1}.

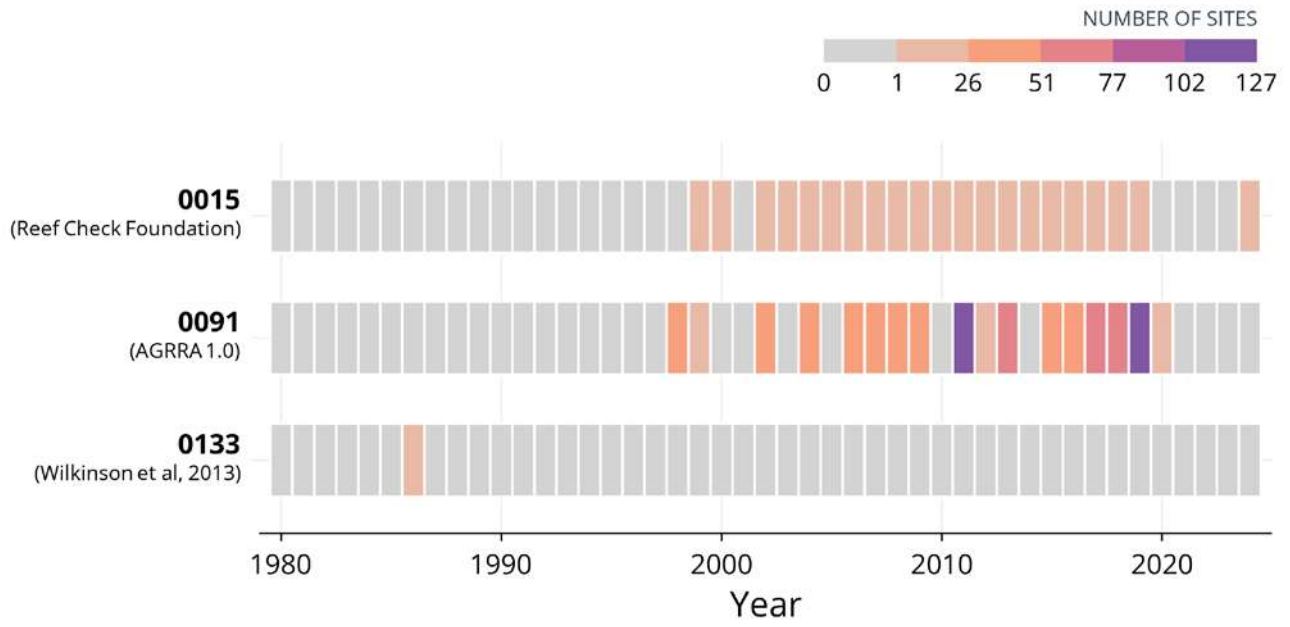


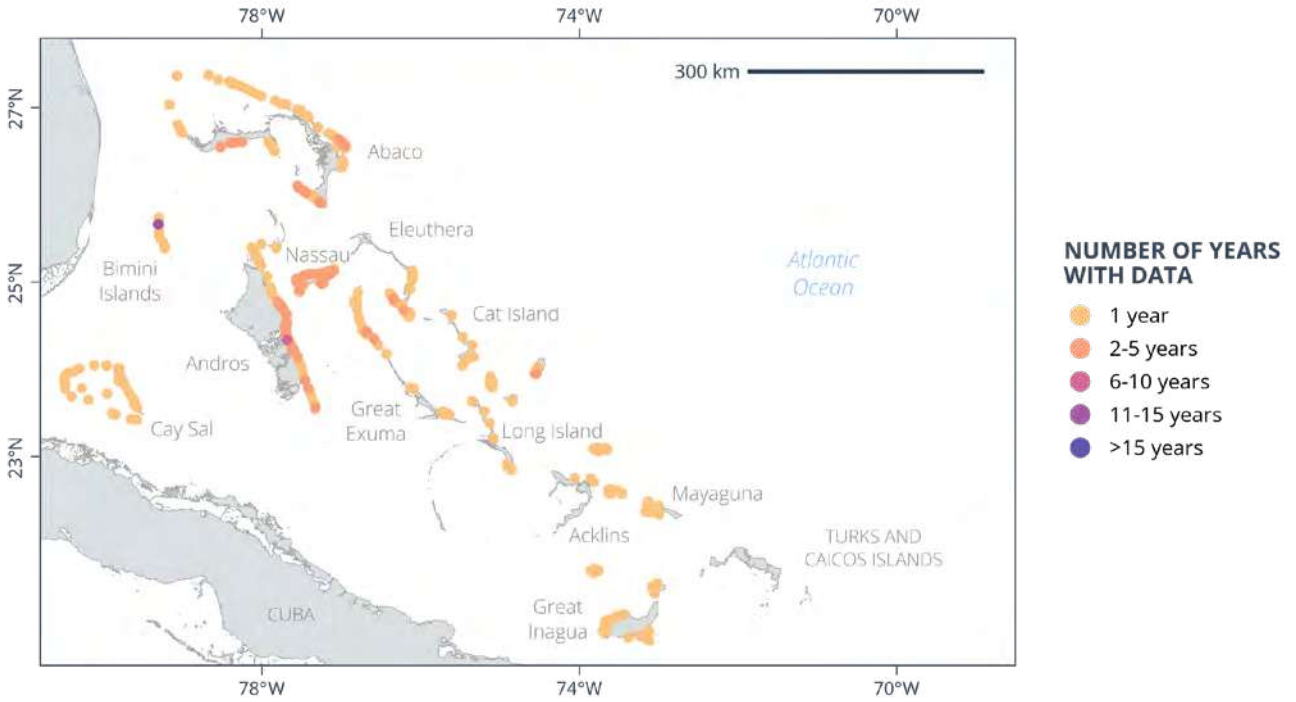
↑ **Figure 2.4.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in The Bahamas. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of The Bahamas are not represented, although they can have had an impact.

TRENDS IN BENTHIC COVER

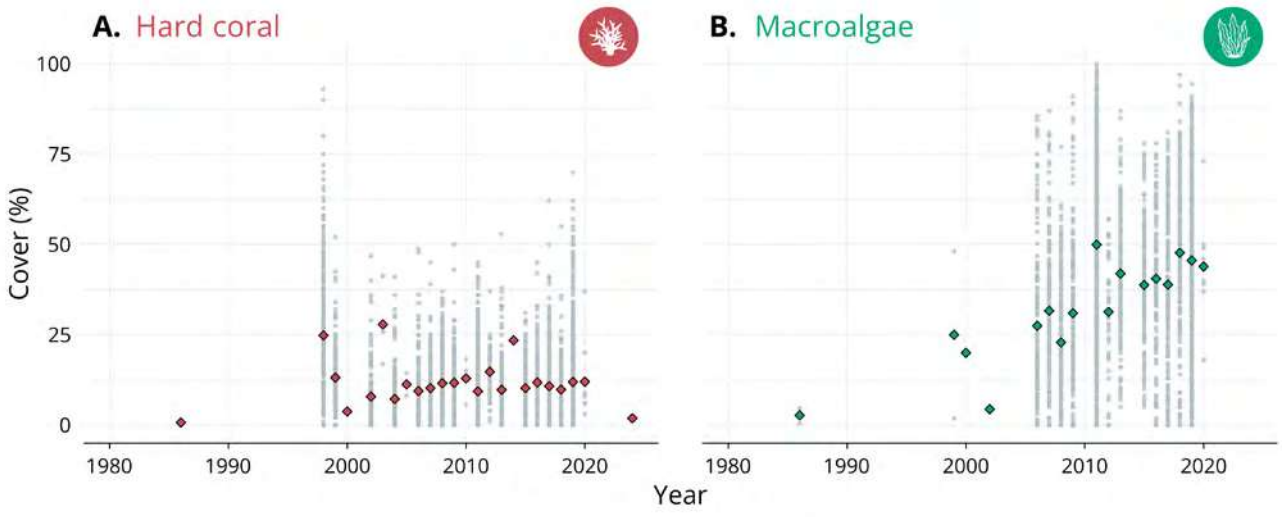
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.4.3**) encompassing 542 monitoring sites. These data were collected through 808 surveys conducted between 1986 and 2024.

↓ **Figure 2.4.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.4.4.** Spatio-temporal distribution of benthic cover monitoring sites across The Bahamas. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.4.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in The Bahamas from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 5

Barbados

Authors: Shoneé Howell, Laëtitia Mathon.

Data contributors: Robert Bourne, Leo Brewster, Amy Cox, Annabel Cox, Renata Goodridge, Alex Henderson, Shoneé Howell, Hazel Oxenford, Micaela Small, Holly Trew, Henri Valles, Julian Walcott, Joseph Weekes.

INTRODUCTION

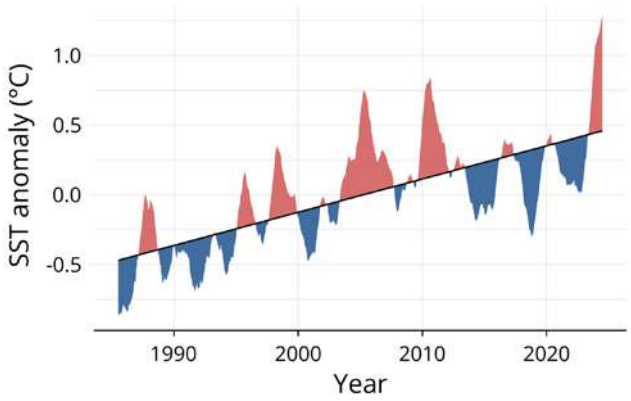
Barbados, the eastern-most island in the Caribbean Region, has a terrestrial extent of 430 km². In contrast to its small land mass, Barbados has an exclusive economic zone of 167,000 km² which contains approximately 62 km² of coral reefs, accounting for just 0.256% of the region's total reef extent. Despite a relatively stable human population over recent decades—rising modestly from 251,000 in 1980 to 282,000 in 2024—these reefs face significant pressures from human activity. Coastal tourism is one of the most important sectors to the island, making coral reefs and associated marine species a major draw for visitors. Therefore, their health is important to the sustainability of the economy. In an effort to protect them, Barbados established the Barbados Marine Reserve ("Folkestone") in 1981 along the island's west coast. This marine protected area covers 2.2km² and was established to protect and preserve marine resources while balancing conservation with recreation and research (Griffith et al., 2025). Efforts to increase protection through expansion and addition of new marine management areas are ongoing.

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Barbados was 27.74 °C. Over this period, SST increased by 1.09 °C, representing a warming rate of 0.028 °C per year (**Figure 2.5.1**).

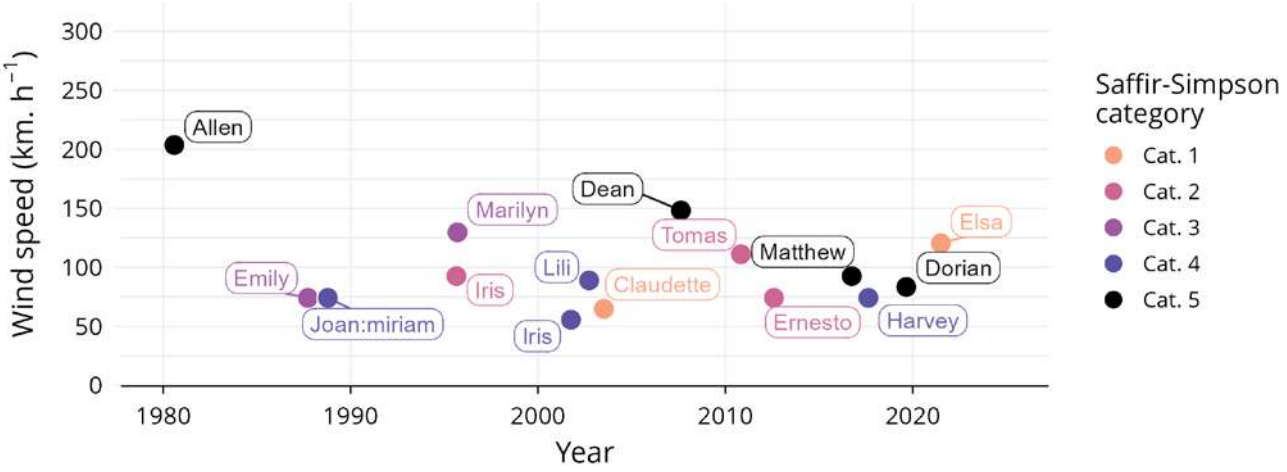
In late August 2005, a widespread coral bleaching event began and lasted for months even after sea temperatures cooled. About 70.6% of coral colonies in Barbados were affected across all reef habitats and taxa, with nearshore reefs (<10 m) bleaching more severely (80.6%) than deeper offshore reefs (>15 m; 60.5%). Susceptibility varied greatly among species, ranging from less than 10% to over 90% of colonies bleaching. Despite the extensive bleaching, mortality remained relatively low, with only 3.8% average partial colony death recorded by February 2006 (Oxenford et al., 2008).



↑ **Figure 2.5.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Barbados. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 27 tropical storms passed within 100 km of Barbados' coral reefs. Six of these were characterized by sustained wind speeds greater than 100 km.h^{-1} (**Figure 2.5.2**). The hurricane with the highest sustained wind speed over the studied period was Hurricane Allen, in 1980, which passed 2 km from a coral reef with sustained wind speed of 204 km.h^{-1}. Hurricane Allen caused severe damage to the northern fringing reef off the west coast of Barbados near the Bellairs Research Institute (Mah and Stearn, 1986). The coral *Porites porites* was the most heavily impacted, with its cover reduced by 96%, while other species—including *Siderastrea siderea*, *Porites astreoides*, *Montastrea annularis*, and *Agaricia agaricites*—experienced losses ranging from 25% to 75%. More recently, Hurricane Béryl in 2024 also led to significant damages.



↑ **Figure 2.5.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Barbados. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Barbados are not represented, although they can have had an impact.

Other threats

Beyond bleaching and storm damage, Barbados' coral reefs are impacted by a range of other local stressors. These stressors include increasing fishing pressure, land-based sources of marine pollution, introduction of alien invasive species, intensified coastal development, elevated sea surface temperatures, declines in key herbivores such as *Diadema antillarum*, coral disease (Stony Coral Tissue Loss Disease) and physical disturbances such as anchor damage.

Fishing activity was noted to have increased with semi-industrial longliners rising from about four vessels in the early 1980s to 80 by the late 1990s, and ice-boats growing from 12 in 1983 to 156 by 2000 (Mohammed, 2007).

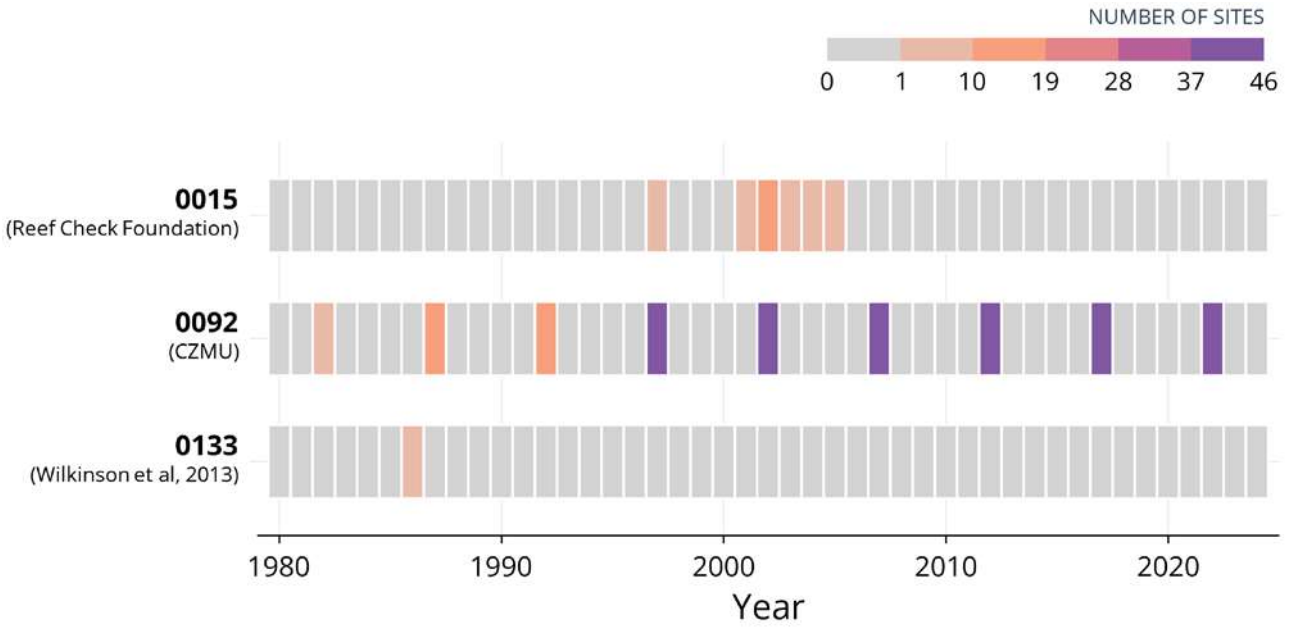
Water quality challenges are related to Barbados' karstic geology as the entire island may be considered a coastal zone. Nutrients from fertilizer use and inland suckwells are able to percolate into coastal waters, upwelling on reefs and contributing to elevated nutrient levels. There have also been notable effects on deepwater corals as varying levels of lead and tin were likely transported by currents, with hurricanes in 1995 dispersing nearshore polluted waters and elevating tin and copper concentrations (Runnalls and Coleman, 2003).

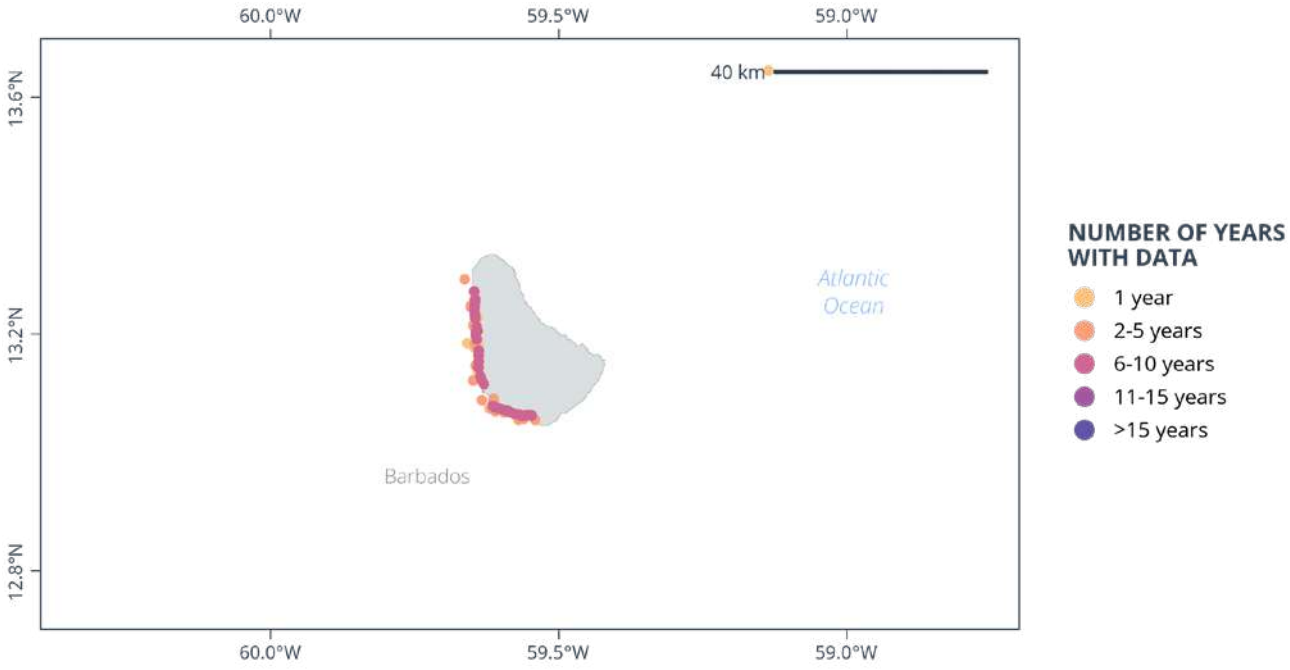
Furthermore, while suspended particulate matter can enhance coral growth at low levels, excessive concentrations, as seen in some sites in Barbados, smother corals, block light, and reduce photosynthesis, ultimately impairing reef health (Tomascik and Sander, 1985). These concerns are related to stormwaters draining into coastal waters.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.5.3**) encompassing 80 monitoring sites (**Figure 2.5.4**). These data were collected through 349 surveys conducted between 1982 and 2022.

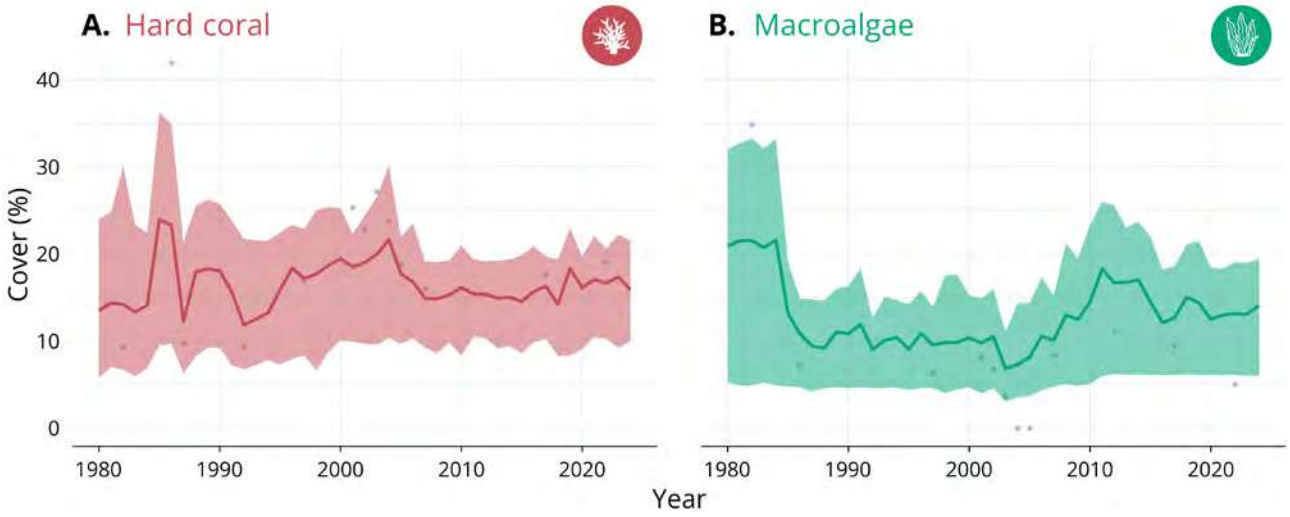
↓ **Figure 2.5.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.5.4.** Spatio-temporal distribution of benthic cover monitoring sites across Barbados. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Barbados' coral reefs have undergone steady decline over the past several decades, shaped by both natural and human-driven disturbances. Between 1950 and 1991, most reefs showed significant reductions in area, with a total loss of 8.4 ha, or about 11% (Lewis, 2002). Storm damage appeared to be the main driver, particularly in exposed spur-and-groove zones, though rapid coastal development and tourism growth also degraded water quality and fueled eutrophication (Bell and Tomascik, 1994).



↑ **Figure 2.5.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Barbados from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

The early 1980s marked a turning point, as Hurricane Allen in 1980 slashed coral cover from 37% to 15% and increased dead coral surfaces from 22% to 43% (Scoffin, 1994). Just a few years later, in 1983, the mass die-off of the grazing urchin *Diadema antillarum* further destabilized reefs, enabling turf algae to overtake dead coral substrates. By 1992, there was little sign of coral recovery, with dead coral cover rising to 60% and crustose coralline algae declining from 42% to 25%, although some species such as *Porites astreoides*, *Agaricia agaricites*, and *Millepora spp.* showed slight increases.

Barbados' coral reefs were impacted by three major disturbance events between 2022 and 2024: the arrival of Stony Coral Tissue Loss Disease (SCTLD) in November 2022, the fourth documented Global Coral Bleaching Event (2023–2024), and the passage of Hurricane Beryl in July 2024. Collectively, these events resulted in substantial shifts in reef condition. Analysis of a subset of locations from the national coral reef monitoring programme recorded a 67% decline in live coral cover and a concurrent 42% increase in macroalgal cover between 2023 and 2024 (Irvine et al., 2025). Stony Coral Tissue Loss Disease was particularly devastating, and intervention efforts were not as effective as hoped. Although treatments temporarily arrested lesion progression, they did not prevent long-term mortality, as treated colonies eventually died. Among affected species, maze coral (*Meandrina meandrites*) exhibited the highest levels of mortality, highlighting its elevated susceptibility within Barbados' reef assemblages.

CHAPTER 6

Belize

Authors: Melanie McField, Raphael Martinez, Lisa Carne, Alicia Eck-Nunez, Andrea Rivera-Sosa.

Data contributors: Belize Fisheries Department, Victor Alamina*, Hazel Araujo*, Andrés B. Alegría*, Tanya Barona*, Sara Beach*, Ivanna Blanco*, Nadia Bood*, Henry Brown*, Ana C. Fonseca*, Reylando Caal*, Lisa Carne*, Wilbert Castillo*, Karl Castillo*, Celso Cawich*, Jennifer Chapman*, Juan Chub*, Elise Cormier*, Nicole Craig*, Annick Cros*, Tisa Devalle*, Graham J. Edgar, Victor Faux*, Michelle Fernández*, Renata Ferrari*, Jocelyn Finch*, Grant Galant*, Olga Garcia*, Ana Giro*, Edgar Gonzalez*, Ruth Gutierrez*, Clay Harris*, Sam Hope*, Rob Landolfi*, Judith Lang*, Ken Marks*, Marieta Marroquin*, Ninon Martinez*, Melanie McField*, Chad Mcnutt*, Meira Mizrahi*, Ángela Mojica*, Alexander Navarro*, Kevin Novelo*, Lizzi Oh, Reynaldo Ortega*, Alerick Pacay*, Mito Paz*, David Penados*, Miguel Perez Castro*, Myles Phillips*, Roberto Pott*, Helder Pérez*, Patricia R. Kramer*, Anthony Rash*, Tyrell Reyes*, Julianne Robinson*, Alicia Rodríguez*, Marie Roman*, Burton Shank*, Celso Sho*, Rick D. Stuart-Smith, Tammy Summers*, Pippa Swannell*, Armeid Thompson*, Gabriela Ugarte*, Nicole Wulf*, Bonnie Young*.

*AGRRA data contributor(s).

INTRODUCTION

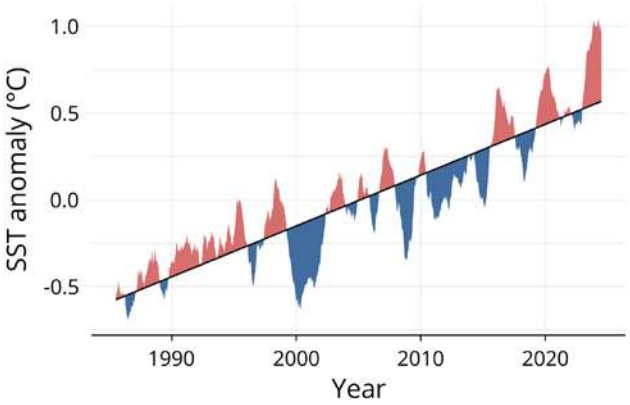
Belize, located in the Mesoamerican Region (MAR), borders Mexico to the north, Guatemala to the west and south, and the Caribbean Sea to the east. Belize hosts 1,506 km² of coral reefs (1,365km² according to Belize Fisheries Department) which represent approximately 6.215% of the Caribbean's total coral reef extent. Belize's marine realm includes the Belize Barrier Reef Reserve System a UNESCO World Heritage Site comprising seven protected areas, offshore atolls, hundreds of cayes, mangrove forests, estuaries, and coastal lagoons supporting nearly 1,400 marine species including threatened species (UNESCO, 2025).

On land, Belize covers about 22,966 km², with a population around 415,000, making it one of the least densely populated countries in the region (CIA, 2025). Economically, coral reefs and mangroves contribute substantially via tourism generated an estimated US$150–196 million annually (around 12–15% of GDP in 2007), reef-dependent fisheries about US$14–16 million, and shoreline protection from reefs and mangroves another US$231–347 million per year in avoided damage (Cooper et al., 2009).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Belize was 28.00 °C. Over this period, SST increased by 1.25 °C, representing a warming rate of 0.032 °C per year (**Figure 2.6.1**), which is the second highest rate in the tropical Caribbean region, as shown in Part 1, **Table 1.2.1**.

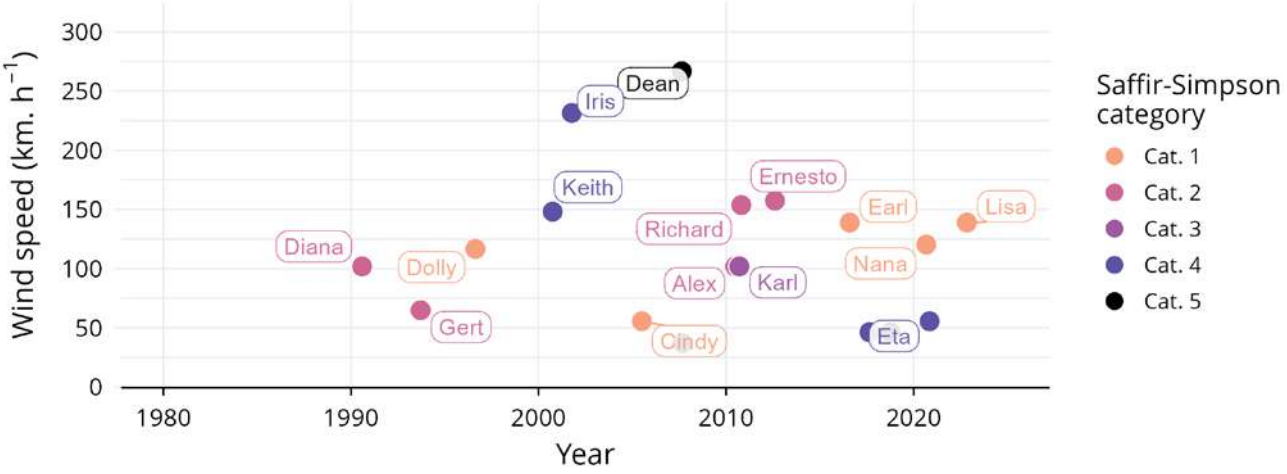


↑ **Figure 2.6.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Belize. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Coral bleaching events have become increasingly frequent in Belize and in the MAR region, with major mass-bleaching documented in 1995 (McField, 1999), 1998 (Aronson et al., 2000), 2005 (Eakin et al., 2010), and during the 2015–2017 warming episode (Muñiz-Castillo et al., 2019). There were also reports of substantial bleaching in 2019 and 2020. Finally, the 2023/24 bleaching event had the highest elevated thermal stress recorded, and it was unprecedented for having over 19 DHW, and affecting 100% of the coral reef, with more widespread mortality than any previous event. Impact monitoring in 2023 found the Northern Barrier Reef and Glovers Reef Atoll were particularly affected, having a bleaching severity index of 60-70 (McField et al., 2024).

Hurricanes

Between 1980 and 2024, a total of 33 tropical storms passed within 100 km of a coral reef in Belize, and of these, 14 were characterized by sustained wind speeds greater than 100 km.h^{-1} (**Figure 2.6.2**). The hurricane with the highest sustained wind speed over the studied period was hurricane Dean, in 2007, which passed 57 km from a coral reef with sustained wind speed of 267 km.h^{-1}.



↑ **Figure 2.6.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Belize. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Belize are not represented, although they may have had an impact, such as Hurricane Mitch in 1998, which resulted in considerable structural damage to reefs across the entire country.

Coral diseases

In addition to climate change, corals in Belize are facing significant degradation from disease and coastal development activities. One of the most pervasive threats is Stony Coral Tissue Loss Disease (SCTLD), which has reached Belize's reef systems is causing high mortality among reef-building corals (Belize Fisheries Department, 2023; Lizama and Belize Audubon Society, 2023). SCTLD was first reported in summer 2019 in the Bacalar Chico Marine Reserve near the Mexican border. In 2020, the disease spread to Southern Ambergris Caye, Caye Caulker, and Lighthouse Reef, followed in 2021 by the Central Barrier Reef and Turneffe. In 2022, many areas of the Southern Barrier Reef and Glovers Reef also became infected (McField et al., 2022). SCTLD was only recorded in Laughing Bird Caye in May 2025. Monitoring conducted in September 2025 in the Port Honduras and Sapodilla Cayes Marine Reserves (including Cayman Crown) found they remain unaffected (Martinez, *pers. obs*).

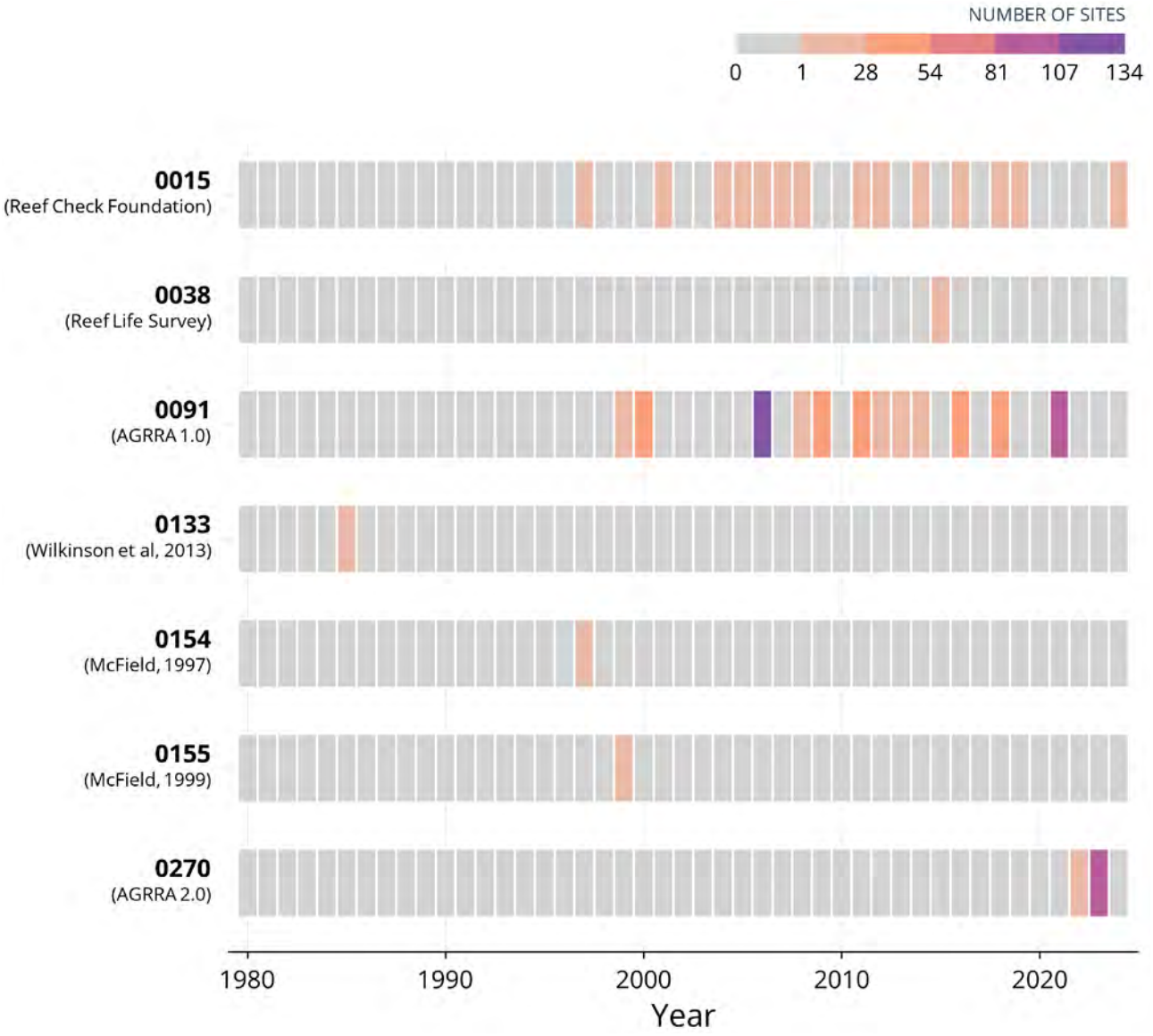
Belize tested a number of treatment options, including CoreRx Base2B + Amoxicillin paste on affected lesions which proved largely successful, with 88% of lesions prevented from expanding, although repeated treatments are needed and can not be sustained on a large scale (Lee Hing et al., 2022). SCTLD is the most virulent and pervasive coral disease to date, although the older diseases are still occurring at lower rates.

Coastal Development

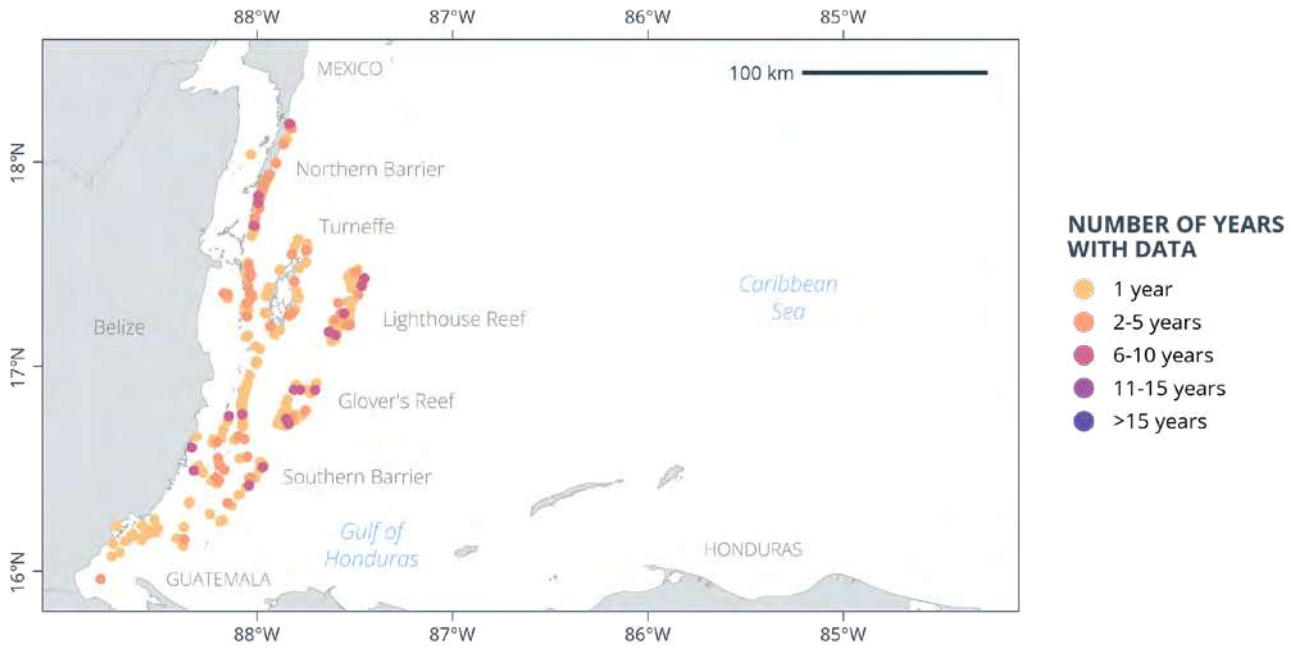
A large number of coastal developments and dredging activities involving port(s) expansion, sedimentation, and shoreline alteration, have severely impacted water quality, increasing turbidity and smothering coral colonies (Carlson et al., 2021). In Belize, dredging near the reef and increased sediment/nutrient runoff from terrestrial sources have been linked to reduced coral cover and shifts toward macroalgal dominance (Strategy for Reef Resilience in Belize, 2023). The combined effect of infectious coral disease plus physical and chemical stress from coastal construction multiplies the risks and compromises reef resilience. Alongside the physical stressors associated with coastal development comes the increasing problem of inadequate sewage treatment in growing coastal towns and tourist destinations. While nitrates and ammonia were mostly in fair condition across Belize, phosphorous and turbidity levels were often poor or unacceptable (McField et al., 2024).

TRENDS IN BENTHIC COVER

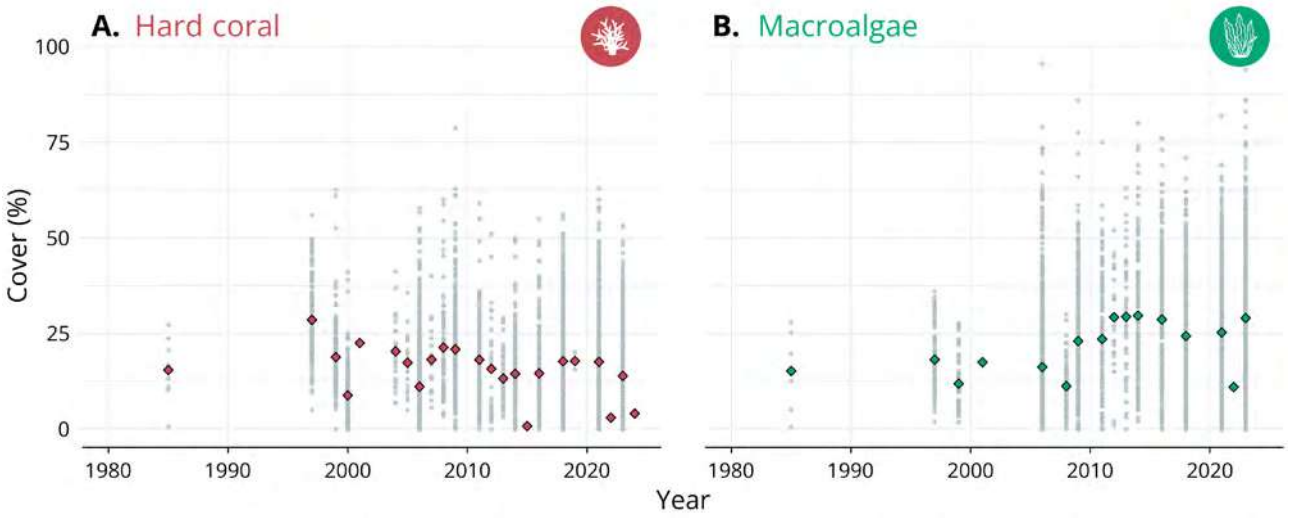
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 7 datasets (**Figure 2.6.3**) encompassing 440 monitoring sites. These data were collected through 703 surveys conducted between 1985 and 2024.



↑ **Figure 2.6.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.6.4.** Spatio-temporal distribution of benthic cover monitoring sites across Belize. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. The number of years with surveys is not equivalent to the timeframe covered by the surveys, thus 10 survey-years covers a 20 year timeframe with biennial sampling. For the purposes of this report, a site is also defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates have resulted in a lower "number of repeated surveys with data" in this figure compared to those reported in other publications based on the same dataset.



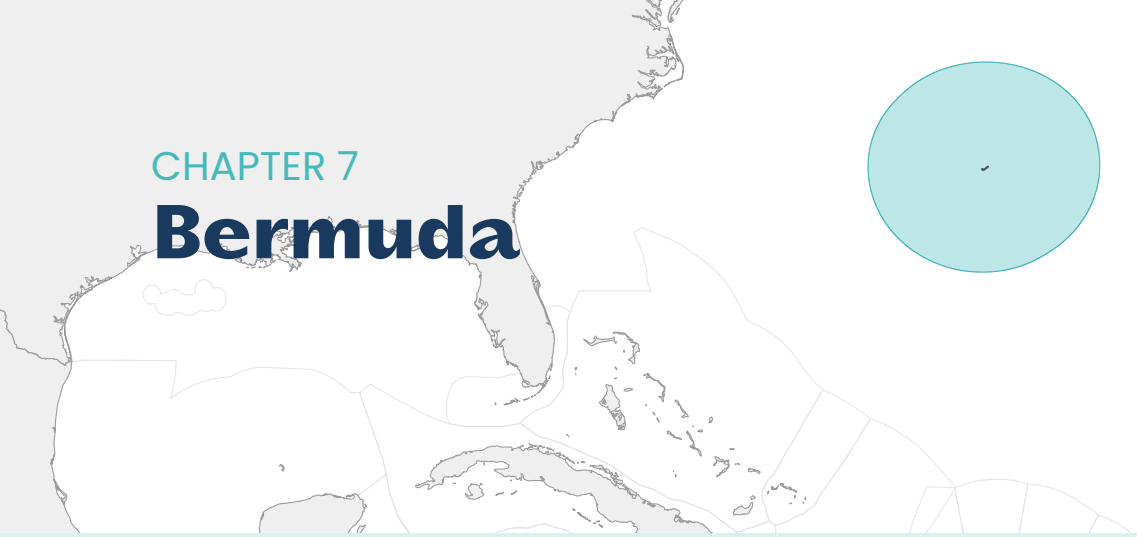
↑ **Figure 2.6.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Belize from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

The data (**Figure 2.6.5**.) depicted above show a general declining trend in coral cover since 1980 along with a general increase in macroalgal cover, generally aligned with the modeled data. The data show the decline of 1998, (going from ~28% to 15% cover) due to a combination of wave damage from Hurricane Mitch and the coral bleaching event (McField, 2002), accompanied by subsequent coral disease outbreaks that continued to reduce coral cover. This was followed by a period of relative coral stability in the 2000s, with the inclusion of the largest, representative reef survey ever conducted in Belize (in 2006), which covered 142 reef sites not chosen for strategic reasons but for their overall representativeness, therefore including more marginal, naturally lower coral cover reef habitats than the earlier studies. The next decade (2010-2020) was also relatively stable for coral cover despite some recorded coral mortality in the 2010 and 2015/17 coral bleaching events. The increase in more 'weedy' genera of corals masks some of the ecological deterioration to the large framework building corals (Orbicella's and Acropora's in particular) that play a critical functional role in reef ecology (McField et al., 2022). The last large sample of reef sites showed a decline from 18% (in 2021) to 15% (in 2023), likely due primarily to the combination of SCTLD and coral bleaching.

The macroalgae trends for both show a generally increasing trend until around 2012 when a period of relative stability ensued and even included some declines in total macroalgae cover (which includes the less palatable calcareous macroalgae). The decline in fleshy macroalgae reported in the 2024 Mesoamerican Reef Report Card, was even greater and likely due to the successful increasing biomass of parrotfish and surgeonfish after the 2009 nationwide full protection of these key herbivores (McField et al., 2024). The increased herbivory was likely countered by generally increasing nutrient pollution emanating from agriculture, and increasing population and developments within the coastal zone (the entire country is within the reef's watershed).

CHAPTER 7

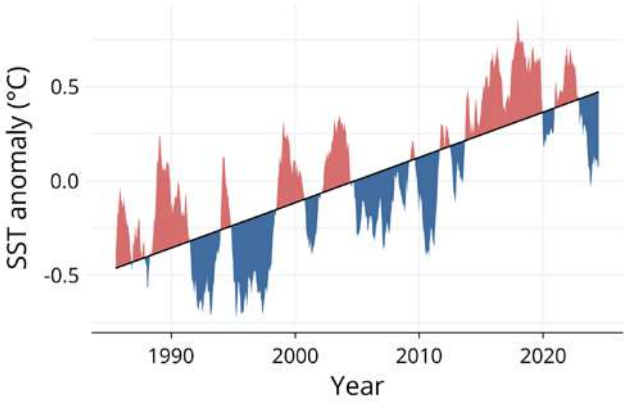
Bermuda



Authors: Struan Robertson Smith, Jérémy Wicquart.
Data contributors: Thaddeus Murdoch, Struan Robertson Smith.

INTRODUCTION

Bermuda, a British Overseas Territory in the North Atlantic Ocean, lies about 1,000 km east-southeast of Cape Hatteras, North Carolina (USA). The territory consists of numerous islands, the largest being Main Island, which hosts the capital city, Hamilton. Coral reef extent is about 672 km², which represents 2.7% of the Caribbean's total coral reef extent.



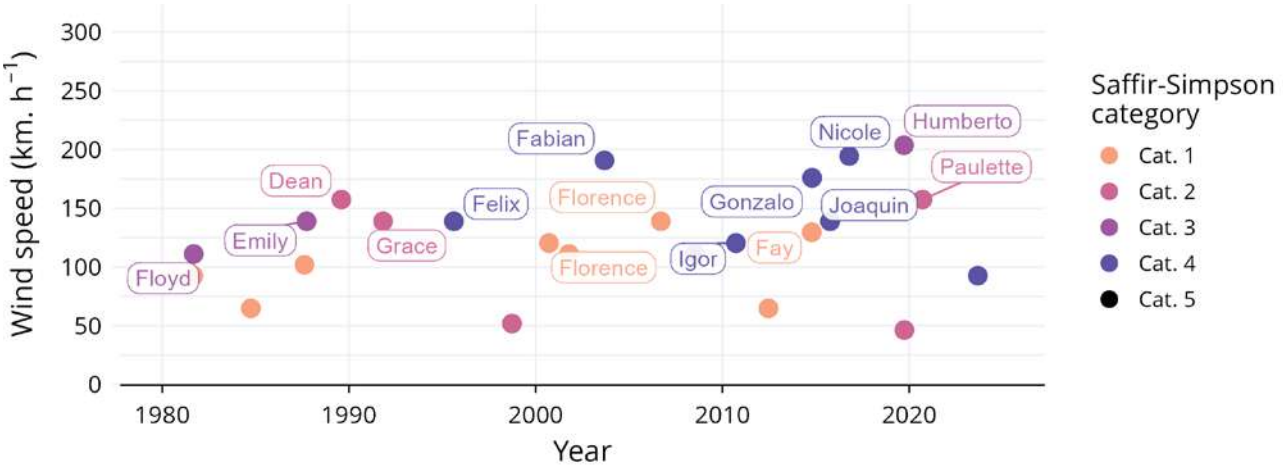
↑ **Figure 2.7.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Bermuda. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Bermuda was 23.30 °C. Over this period, SST increased by 1.10 °C, representing a warming rate of 0.028 °C per year (**Figure 2.7.1**).

Hurricanes

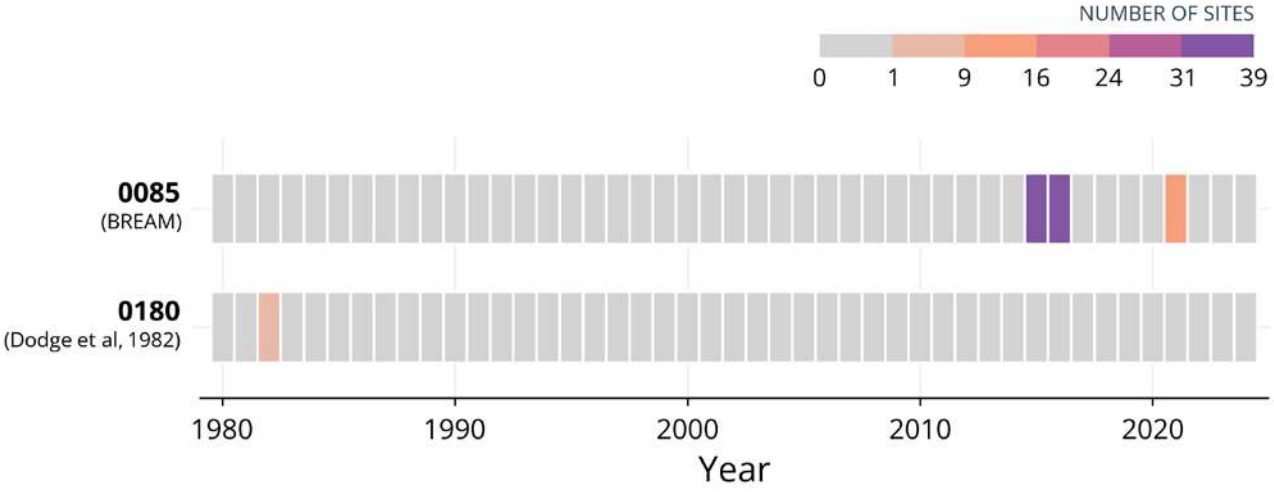


↑ **Figure 2.7.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Bermuda. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h⁻¹). Note that cyclones passing more than 100 km away from coral reefs of Bermuda are not represented, although they can have had an impact.

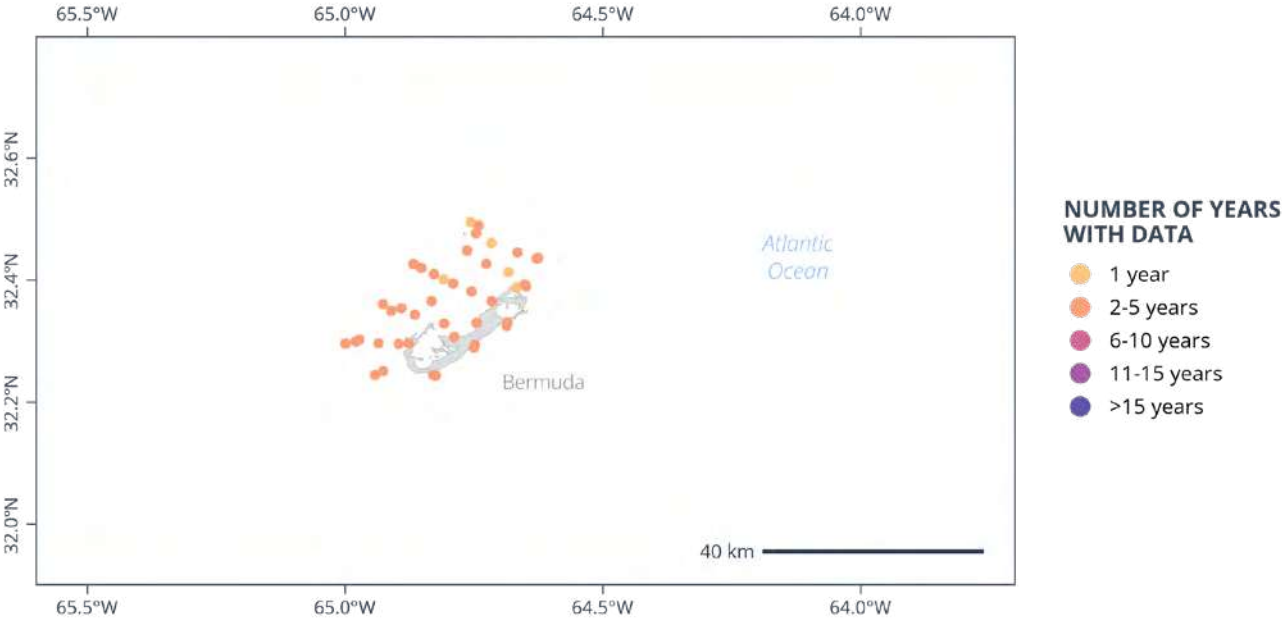
Between 1980 and 2024, a total of 36 tropical storms passed within 100 km distance from a coral reef of Bermuda, and of these 19 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.7.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Humberto, in 2019, which passed 77 km from the reef system, with sustained wind speed of 204 km.h^{-1}.

TRENDS IN BENTHIC COVER

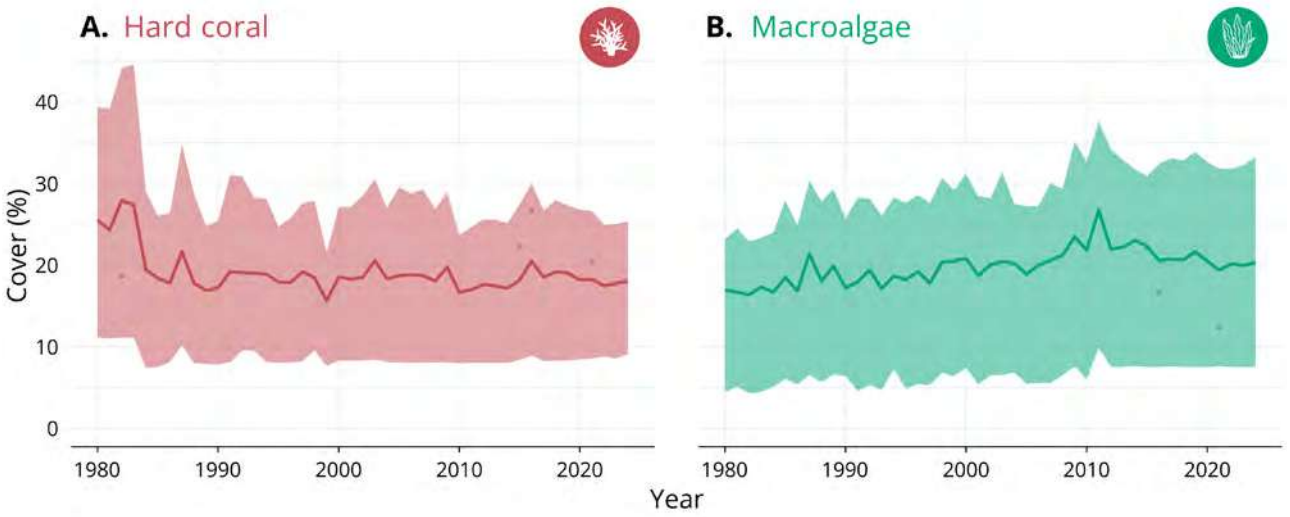
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.7.3**) encompassing 43 monitoring sites. These data were collected through 91 surveys conducted between 1982 and 2021. The entire BREAM dataset was not included to perform the analyses, although the first surveys were conducted in 2004.



↑ **Figure 2.7.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.7.4.** Spatio-temporal distribution of benthic cover monitoring sites across Bermuda. Sites that were monitored for the longest period of time are represented on top of the sites monitored for fewer years. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.7.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Bermuda from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

CHAPTER 8

Bonaire

Authors: Robert Steneck, Erik H. Meesters, Thomas Kemenes, Danielle De Kool, Matthijs Van Der Geest, Didier M. De Bakker, Ramon DeLeon, Laëtitia Mathon.

Data contributors: Rolf P.M. Bak†, Andrew W. Bruckner*, Didier M. De Bakker, Danielle De Kool, Kalli De Meyer*, Caren Eckrich, Graham J. Edgar, Tessa Haanskorf, Andreas F. Haas, Thomas Kemenes Van Uden, Philip Kramer*, Patricia R. Kramer*, Marnik Lehwald, Cindy Lott*, Erik H. Meesters, Gerard Nieuwland, Ernst Noyons, Maggy Nugues, Lizzi Oh, Tom Shyka*, Robert Steneck, Rick D. Stuart-Smith, Sanne Tuijten, Angelica Verschragen, Matthijs Van Der Geest.

*AGRRA data contributor(s).

INTRODUCTION

Bonaire, an island of 288km², hosts 68 km² of coral reefs, which represent 0.282% of the Caribbean's total coral reef extent. These reefs have been described and studied since the 1960s, making them one of the longest studied reefs in the region (Bak, 1977). The leeward coast of the island was mapped in detail by van Duyl (1985). Recently extensive reefs have also been mapped on the windward side of the island (Kemenes et al., 2024).

Tourism is the main ecosystem service the reef provides to the island. In 2022, Bonaire had a GDP of 662 million USD, half of which was generated through tourism (CBS, 2024). Two-thirds of the 182.000 visitors in 2024 came to Bonaire for vacation or diving. Fisheries on Bonaire consists primarily of artisanal subsistence fisheries on pelagic species, along with a handful of fishing charters.

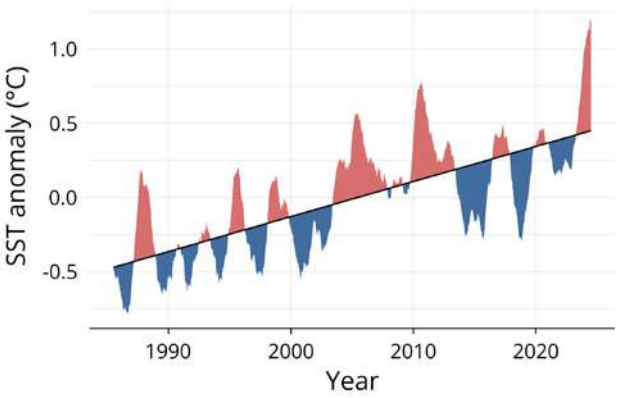
The reef is also important for recreational activities of the local inhabitants, and for coastal protection, mostly around the city of Kralendijk. Perry et al. (2025), however, show the coastal protection function to be seriously threatened by sea level rise.

The entire island is an MPA since spearfishing was banned in 1971 and parrotfish landings were prohibited since 2010, as well as fish traps. Since 1979 this protection has been recognized under the Bonaire National Marine Park protected area, which extends down to 60m depth along the coast. Protective measures have also included scuba-diving admission fees, no-dive zones, no-take zones, minimum catch sizes for lobster, the protection of sea turtles and sea turtle nesting areas, and a ban of coral harvesting.

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Bonaire was 27.47 °C. Over this period, SST increased by 1.06 °C, representing a warming rate of 0.027 °C per year (**Figure 2.8.1**).



↑ **Figure 2.8.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Bonaire. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Coral bleaching on Bonaire has grown more severe in recent years. The marine heatwave in 1990 led to significant bleaching on Bonaire's reefs (Meesters and Bak, 1993). In 2005, while severe bleaching was observed in many parts of the Caribbean, the reefs in Bonaire were relatively preserved (Eakin et al., 2010). After the massive bleaching event in 2010, no significant bleaching was detected on the reefs until 2016. Since then, yearly bleaching has become the norm with an average of 25% of coral colonies affected. In 2023 and 2024, 80% of the corals bleached, with ~20% average mortality (De Kool, Eckrich, Francisca, 2025).

Hurricanes

The southern Caribbean is relatively free from hurricanes. Between 1980 and 2024, a total of 9 tropical storms passed within 100 km distance from Bonaire, and of these, only Hurricane Felix was characterized by sustained wind speed greater than 100 $km.h^{-1}$, but didn't cause any damage to Bonaire's coral reefs.

Hurricanes Lenny (1999) and Omar (2008) were the most destructive storms of the study period in Bonaire, leaving severe and lasting impacts on coral reef ecosystems. Lenny was particularly unusual and damaging because it tracked from west to east, driving waves directly onto normally sheltered, westward-facing reefs. This resulted in widespread coral destruction, including toppling of massive colonies over 2–3 meters tall and centuries old, fragmentation of smaller corals, tissue damage, bleaching, and smothering from storm debris, drastically altering reef structure and ecosystem stability (Bries et al., 2004).

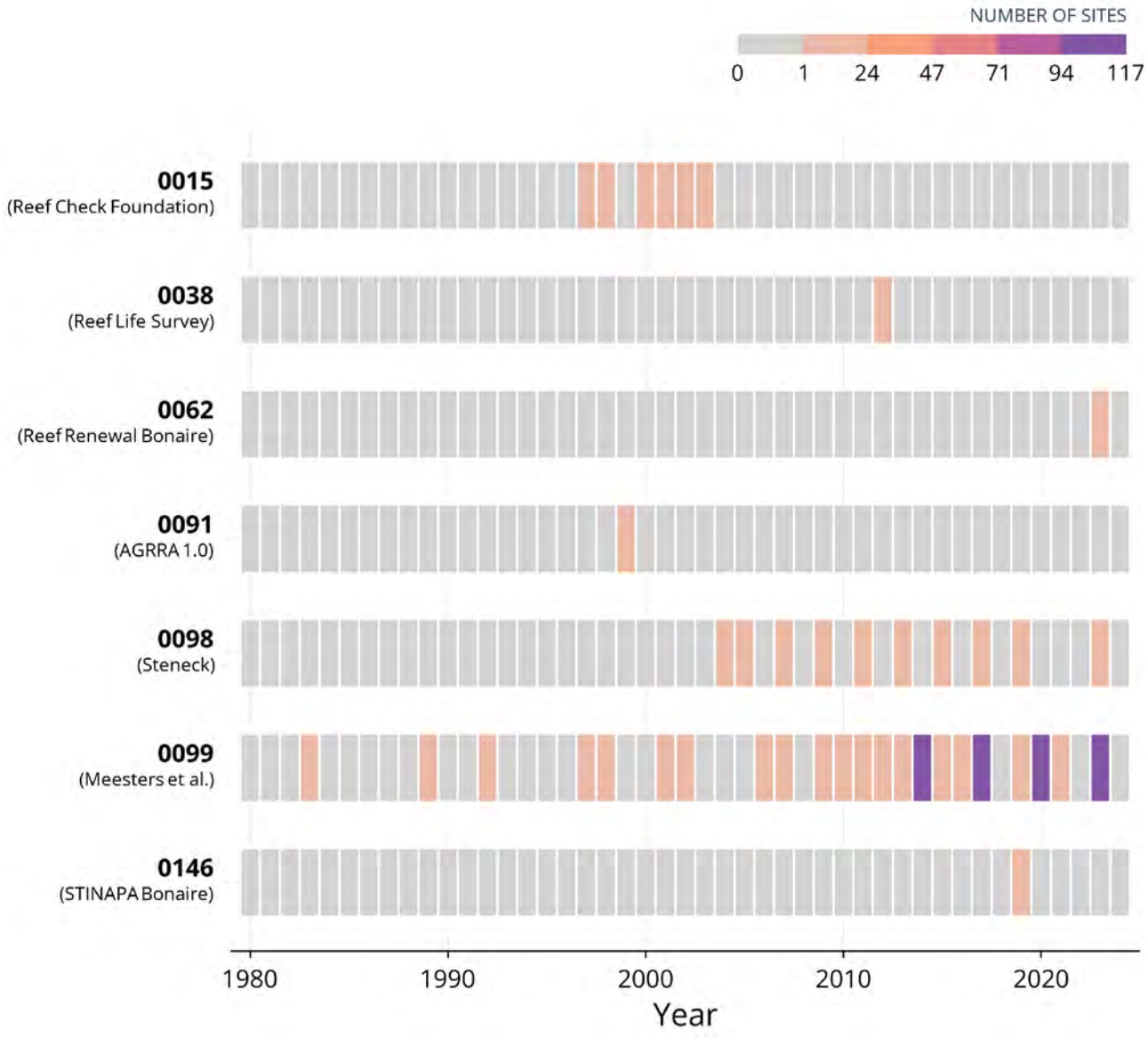
Other threats

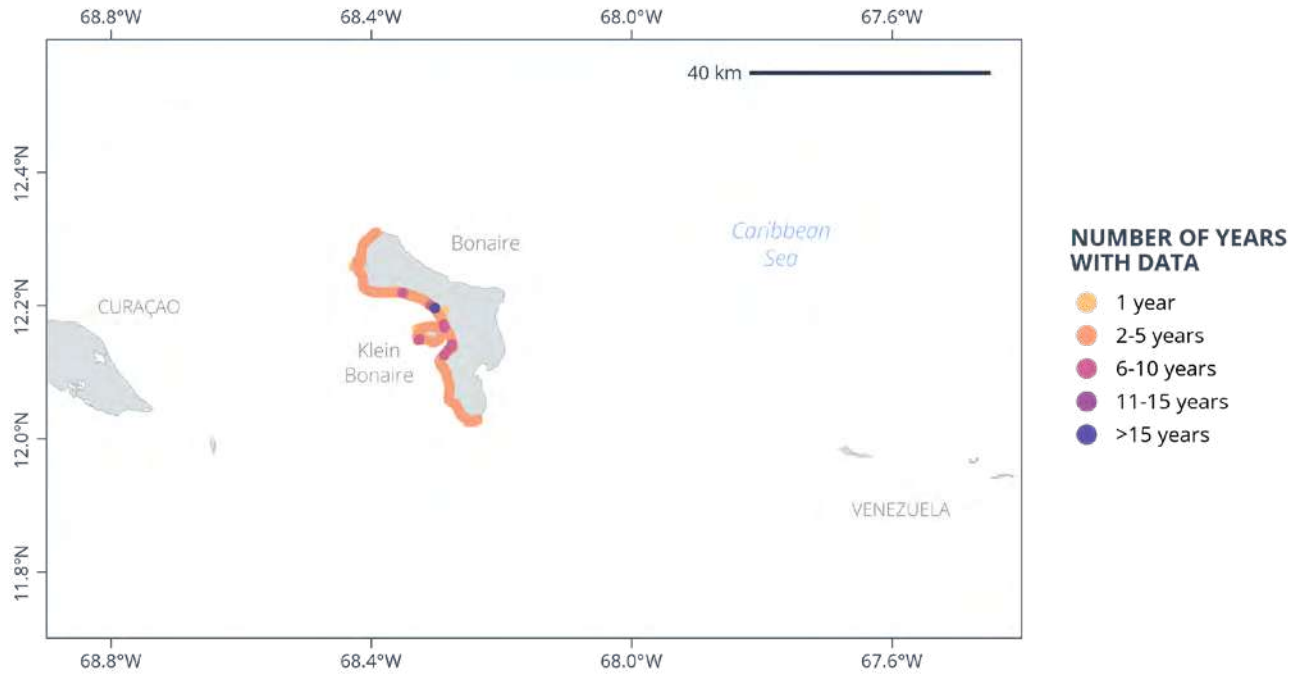
The main threats to the reefs of Bonaire nowadays are related to climate change (Perry et al., 2025), diseases (Pepe et al., 2024), illegal fishing, terrestrial runoff (Van der Geest et al., 2020), and water quality (Slijkerman et al., 2014). Coral cover already decreased strongly in the early eighties of the previous century (Bak and Nieuwland, 1995), when the white band disease decimated much of staghorn fields that dominated the southern part of the island (Bak and Criens, 1981; van Duyl, 1985) and mass mortality of the black sea urchin (Bak et al 1984) led to a strong increase in benthic algae (van Steveninck and Bak, 1986). In 2010 Bonaire became a special municipality of the Netherlands leading to a strong increase in population density. It has been estimated to have doubled from 12.000 in 2012 to 24.000 in 2024 increasing pressures on the natural environment. In the same time, the annual number of tourists and cruise ships steadily increased. While buildings, hotels, and houses on the coast are connected to a central sewage system, most other households are not and use a septic tank or cesspool. Groundwater wells contain high values of nitrate and ground water discharge and run-off during rain events are probably eutrophying the shallow reef zone (Van Der Geest et al., 2020). Dissolved Inorganic Nitrogen (DIN) concentrations orders of magnitudes higher than 1 micromol/l have been measured during the last 3 years (Meesters unpublished data).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 7 datasets (**Figure 2.8.2**) encompassing 154 monitoring sites (**Figure 2.8.3**). These data were collected through 618 surveys conducted between 1973 and 2023.

↓ **Figure 2.8.2.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

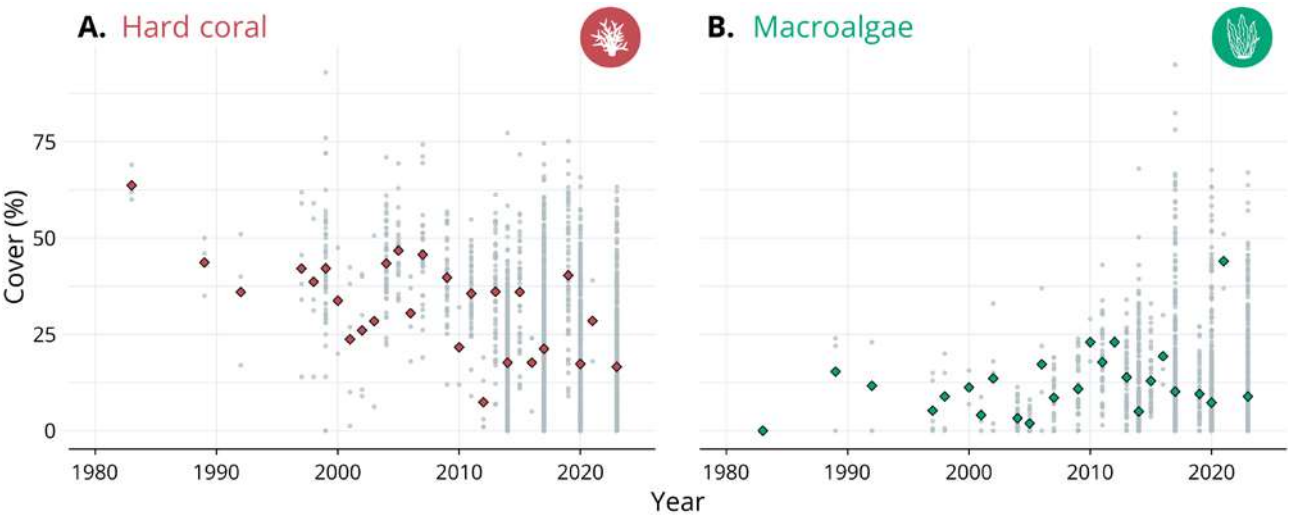




↑ **Figure 2.8.3.** Spatio-temporal distribution of benthic cover monitoring sites across Bonaire. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Bonaire's coral reefs have higher coral cover, lower macroalgal abundance and greater parrotfish abundance than average (Jackson et al., 2014, Steneck et al., 2019). Banning parrotfish spearfishing in 1971 contributed to this relative success. Moreover, in 2010, all parrotfish were protected from harvesting and fish traps were phased out. As a result of these management decisions, harmful macroalgae was kept cropped short and coral settlement was among the highest in the Caribbean (Arnold et al., 2010).

Finally, the thriving tourist economy resulted in a high per capita income (*i.e.* high GDP) and thus a lower "need" to fish. However, long-term trends (Bak and Nieuwland, 1995, Bak et al., 2005, de Bakker et al., 2017) and island-wide assessments show that the reef of Bonaire is now experiencing unprecedented low coral cover (14.1%, 95% CI [8.8% - 18.0%]) and reef growth in most locations is not keeping pace with predicted rates of sea level rise (de Bakker et al., 2019, Perry et al., 2025).



↑ **Figure 2.8.4.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Bonaire from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Case study n°1

A tale of two islands: Lessons from monitoring at fixed sites with the same researchers

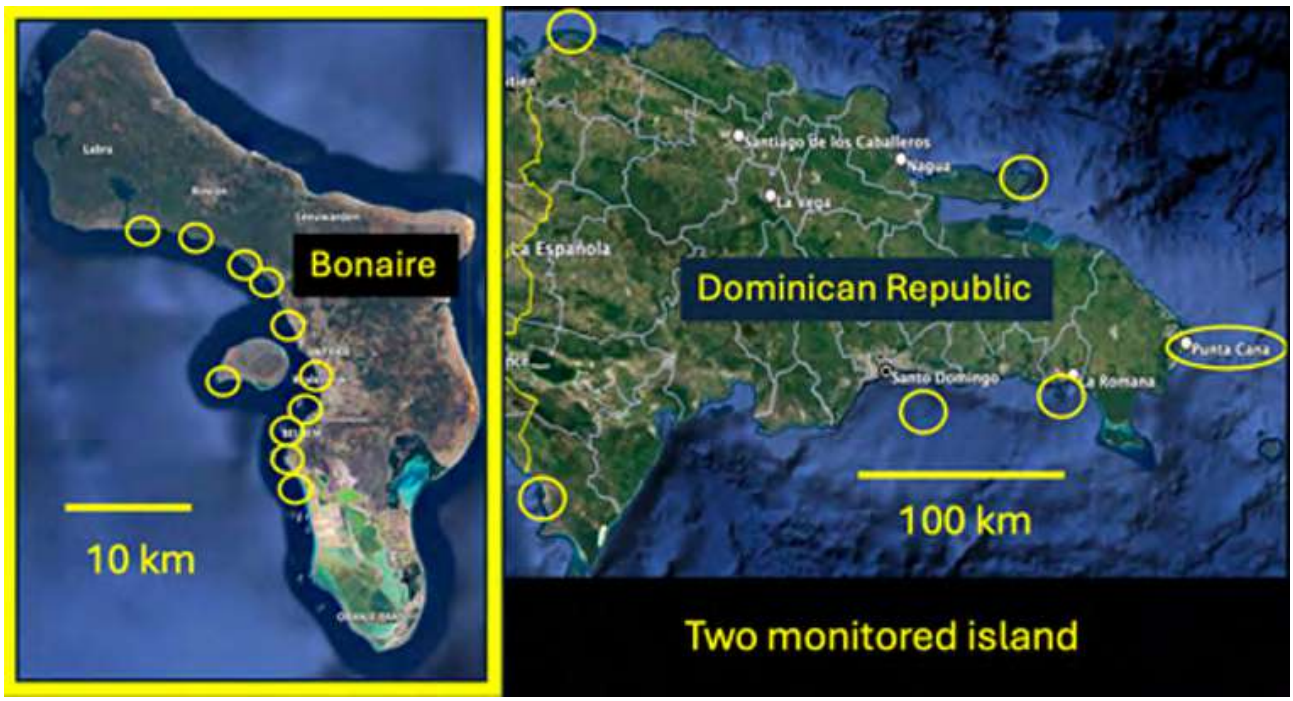
Authors: Bob Steneck[1], Doug Rasher[2], Ramón de León[3], Ruben Torres[4]

[1] University of Maine, School of Marine Sciences, Maine, USA
[2] Bigelow Laboratories for Ocean Sciences, Maine, USA
[3] REEF SUPPORT BV, Bonaire
[4] Reef Check DR, Dominican Republic

Comparing the status and trends of coral reefs is often complicated by the variance introduced by different investigators applying divergent methods. Programs such as AGRRA were developed to standardize assessments and monitoring efforts. However, variations in experience and training among assessors still create inconsistencies. In this study, we used fixed transect sampling to compare long-term trends in coral cover, macroalgae abundance, parrotfish biomass, juvenile coral density, and total fish biomass between Bonaire and the Dominican Republic (DR). Importantly, we employed identical methods and relatively few investigators to ensure consistency. Bob Steneck monitored corals and algae in Bonaire from 2003 to 2023, and in DR from 2015 to 2022. Ruben Torres monitored fish in DR, and Ramón de León conducted most juvenile coral surveys on both islands. In Bonaire, fish surveys were conducted by relatively few individuals over time. Separate reports were published for Bonaire and DR (Steneck et al. 2019; Steneck and Torres 2023), but direct comparison had not been made until now. The value of this analysis is that all trends were generated in situ, from replicate transects conducted consistently at the same sites and regions through time (**Figure 1**).

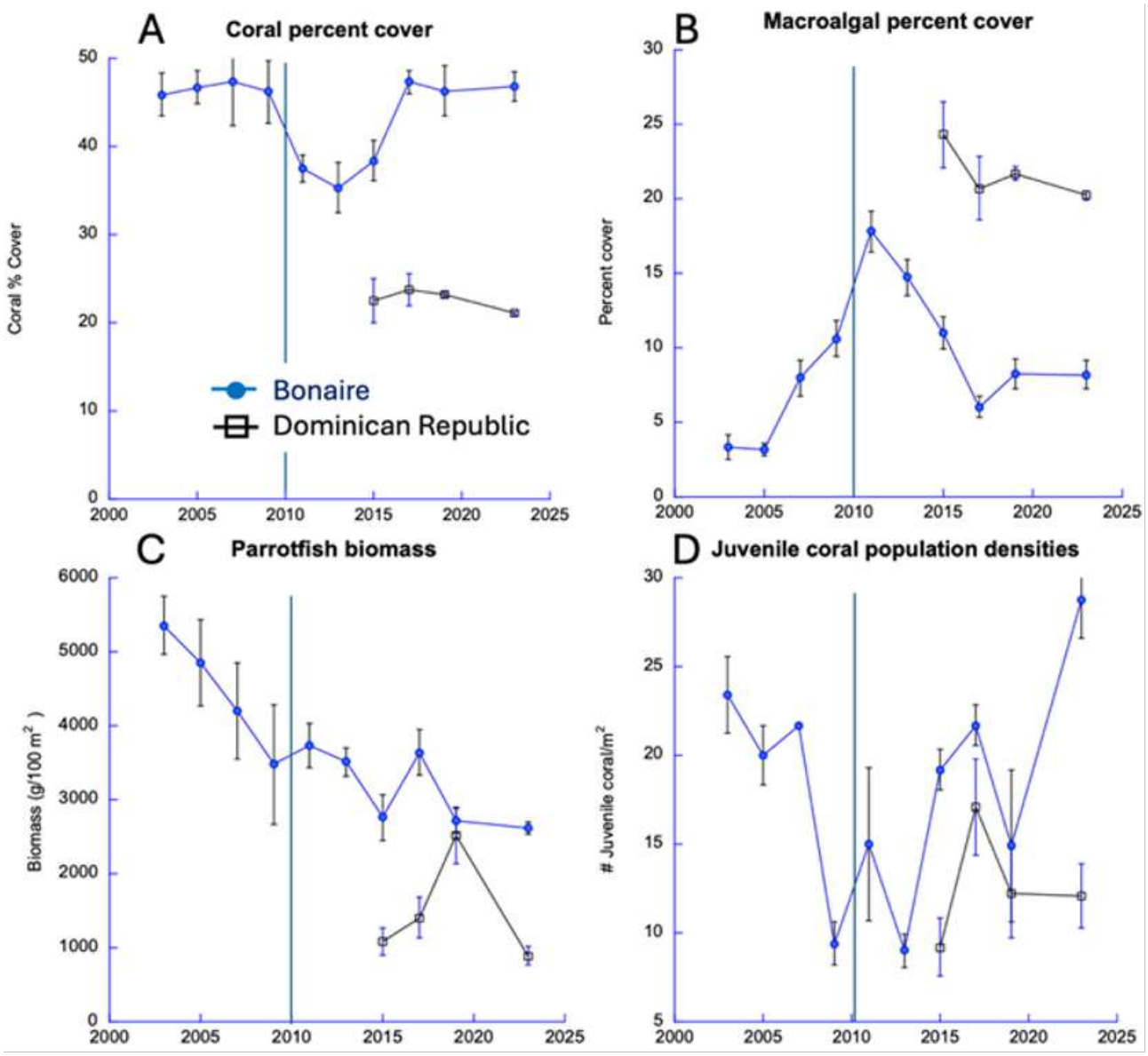
Although monitoring in Bonaire started earlier, the reefs of the two islands were distinctly different. Coral cover, parrotfish biomass, and juvenile coral densities were all higher on Bonaire's reefs, while macroalgae were significantly more abundant in DR (**Figure 2**).

The longer time series from Bonaire captured two major events in 2010. First, that year witnessed an intense coral bleaching event, causing widespread coral mortality. Second, the island implemented a complete ban on parrotfish harvest and phased out fish traps.



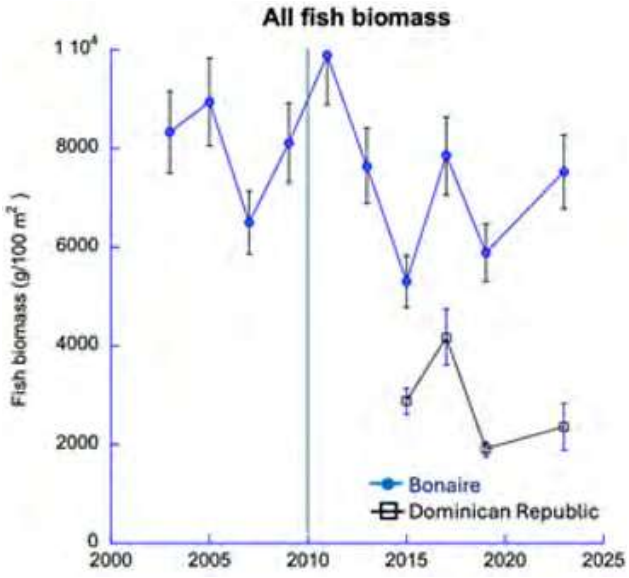
↑ **Figure 1.** Monitored reef sites (yellow circles) in Bonaire (left) and monitored regions with two sites per region in DR (right).

Management action to protect parrotfish was triggered by observed declines in parrotfish abundance (**Figure 2.C**) and increasing macroalgal cover (**Figure 2.B**). Semiannual monitoring reports had warned managers of troubling trends, calling for immediate action (Steneck and Arnold 2009). Coral cover declined steeply (**Figure 2.A**), reaching a low around 2013. Meanwhile, macroalgae peaked in 2011 and then steadily declined to pre-bleaching levels by 2017 (**Figure 2.B**). Parrotfish biomass stabilized at 2009 levels (**Figure 2.C**), while juvenile coral abundance increased, reaching record highs in 2023 (**Figure 2.D**).



↑ **Figure 2.** Comparisons of coral, macroalgae, parrotfish, and juvenile coral abundances in Bonaire (blue) and DR (black). Vertical line marks 2010 when Bonaire's coral reefs bleached and parrotfish protection began.

Larger geographic and socio-economic factors likely influenced observed patterns. Fishing pressure is significantly lower on Bonaire's reefs compared to DR, resulting in substantially higher fish biomass (**Figure 3**). In the Dominican Republic, there is greater reliance on fishing for food and income. Conservation measures are fewer and often poorly enforced. In contrast, Bonaire banned all spearfishing as early as 1971 when dive tourism became its primary economic driver. A reef usage tax funds STINAPA, Bonaire's reef management organization. The long-standing bans on spearfishing and parrotfish harvesting contributed to maintaining high herbivore populations.



↑ **Figure 3.** Biomass of all fishes across monitored periods at all sites in Bonaire and DR.

Geographic factors such as hurricane frequency and intensity also affect reef resilience. After disturbances like hurricanes or bleaching, algal abundance naturally increases, but the extent and duration depend on herbivory. Recently, the spread of coral diseases, notably Stony Coral Tissue Loss Disease (SCTLD), has worsened conditions across the region. Although both Bonaire and DR have experienced SCTLD outbreaks, Bonaire was infected later, limiting its impact on this analysis.

Consistent monitoring by trained observers provides the highest-resolution detection of true ecological trends. This study shows how timely management actions—such as banning parrotfish fishing and creating sustainable conservation financing—can significantly improve reef resilience. Protecting herbivores, enforcing conservation measures rigorously, and ensuring sustainable financing are critical to reef recovery, particularly after disturbances. Conversely, weak enforcement and high fishing pressure, as observed in the DR, hinder reef recovery. Moving forward, coral reefs urgently need stronger enforcement mechanisms, rapid responses to early warning signs, and proactive strategies to enhance resilience against climate change and coral diseases.

CHAPTER 9

British Virgin Islands

Authors: Graham Forrester, Argel Horton, Louise Anderson, Jérémy Wicquart.

Data contributors: Graham Forrester, Lonnie Kaczmarsky*, Nicole Kellum*, Brianne Loran*, Peiro Mannoni*, Richard Nemeth*, Jennifer Osborn*, Adam Quandt*, Laurie Requa*, Paige Rothenberger*, Mayra Suarez-Velez*.

*AGRRA data contributor(s).

INTRODUCTION

The British Virgin Islands (BVI), a British Overseas Territory in the Lesser Antilles, are located about 60 miles east of Puerto Rico and the U.S. Virgin Islands. The Territory is considered a large ocean state, composed of four main islands - Tortola, Virgin Gorda, Anegada, and Jost Van Dyke - along with more than 50 smaller islands and cays (**Figure 2.9.4**). Coral reef extent is about 266 km², which represents 1% of the Caribbean's total coral reef extent.

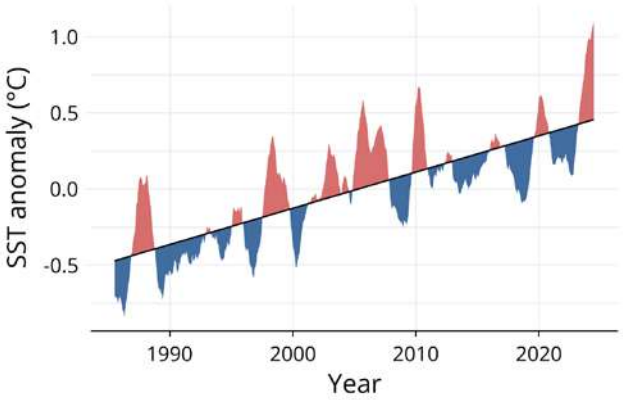
Coral reefs are vital for coastal protection, food security and biodiversity. They support artisanal fisheries and a vibrant tourism sector, valued at 125 million US dollars annually (Burke and Maidens, 2004). More than 80% of the Territory's gross domestic product depends on nature based sectors, yet reefs face increasing pressures from climate change, coral diseases and local impacts such as coastal development, anchor damage, sedimentation and declining water quality (Wilkinson, 2008).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of the British Virgin Islands was 27.49 °C. Over this period, SST increased by 1.08 °C, representing a warming rate of 0.028 °C per year (**Figure 2.9.1**).

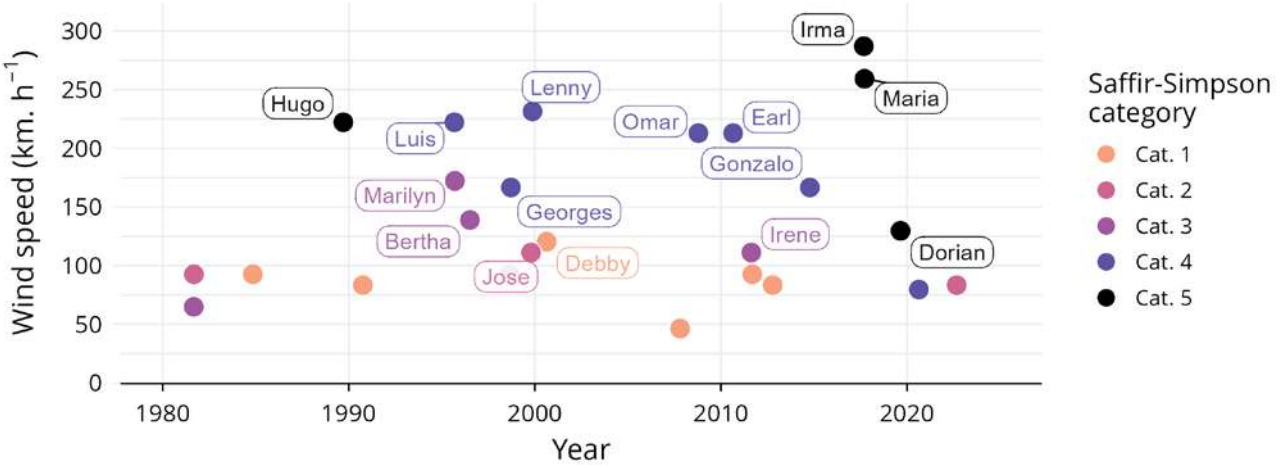
Bleaching events occurred in 1998, 2005, 2010 and most recently in 2023-2024 during the Caribbean wide heatwave. Species such as *Orbicella spp.*, *Colpophyllia natans* and *Acropora spp.* were affected to various degrees. Repeated thermal stress slows recovery and increases vulnerability to other pressures including coral diseases and hurricane damage (Hughes et al., 2018).



↑ **Figure 2.9.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of British Virgin Islands. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 33 tropical storms passed within 100 km distance from a coral reef of the British Virgin Islands, and of these 15 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.9.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed 0 km from a coral reef with sustained wind speed of 287 km.h^{-1}.



↑ **Figure 2.9.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in British Virgin Islands. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of the British Virgin Islands are not represented, although they can have had an impact.

Coral diseases

Stony Coral Tissue Loss Disease (SCTLD) was first detected in May 2020 at the RMS Rhone Marine Park, the Territory's only designated marine park. It spread rapidly to reefs around the Virgin Islands, causing mortality among framework building species including *Colpophyllia natans* and *Orbicella spp*. Disease infections advanced at 0.6-0.8 cm per day, often killing colonies within weeks to months (Forrester et al., 2024). Coral colonies such as *Dendrogyra cylindrus* suffered high mortality with live coral cover declining by up to 40% within a year at some monitored sites. SCTLD continues to transform BVI reefs by removing reef building corals, reducing structural complexity and facilitating the spread of algae, peyssonnelid algal crust and other opportunistic colonisers.

Peyssonnelid Algal Crust (PAC)

Peyssonnelid algal crust (PAC) is an emerging threat in the BVI. PAC overgrows hard corals, inhibits larval settlement and is potentially resilient to acidification and herbivory. Its spread is exacerbated by the low densities of the long-spined sea urchin (*Diadema antillarum*) which collapsed during the 1983 Caribbean wide die off (Lessios, 2016) and again in 2022 (Hylkema et al., 2023). Baseline surveys conducted across 18 sites found that PAC distribution is patchy with overall mean cover relatively low (5.65%) but localised hotspots exceeding 40% raised concern for the reef's resilience.

Boat anchoring

Boat tourism is a major contributor to the BVI economy (Everitt, 2007), with roughly 30 charter yacht companies and 1100-1500 charter yachts operating in BVI waters (CharterWorld, 2024; Forrester, 2020). Despite a network of moorings and regulations to protect coral reefs from boat anchoring, substantial impacts have been documented (Flynn and Forrester, 2019; Forrester et al., 2015). Roughly 25% of BVI reefs are subject to chronic yacht anchoring, and coral cover and species richness are reduced by half at these sites relative to equivalent sites where anchoring is rare (Flynn and Forrester, 2019). Larger mega-yachts (> 50m in length) are a growing threat because they are too large for moorings and can cause long-lasting and substantial damage in just one day. A one-off anchoring event in 2004 caused a 50% reduction in coral cover that persisted for over a decade (Forrester et al., 2015).

Overfishing

Artisanal fisheries in the BVI rely on small vessels using traps, handlines and diving for lobster, conch and reef fish as primary targets (Franklin, 2007). While lobster (*Panulirus argus*) and Queen conch (*Aliger gigas*) are regulated through closed seasons, minimal size limits and gear restriction and enforcement remains weak. In addition, illegal harvest persists within Territorial waters and within the wider Exclusive Zone. Speargun fishing is prohibited under the 2003 Fisheries law, yet spearguns are sold and used locally contributing to the depletion of reef fish such as parrotfish (*Scaridae*).

Parrotfish are critical grazers that prevent algal dominance and facilitate coral recruitment (Mumby et al., 2006; Butler et al., 2024). Larger species, including Stoplight parrotfish (*Sparisoma viride*) and Queen parrotfish (*Scarus vetula*) are particularly vulnerable to traps and illegal spearfishing. Their decline, together with the collapse of *Diadema antillarum* populations has reduced grazing pressure and accelerated the spread of macro-acroalgae and PAC. (Hylkema et al., 2023). Regional studies in the USVI demonstrate severe declines in herbivorous fish biomass due to overfishing, providing a precautionary parallel for the BVI (Kadison et al., 2017). The combined loss of herbivores undermines ecological resilience, limiting the capacity of BVI reefs to recover from bleaching, diseases, and other stressors (Alvarez-Filip et al., 2019).

Sargassum Influxes

Since 2011, seasonal influxes of pelagic *Sargassum* have become a major stressor in the BVI, particularly for shallow coral communities. When large mats accumulate in enclosed bays, it blocks light and deplete oxygen during decomposition, triggering mortality of corals, seagrasses, fish, and invertebrates (van Tussenbrock et al., 2017). In Handsome Bay, Virgin Gorda, located within a Fisheries Protected Area, severe hypoxia driven fish kills were recorded during the 2015-2018 strandings.

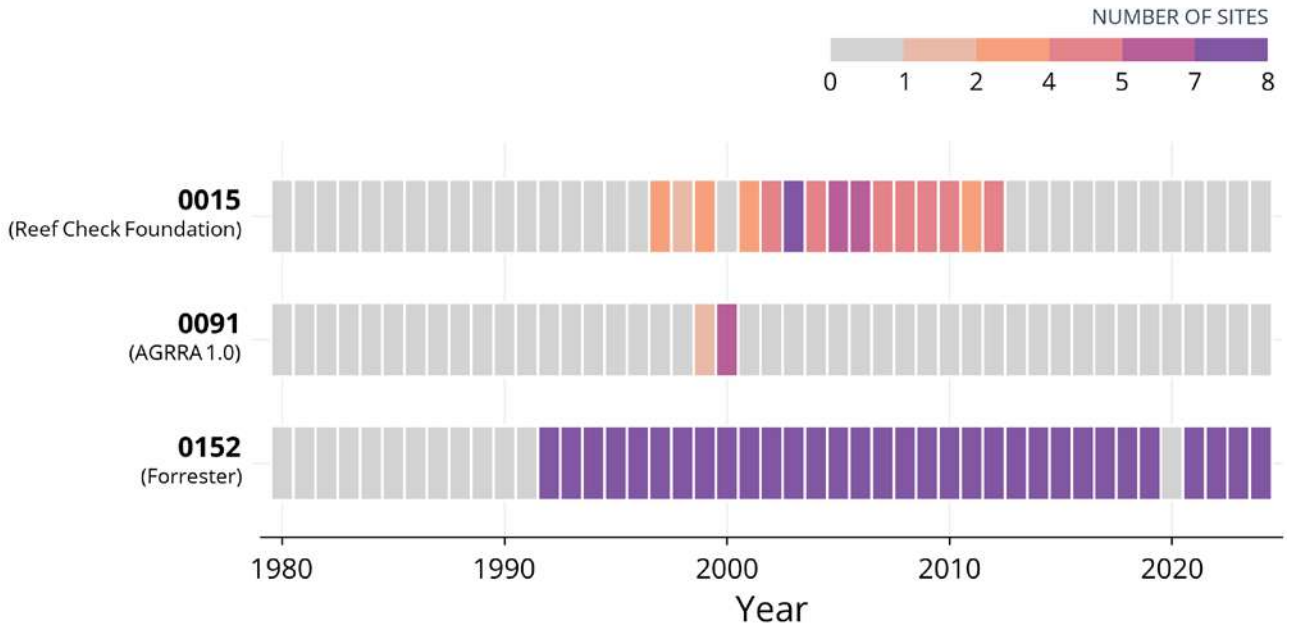
Pollution and Nutrient Runoff

Coastal development and urbanisation have placed increased pressure on nearshore waters. Sediment and waste from land, combined with untreated sewage contribute to eutrophication, promoting algal overgrowth and stressing corals (Burke and Maidens 2004). Wastewater from homes, tourism, and marinas adds further stress as treatment systems and enforcement remain limited. Recognising these risks, the Territory has developed a Water Quality Monitoring Programme to track nutrient and pollution levels in coastal waters, providing evidence to guide future management and planning.

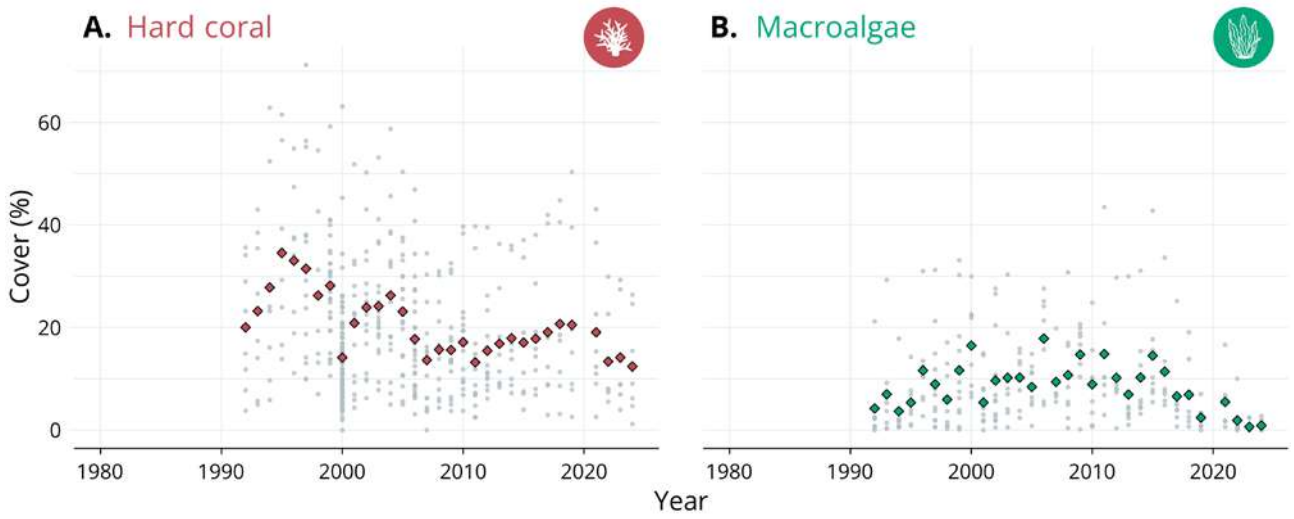
TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.9.3**) encompassing 34 monitoring sites. These data were collected through 328 surveys conducted between 1992 and 2024.

↓ **Figure 2.9.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.9.4.** Spatio-temporal distribution of benthic cover monitoring sites across British Virgin Islands. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.9.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in the British Virgin Islands from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 10

Cayman Islands

Authors: Gretchen Goodbody-Gringley, Jack V. Johnson, Laëtitia Mathon.

Data contributors: Graham J. Edgar, Gretchen Goodbody-Gringley, Robert Graifman*, Jerome Hall*, Casey Hermoyian*, Kaho Hoshino*, Jack V. Johnson, Carrie Manfrino*, Croy McCoy, Lizzi Oh, Bernhard Riegl*, Rick D. Stuart-Smith.

*AGRRA data contributor(s).

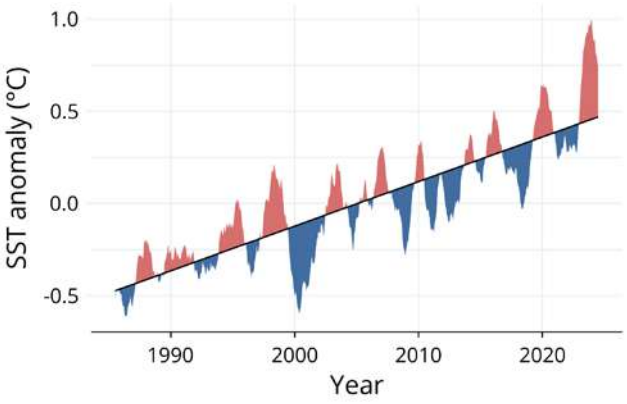
INTRODUCTION

The Cayman Islands, a British Overseas Territory located in the northwestern Caribbean Sea, comprise Grand Cayman, Cayman Brac, and Little Cayman. The islands are fringed by shallow reefs, with Grand Cayman and Little Cayman featuring extensive lagoonal systems behind linear, wave-resistant reef crests, while Cayman Brac's lagoons are less developed (Logan, 2013). Together, the islands host 231 km² of coral reefs, accounting for 0.953% of the Caribbean's total reef extent. Once sparsely populated, the Cayman Islands' population more than doubled between the 1960s and 1980s, rising from 8,500 in 1960 to 25,900 in 1988 (Turner et al., 2013), and has since grown to approximately 74,500 by 2024. Cayman Brac and Little Cayman, both small (<30 km²) and remote, remain relatively low in population, with limited fishing activity and no commercial fisheries; their Marine Protected Areas (MPAs), established in 1986, continue to support reef conservation (Dromard et al., 2010).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Cayman Islands was 28.17 °C. Over this period, SST increased by 1.05 °C, representing a warming rate of 0.027 °C per year (**Figure 2.10.1**).



↑ **Figure 2.10.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Cayman Islands. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

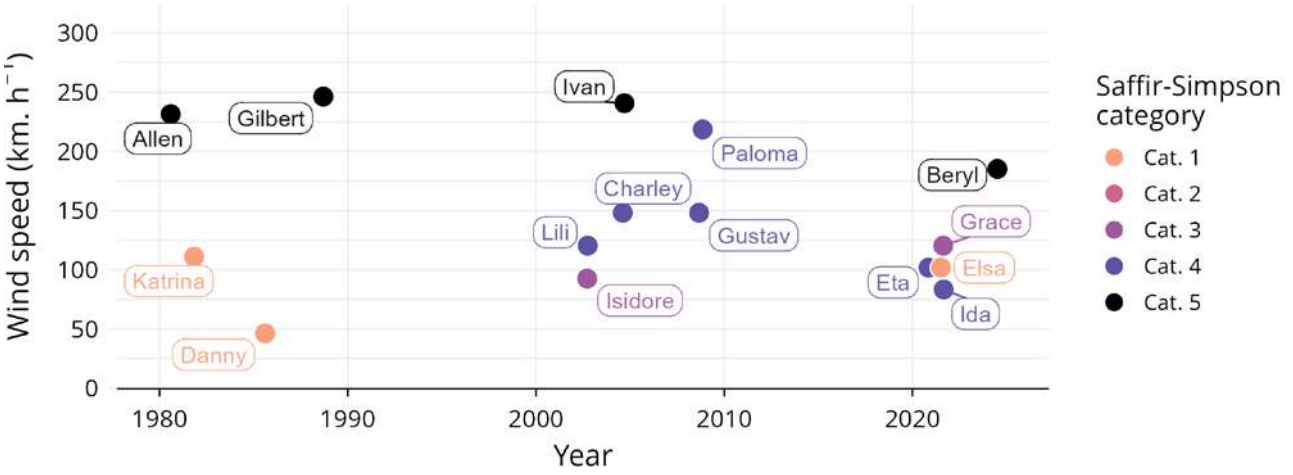
Coral reefs in the Cayman Islands have experienced multiple major bleaching events, with varying severity. The 1998 global bleaching event was the most acute on record, affecting ~90% of corals in Grand Cayman and causing ~10% mortality at two 9 m reef sites (Turner et al., 2013). Subsequent events in 2005 and 2009 caused widespread bleaching but relatively low mortality, though the latter was the most intense episode ever recorded at Little Cayman, severely impacting *Siderastrea siderea* and *Montastraea* spp. (van Hooidonk et al., 2012). In 2023, however, reefs faced unprecedented stress, with Little Cayman experiencing ~80% bleaching at peak, and ~54% coral mortality; shallow specialists species were most affected, while deeper and stress-tolerant taxa showed higher survival (Goodbody-Gringley and Chequer, 2025; Doherty et al., 2025).

Hurricanes

Between 1980 and 2024, a total of 23 tropical storms passed within 100 km distance from a coral reef of Cayman Islands, and of these 12 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.10.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Gilbert, in 1988, which passed 32 km from a coral reef with sustained wind speed of 246 km.h^{-1}. In June/July 2024, while category 5 Hurricane Beryl over Little Cayman caused some damage, it also cooled the water temperature and prevented dramatic bleaching from an ongoing marine heatwave.

Other threats

Coral diseases such as yellow band, white plague, and black band are among the most severe biological threats to coral reefs (Turner et al., 2013). Stony Coral Tissue Loss Disease (SCTLD) was first noted in the Cayman Islands in 2020 and is now also present in Little Cayman. The disease resulted in widespread and significant losses across a range of coral taxa, particularly in Grand Cayman. Human activities also drive significant damage: reefs near George Town harbour have been almost entirely destroyed by heavy marine traffic, including cruise ships. Johnson et al. (2023) revealed that reef fish populations were degraded due to human impacts, but recovered during COVID-19 lockdown. Meanwhile, coastal development has intensified nutrient and sediment runoff through land clearance, road construction, and loss of coastal vegetation, further degrading reef ecosystems (Turner et al., 2013).

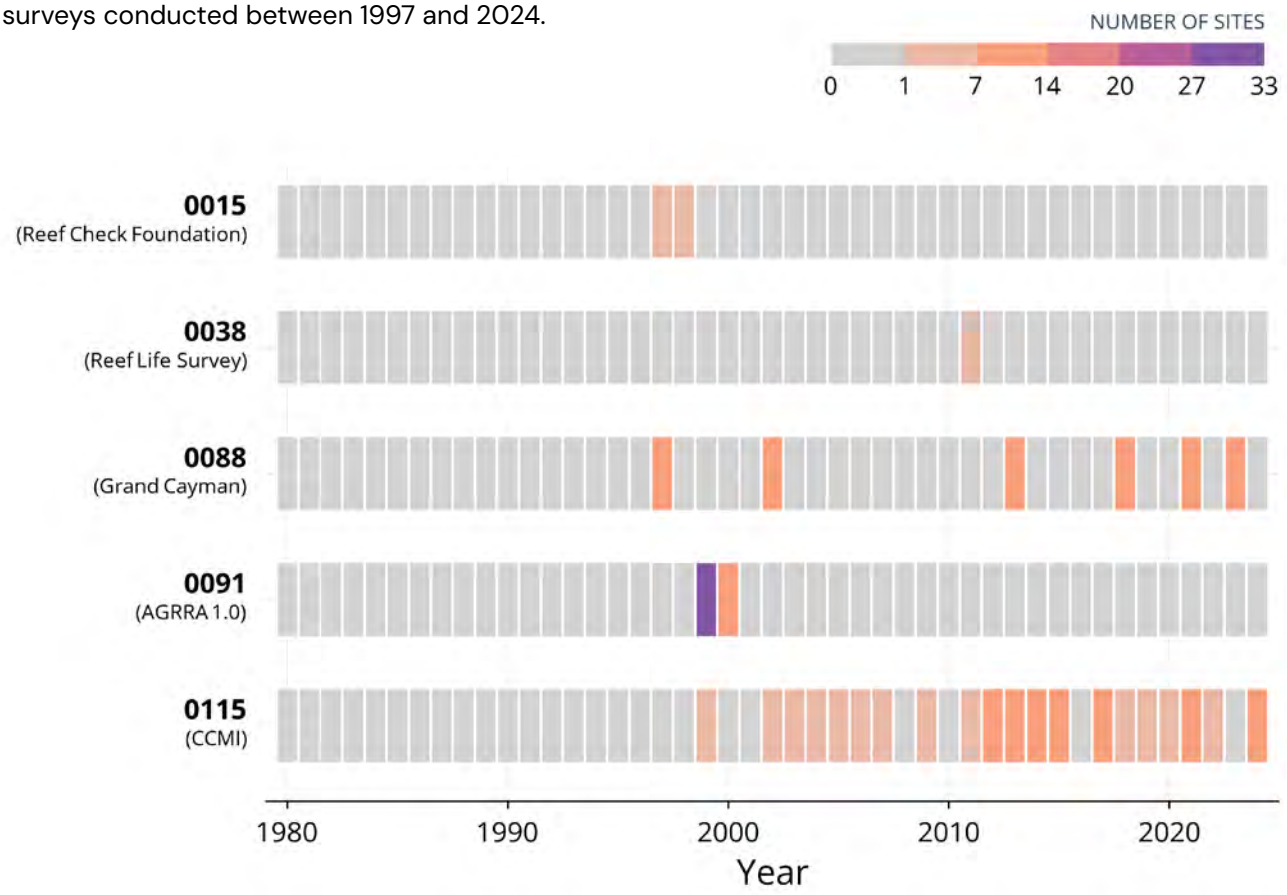


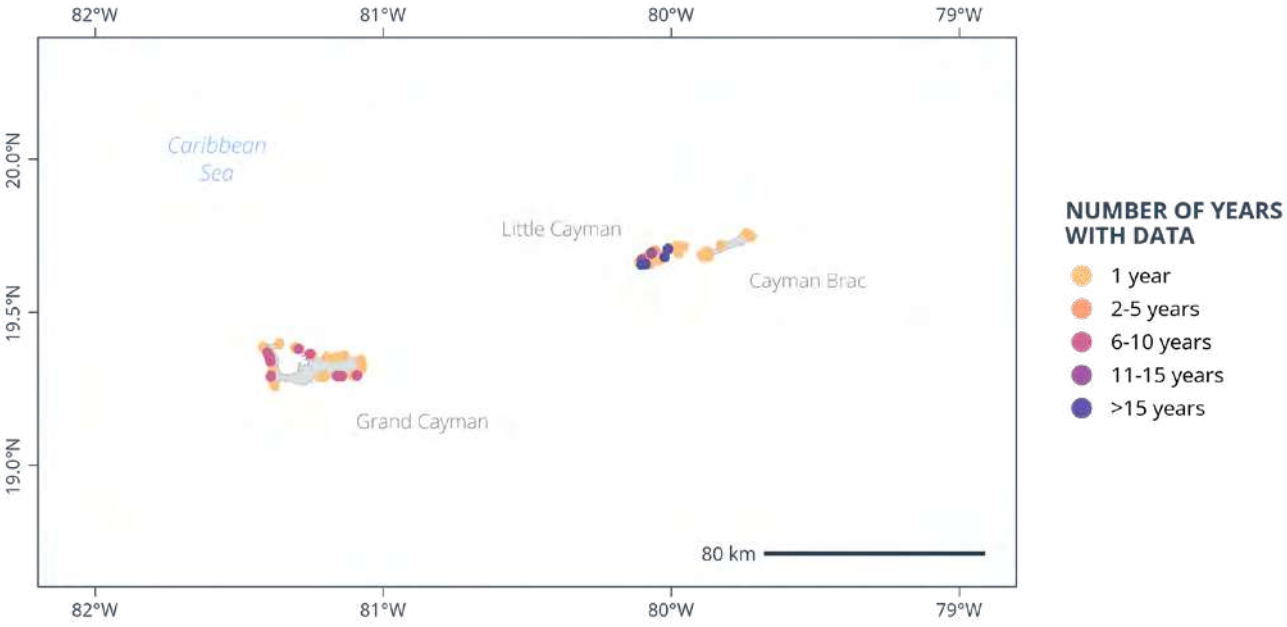
↑ **Figure 2.10.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Cayman Islands. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Cayman Islands are not represented, although they can have had an impact.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 5 datasets (**Figure 2.10.3**) encompassing 70 monitoring sites (**Figure 2.10.4**). These data were collected through 227 surveys conducted between 1997 and 2024.

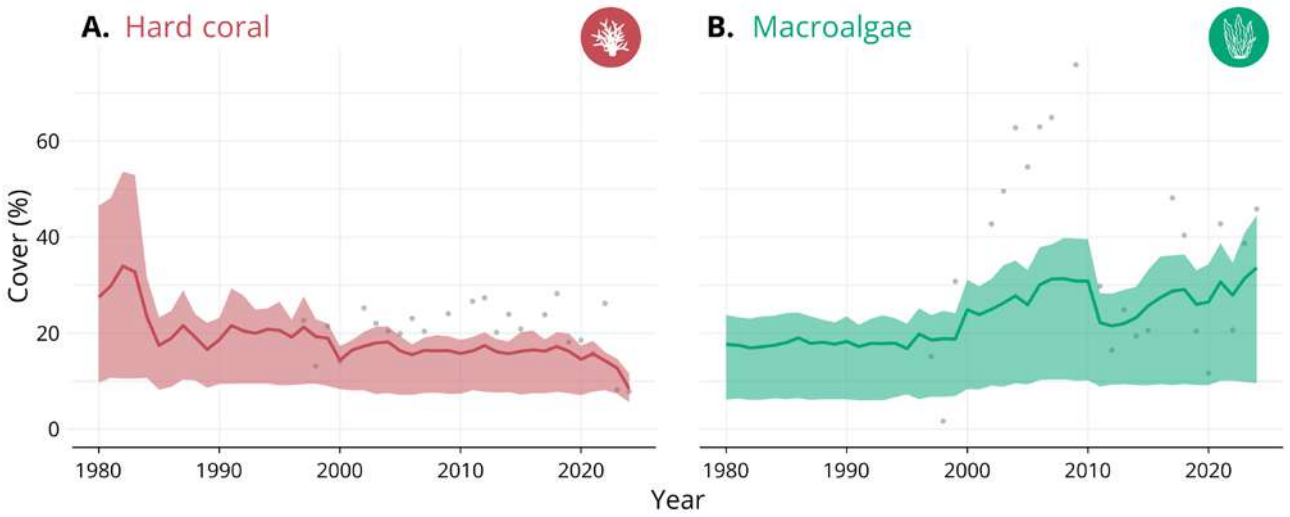
↓ **Figure 2.10.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.10.4.** Spatio-temporal distribution of benthic cover monitoring sites across Cayman Islands. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Trends in coral cover across the Cayman Islands reflect a dynamic balance between disturbance and resilience, with climate-driven stressors and disease emerging as dominant pressures. In the late 1990s, Little Cayman's reefs supported larger fish and appeared healthier than those of Grand Cayman, reflecting the negligible anthropogenic disturbance (Pattengill-Semmens and Semmens, 1999; Coelho and Manfrino, 2007). Despite protection of ~20% of its reefs by no-take reserves since the mid-1980s, monitoring from 1999 to 2004 revealed a 40% relative decline in live coral cover (26% to 16%), with similar losses inside and outside reserves, linked to disease outbreaks and the 2003 bleaching event (Coelho and Manfrino, 2007). By 2010, mean cover ranged from 21% at Little Cayman to 26% at Grand Cayman, but macroalgae dominated ~50% of reef benthos, while white plague disease caused extensive mortality, particularly of the reef-building *Orbicella annularis* (Bruckner, 2010). Encouragingly, Little Cayman showed signs of resilience: between 1999 and 2012, reefs that had declined to 14% cover recovered fully within seven years, with dominant coral species persisting despite temporal and spatial variation in size distributions (Manfrino et al., 2013).



↑ **Figure 2.10.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Cayman Islands from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

However, recent thermal stress events underscore the limits of this resilience. In 2023, Little Cayman experienced >17 Degree Heating Weeks, the most severe heat stress ever recorded, with ~80% of corals bleached at peak and ~54% mortality by the end of monitoring. Weedy taxa such as *Agaricia* spp. and *Porites* spp. were the most affected, while stress-tolerant builders like *Orbicella* spp. showed lower mortality, highlighting differential species vulnerability (Goodbody-Gringley and Chequer, 2025; Doherty et al., 2025).

CHAPTER 11

Colombia

Authors: Andres Felipe Acosta Chaparro, Juliana Vanegas, Valeria Pizarro, Andrea Rivera-Sosa.

Data contributors: Andrés Felipe Acosta-Chaparro, David Alejandro Alonso-Carvajal, Francisco Arias Isaza, Carlos Daza-Guerra, Diana Isabel Gómez López, Catalina Gómez-Cubillos, Katalina Henao Orozco, Pilar Herrón, Thomas Kemenes Van Uden, Raúl Navas-Camacho, Valeria Pizarro, Laura Sánchez-Valencia, Juliana Vanegas Gonzalez, Sven Zea.

INTRODUCTION

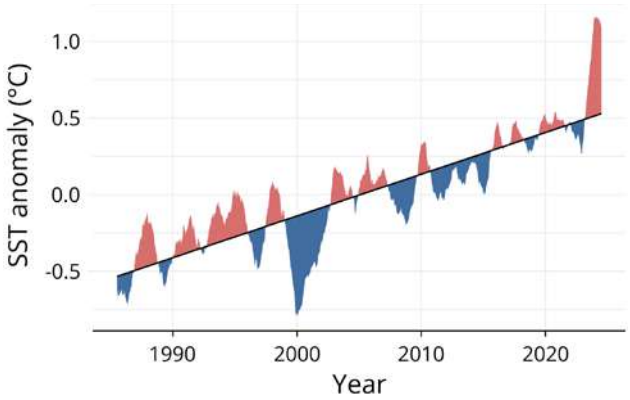
Colombia, located in the Southwest of the Caribbean, hosts an estimate of 494 km² of coral reefs, which represent 2.038% of the Caribbean's total coral reef extent. This shows a higher coral reef area in comparison to the Colombia coral atlas which estimated continental reef areas of 268 km² (Invemar and MinAmbiente, 2020). Coral formations show considerable variation in extent, type, and degree of development, and are widely distributed along the coast and continental shelf, from Cabo Tiburón in the southwest to Puerto López in La Guajira in the northeast.

On the continental shelf, these formations are mainly concentrated in the central zone, associated with topographic elevations produced by mud diapirism, located close to the shoreline (Díaz et al., 2000). Well-structured reefs are also present in the Nuestra Señora del Rosario and San Bernardo Archipelago, as well as along the coast of Tierra Bomba and Barú, where fringing reefs and coral patches predominate. Additional coral growth occurs in coastal areas where the alternation of bays and rocky cliffs creates favorable conditions, such as in the coral zones of Urabá Chocoano and the Santa Marta region (Invemar and MinAmbiente, 2020).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Colombia was 28.83 °C. Over this period, SST increased by 1.18 °C, representing a warming rate of 0.030 °C per year (**Figure 2.11.1**).



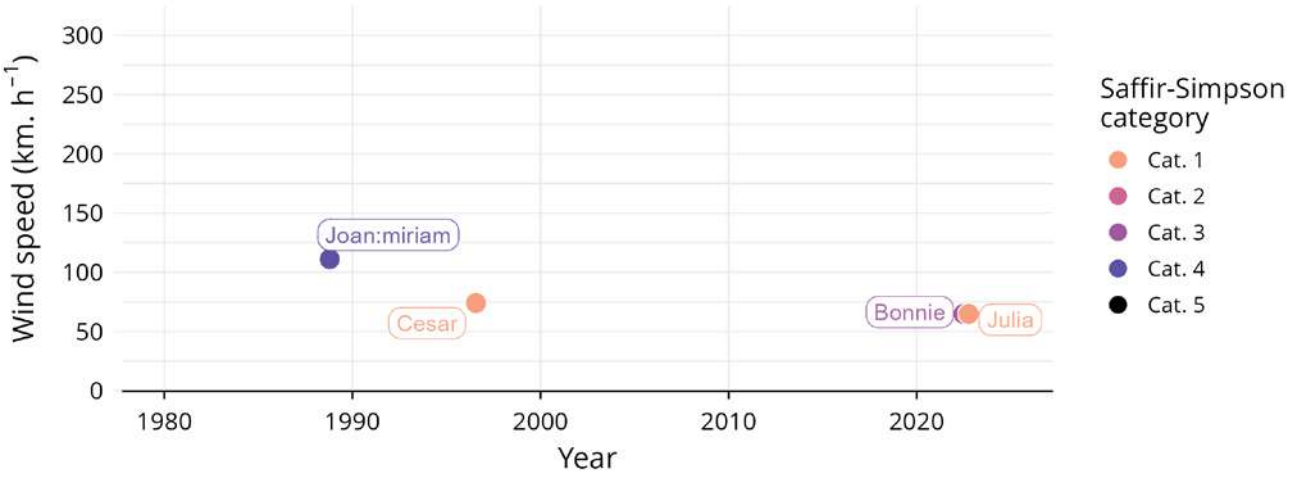
↑ **Figure 2.11.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Colombia. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

During 1997–1998, the increase in SST triggered significant bleaching events in localities such as Santa Marta and the Rosario Islands, severely affecting the main coral formations (Romero-Rodríguez et al., 2014). In 2005, additional bleaching events impacted all localities of the Colombian Caribbean. The effects were particularly severe in the Rosario and San Bernardo Islands, where approximately 70% of coral cover exhibited signs of bleaching and paling (Romero-Rodríguez et al., 2014), with an associated mortality of nearly 8% (Gil-Agudelo et al., 2005).

Other areas also experienced bleaching, but to a comparatively lesser extent (Wilkinson and Souter, 2008; Romero-Rodríguez et al., 2014). In 2023 and 2024, bleaching impact was severe across most localities of the Colombian Caribbean, reaching unprecedented levels. Although many results are still under publication and dissemination, data indicate that sites such as the Rosario and San Bernardo Archipelago, Varadero, and the Santa Marta region showed bleaching in more than 80% of coral cover.

Hurricanes

Due to its southern location, Colombia continental coral reefs are not usually affected by tropical storms and hurricanes. Between 1980 and 2024, a total of 9 tropical storms passed within 100 km distance from a coral reef of Colombia, and of these 1 was characterized by sustained wind speeds greater than 100 km.h^{-1} (**Figure 2.11.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Joan-Miriam in 1988, which passed 4 km from a coral reef with sustained wind speed of 111 km.h^{-1}.



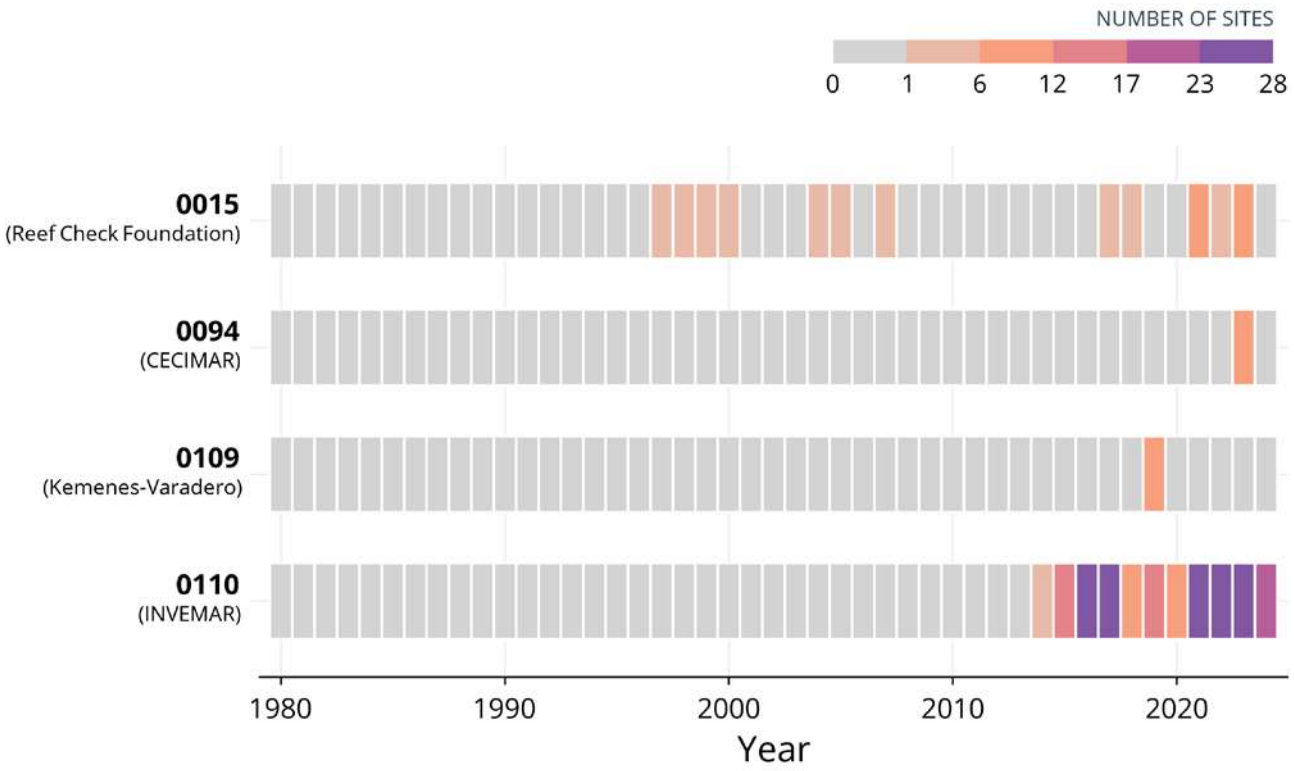
↑ **Figure 2.11.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Colombia. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Colombia are not represented, although they can have had an impact.

Other Threats

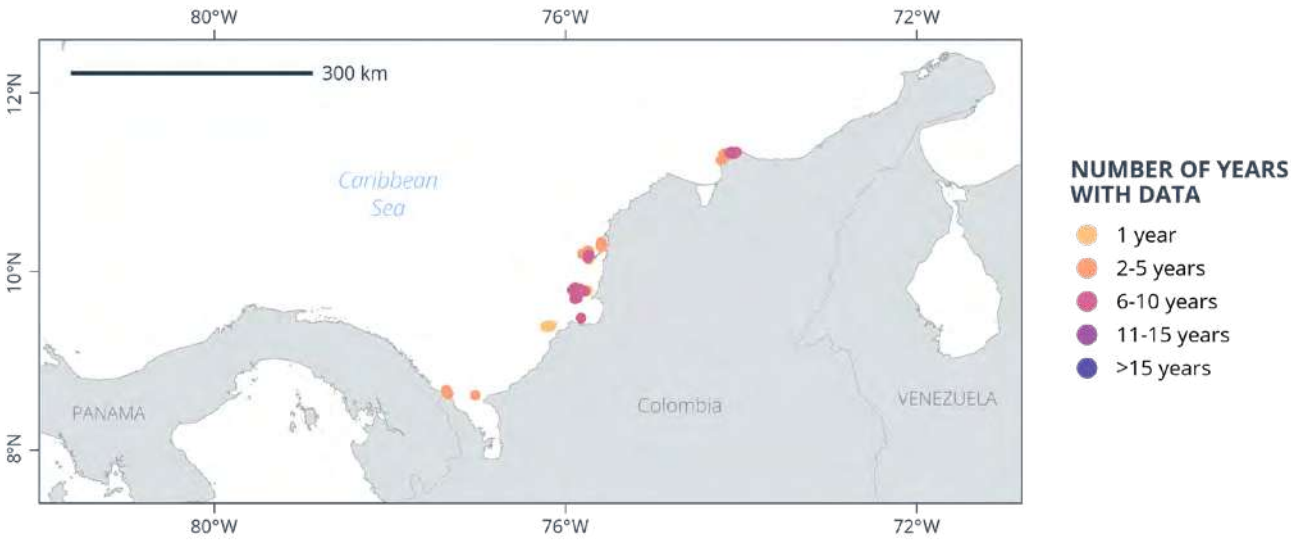
Port activity, demersal and artisanal fishing, nutrient inputs, marine traffic, invasive species, ocean acidification, and rising seawater temperatures have been identified as the main threats to continental Colombian coral reefs (Uribe et al., 2022). Coral communities and populations have been severely affected by these pressures, with bleaching events and disease outbreaks further exacerbating coral mortality, disrupting sexual reproduction, and reducing recruitment, ultimately diminishing reef resilience (Vega et al., 2025). Reported disease outbreaks include all major Caribbean coral diseases, such as White Band Disease, which caused high mortality in *Acropora* spp., as well as White Plague, Black Band, Yellow Band, and Dark Spot Disease, all of which have impacted reefs along the continental coast (Gil-Agudelo et al., 2004; Navas-Camacho et al., 2010, Sánchez et al., 2010). Fortunately, Stony Coral Tissue Loss Disease (SCTLD) has not yet been detected in these continental reefs, and measures can be taken with time. Disease outbreaks and mortality events affecting Diadema (long-spined black sea urchin) and the common sea fan are also threats. The combined effects of local stressors and global stressors not only threaten the survival of corals but also compromise the ecosystem services they provide (*e.g.*, fisheries, coastal protection, and tourism), further weakening the resistance and resilience of Colombian coral reefs.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 4 datasets (**Figure 2.11.3**) encompassing 90 monitoring sites. These data were collected through 258 surveys conducted between 1997 and 2024.



↑ **Figure 2.11.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

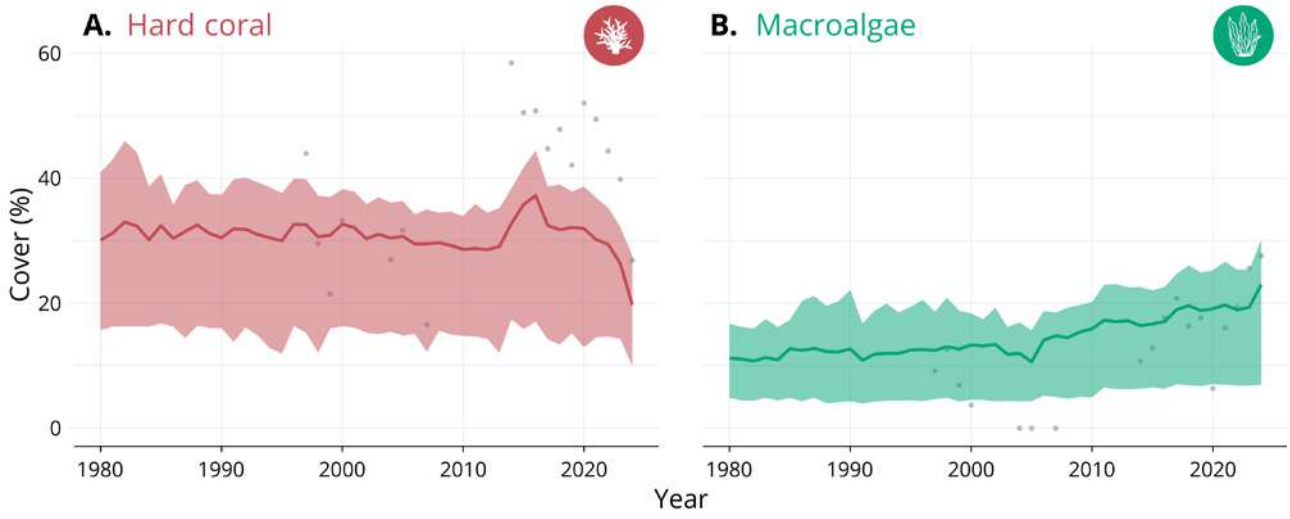


↑ **Figure 2.11.4.** Spatio-temporal distribution of benthic cover monitoring sites across Colombia. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

The temporal trend (**Figure 2.11.5**) shows a rapid decline in the coral cover, especially in the last decade, along with a significant rise in macroalgae and turf algae cover, suggesting a concerning shift in community structure and ecosystem health. Nuestra señora del Rosario and San Bernardo Archipelago are one of the areas with higher extensions of coral reefs in the country but also one of the most affected ones by the high macroalgae cover (40-60%) (INVEMAR, 2025). For the Tayrona National Natural Park, the challenges not only include the algae increase but also the decline in carnivorous fish (INVEMAR, 2025). Coral bleaching also plays an important role in the trend observed. In 2005, widespread coral bleaching events affected all major reef areas of the Colombian Caribbean with approximately 70% of coral cover exhibiting signs of bleaching in the Nuestra señora del Rosario and San Bernardo Archipelago (Gil-Agudelo et al., 2005). More recently, during 2023 and 2024, coral bleaching reached unprecedented levels across the majority of the reefs present in the Colombian Caribbean, with impacts also recorded in the Pacific.

In the Colombian Pacific, although some loss of live coral cover has been attributed to the exposure of shallow-water corals, additional stressors have also been identified. These include physiological bleaching and coral diseases resembling White Plague and White Band (Navas-Camacho et al., 2010).

The **Figure 2.11.5** shows an increase in the percentage of coral cover between 1980 and 1990. That increase was due to the expansion of the national reef monitoring system (SIMAC acronym in Spanish) and does not reflect an increase in coral cover in Colombia. Between 2010 and 2015, another increase in coral cover percentage was observed. However, these changes should be interpreted with caution, as they may be influenced by a shift in monitoring methodology from the chain transect method used by the Caribbean Coastal Marine Productivity (CARICOMP) methodology to the point intercept method.



↑ **Figure 2.11.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Colombia from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

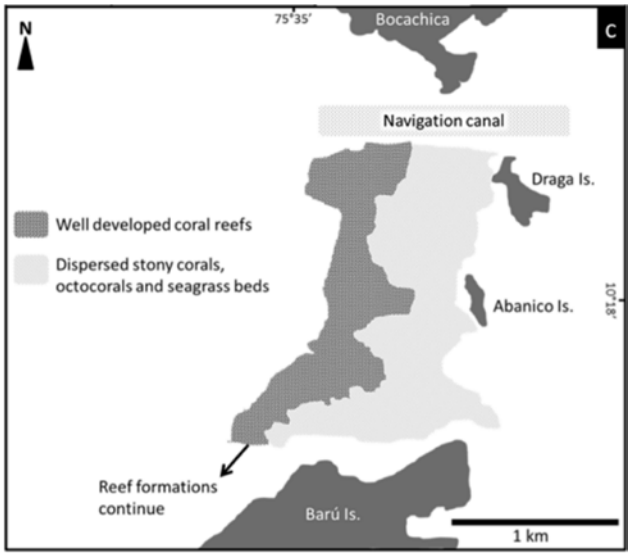
Colombia's Paradoxical Varadero Reef

Authors: Thomas Kemenes van Uden[1], Juliana Vanegas[2], Valeria Pizarro[2,3]

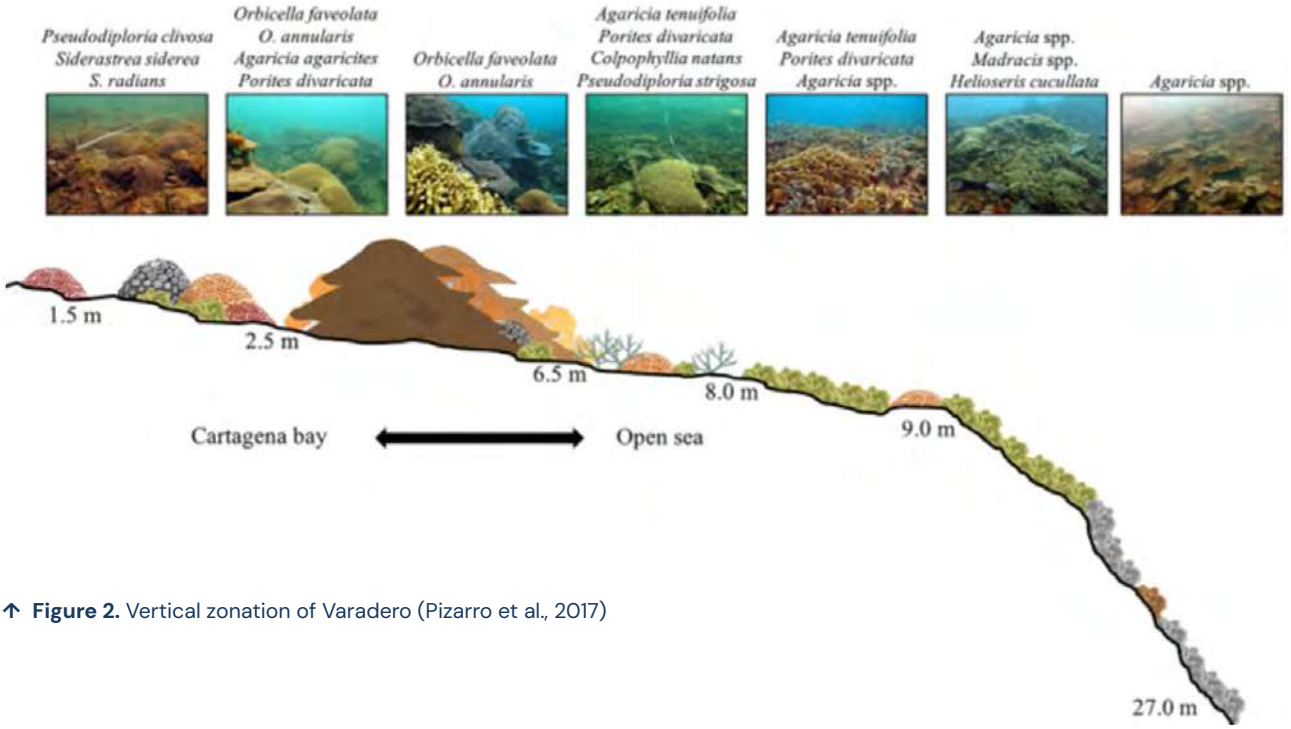
[1] Wageningen Marine Research, Ankerpark 27, 1781 AG Den Helder, The Netherlands
[2] Perry Institute for Marine Science, 5356 Main Street, Suite 2, P.O. Box 435, Waitsfield, VT 05673, USA
[3]ECOMARES Foundation for Marine Biological Research and Conservation, Cali, Valle del Cauca, 760050, Colombia

Varadero Reef, located on Colombia's Caribbean coast, was first described in 2017 (Pizarro et al., 2017). This reef was found thriving at the edge of Cartagena Bay, a highly polluted area impacted by the Magdalena River. Throughout the 1900s the Magdalena River was connected to Cartagena Bay to accommodate the traffic of vessels. The resulting discharge led to the decline of seagrass beds and coral reefs within the bay due to sedimentation and nutrient pollution (Tosic et al., 2019). The Magdalena River releases around 8,800m³/s of freshwater and 1.9 million tons of sediment annually into Cartagena Bay, resulting in elevated levels of nitrate, phosphate, turbidity, biological oxygen demand, heavy metals, hydrocarbons, and pesticides (Tosic et al., 2019). Yet, Varadero supports one of the highest coral covers in the Caribbean (Pizarro et al., 2017).

Varadero extends roughly 1.12 km² and exhibits two main zones (Pizarro et al., 2017): a well-developed reef dominated by scleractinian corals and a carbonate terrace with scattered hard and soft coral, and seagrass (**Figure 1**). The reef extends from 3 to 15m depth, with approximately 45% live hard coral and 48% algae-covered dead coral (Pizarro et al., 2017). The calcareous reef framework slopes down to 27–35m at 45°, transitioning to a sandy seabed interspersed with sponges and black corals (Antipatharia). Coral cover from around 10 to 35 m depth declines to just 1–5% (**Figure 2**). At about 10m depth, the benthos is dominated by Agaricia tenuifolia, with coral cover around 40–45%. In shallower zones Orbicella spp. dominate, many over 5m in diameter (Pizarro et al., 2017). Coral cover peaks at 50-60% at 6m depth. Overall, Varadero harbours 42 scleractinian coral species - including scattered colonies of *Acropora palmata*, *A. cervicornis*, and *A. prolifera* - as well as four hydrocoral species, 38 sponge species, and 61 fish species.



↑ **Figure 1.** Main zones of Varadero Reef (Pizarro et al., 2017)



↑ **Figure 2.** Vertical zonation of Varadero (Pizarro et al., 2017)

Following its initial description, studies on abiotic conditions, coral physiology and microbiomes, and herbivory patterns have been carried out to understand the reef's resilience. The size of Orbicella colonies suggests that the reef's state predates the diversion of the Magdalena River. Varadero's persistence contrasts with the Caribbean long-term average hard coral cover of 23.1%.

One major influence of the Magdalena River plume on Varadero is the alteration of the light regime. Because the bay's northern entrance is closed, the river outflow forms a surface layer around 1m thick that traps sediments above the reef. This stratified water column isolates the reef from direct exposure to the plume but significantly reduces light penetration. The light regime at 4m depth at Varadero is comparable to that at 12m at a nearby clear-water reef at the Rosario Islands (Tosic et al., 2019). Reduced light availability is thought to drive Varadero's compressed vertical zonation. Hard coral communities are virtually absent below 12m depth (Tosic et al., 2019).

However, low light conditions may benefit the coral community at shallower depths by reducing algal overgrowth and bleaching stress (Tosic et al., 2019). Symbiont communities in Orbicella colonies at Varadero are associated with low-light environments (Tosic et al., 2019) and the prevalence of bleaching on Varadero has been reported as five times lower than at the Rosario Islands (López-Londoño et al., 2024). Unfortunately, there is no data from either site regarding the 2023-2024 bleaching event to confirm whether this resistance still holds.

Low-light conditions may also reduce the need for herbivory. Herbivore biomass, size structure, and grazing rates at Varadero are notably low compared to the Rosario Islands. Parrotfish biomass at Varadero was estimated at 278g/100m^2 vs. 910 g/100m^2 at the Rosario Islands (Shantz et al., 2024). Despite relatively low herbivore biomass, algal cover at Varadero remains does not overgrow coral and is dominated by Halimeda species, in contrast to degraded reefs with high proliferation of turf and fleshy algae (*e.g.*, Lobophora and Dictyota).

Local Afro-Caribbean communities have traditionally relied on the ecological services provided by Varadero, and the reef's broader ecological significance is reinforced by high genetic connectivity with other Colombian reefs (Alegría-Ortega et al., 2021). However, Varadero faces multiple threats. Maintenance dredging of an existing navigation canal, and proposals to dredge a second canal directly through the reef, have raised concerns about increased sedimentation and chemical pollution. Strong stakeholder engagement led to the designation of Varadero as a «Hope Spot» by Mission Blue. However, efforts to secure national protection have been challenging and remain ongoing. Achieving formal protection and continuing to study the reef's unique ecological dynamics could support not only the conservation of Varadero but also offer critical insights for coral reef restoration across the region.

CHAPTER 12

Costa Rica

Authors: Juan José Alvarado Barrientos, Jorge Cortés, Andrea Rivera-Sosa.

Data contributors: Juan José Alvarado Barrientos, Jorge Cortés, Ana C. Fonseca†*, Cindy Fernández-García, Fabio Quesada-Perez.

*AGRRA data contributor(s).

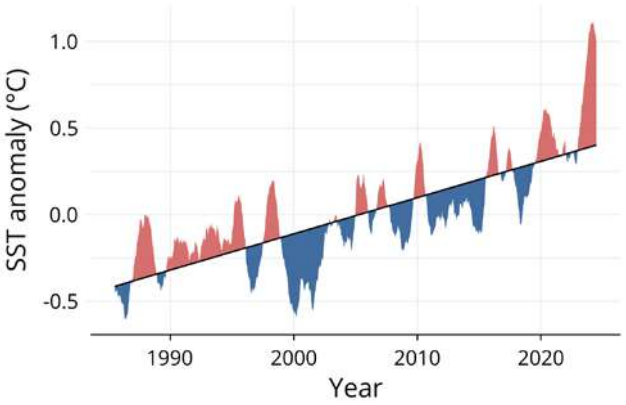
INTRODUCTION

Costa Rica, located in Central America, with coast on the Pacific Ocean and Caribbean Sea. It is renowned for its rich marine biodiversity and commitment to conservation. Costa Rica hosts 21 km² of coral reefs, which represent 0.086% of the Caribbean's total coral reef extent. On the Caribbean coast of Costa Rica, coral reefs can only be observed in its southern section, from Moín Bay to the border with Panama, where a complex reef system continues to Bocas del Toro (Cortés and Jiménez 2003). Three coral reef areas are recognized: 1) between Moín and Limón, 2) Cahuita National Park, and 3) between Puerto Viejo and Punta Mona (Cortés and Jiménez 2003). Two marine protected areas with coral reefs are found along the Caribbean coast, encompassing areas such as Cahuita National Park, and Gandoca-Manzanillo National Wildlife Refuge which features fringing reefs and carbonate banks (Alvarado et al., 2012). Uvita Island, in its terrestrial part, is a protected area under the category of National Monument due to its cultural heritage, however the marine surroundings have a high coverage of coral reefs that require a greater conservation effort (Quesada-Perez et al. 2023). Costa Rica's coral ecosystems support a diverse range of species and provide essential services, including coastal protection and habitat for marine life (Cortés, 2016).

THREATS

Thermal regime

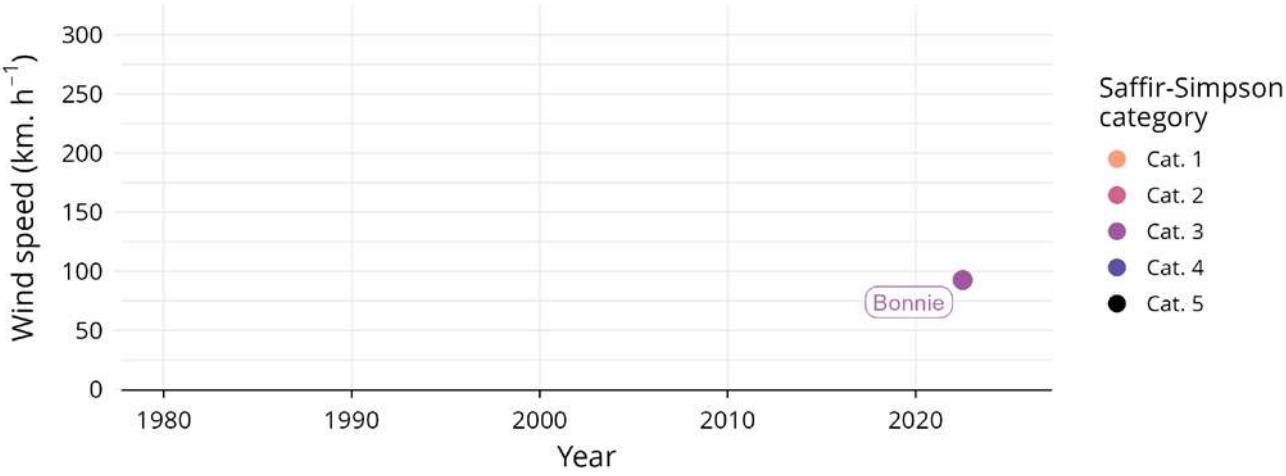
The long-term average in SST from 1985 to 2024 over coral reefs of Costa Rica was 28.11 °C. Over this period, SST increased by 0.91 °C, representing a warming rate of 0.023 °C per year (**Figure 2.12.1**). Historically, coral bleaching was observed as early as June 1983 in Parque Nacional Cahuita, Limón in the Caribbean coast of Costa Rica (Cortés et al. 1984). In 1995, coral bleaching and mortality due to heat stress were recorded in Refugio Nacional de Vida Silvestre Gandoca-Manzanillo and Parque Nacional Cahuita (Jiménez, et al 2001). The region was highly exposed to heat stress in 2011 (Muñiz-Castillo, et al. 2019) and extremely high sea surface temperature anomalies were registered in 2023-2024 (Goreau and Hayes 2024).



↑ **Figure 2.12.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Costa Rica. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Costa Rica is south of most hurricane tracts, with very few historical land falls (Alvarado and Alfaro 2003). Hurricane Allen in 1979 caused breakage and upturning of corals at Cahuita (Cortés 2016). Between 1980 and 2024, a total of 1 tropical storm passed within 100 km distance from a coral reef of Costa Rica (**Figure 2.12.2**). The hurricane with the highest sustained wind speed over the studied period was hurricane Bonnie in 2022, which passed less than 100 km from a coral reef with sustained wind speed of less than 100 km.h^{-1} it did not cause any effect on the reefs in the area. However, Hurricane Otto in November, 2016 had maximum winds of 185 km/h and was a Category 3. Although there was flooding on the Caribbean coast, it passed by the Nicaragua and Costa Rican border, further than 100 km from the coral reefs extent. Bonnie's path was very similar to Otto's, where both hurricanes exited the Pacific coast.



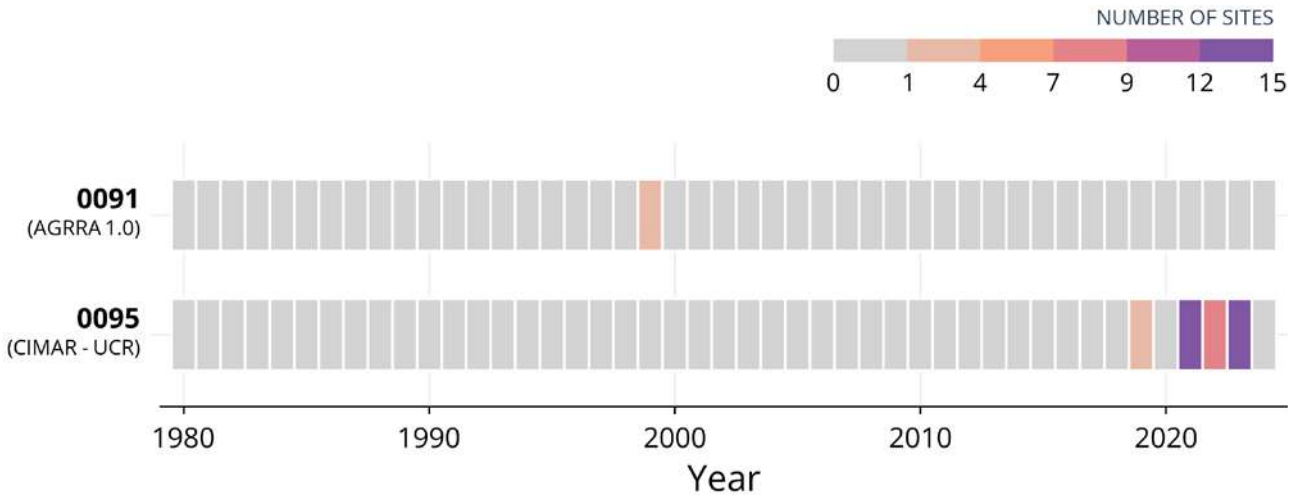
↑ **Figure 2.12.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Costa Rica. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Costa Rica are not represented, although they can have had an impact (*e.g.* hurricane Otto in 2016).

Other threats

Coral reefs along Costa Rica's Caribbean coast face mounting pressures from both land-based and ocean-based sources. Chronic siltation, nutrient enrichment from agriculture, and untreated wastewater degrade water quality and favor algal overgrowth, while unregulated tourism, coastal development, and sediment runoff intensify habitat disturbance (Cortés and Jiménez, 2003, Cortés, 2016). Reefs are further strained by warming-driven bleaching events (Jiménez, 2001), increasing macroalgal dominance (Quezada-Pérez et al., 2023). Overfishing on Costa Rica's Caribbean coral reefs threatens ecosystem function by reducing key fish populations, especially large predatory and herbivorous species, which can destabilize the reef food web and promote macroalgal overgrowth (Quezada-Pérez et al., 2023). Together, these interacting stressors erode coral resilience, reduce biodiversity, and threaten the ecological and socioeconomic value of Costa Rica's Caribbean reefs.

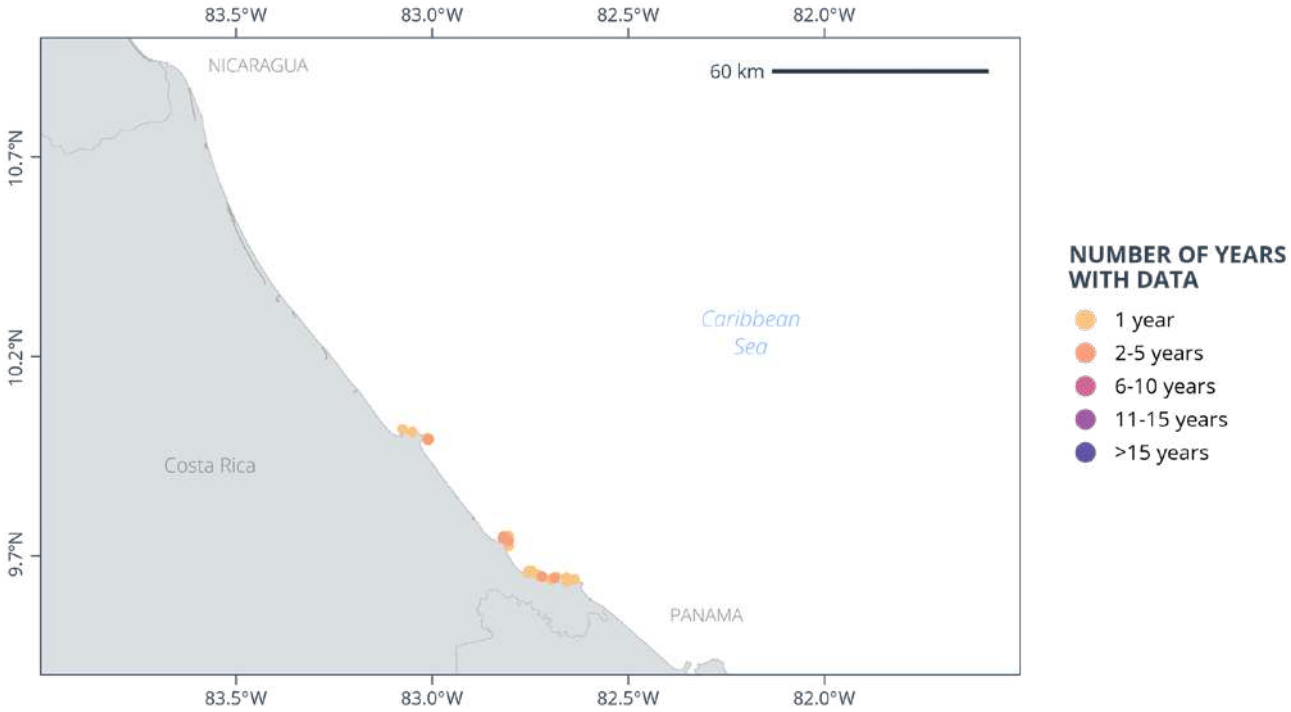
TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.12.3**) encompassing 30 monitoring sites. These data were collected through 39 surveys conducted between 1999 and 2023.

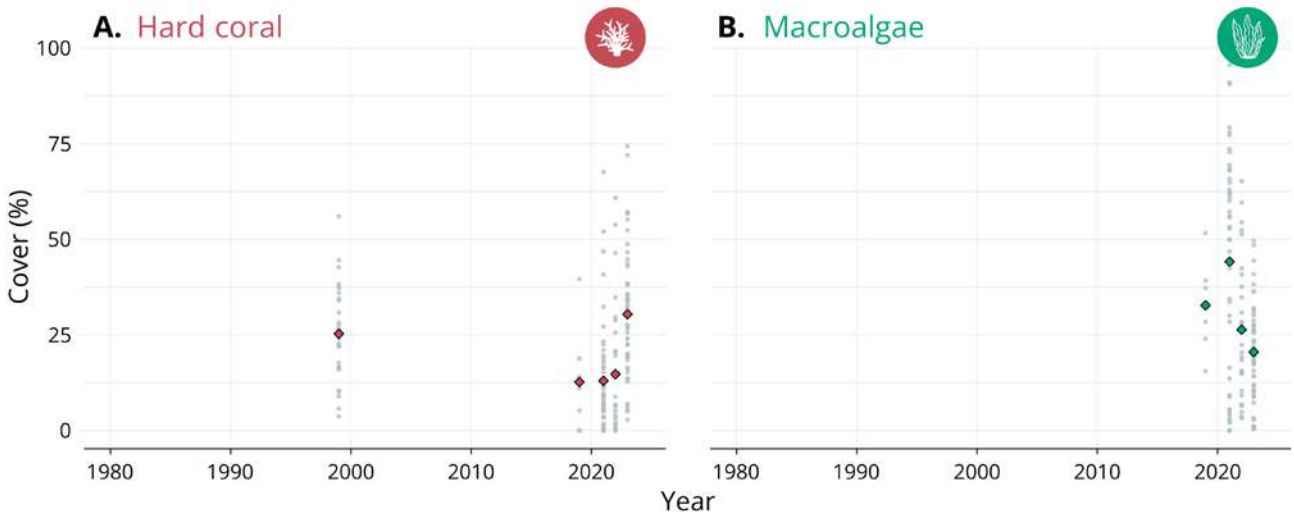


↑ **Figure 2.12.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text §below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

Although not integrated in this study, assessments of the condition of Caribbean coral reefs began in the early 1980s in Cahuita National Park (Cortés and Risk 1985). In the 1990s, more sites were assessed more systematically following the CARICOMP protocol (Cortés et al. 2010), which was further reinforced by the AGRRA monitoring in Cahuita (Fonseca, 2003). The CARICOMP protocol was the longest-standing initiative, collecting data from 1999 to 2008 (Cortés et al. 2010), but its focus was primarily on Cahuita National Park. Other initiatives made specific assessments in the 2000s and 2010s, but they were not sustained over time (Fernández-García and Alvarado 2005, Sánchez-Godínez 2014, Williams et al. 2015, Araya-Vargas and Nova-Bustos 2017). From 2019 to date, the Neotropical Reef Ecology and Conservation Laboratory (LECAN), of the Center for Research in Marine Sciences and Limnology (CIMAR) of the University of Costa Rica (UCR), established a monitoring protocol that is applied annually and covers 24 sites (Quesada-Perez et al. 2023).

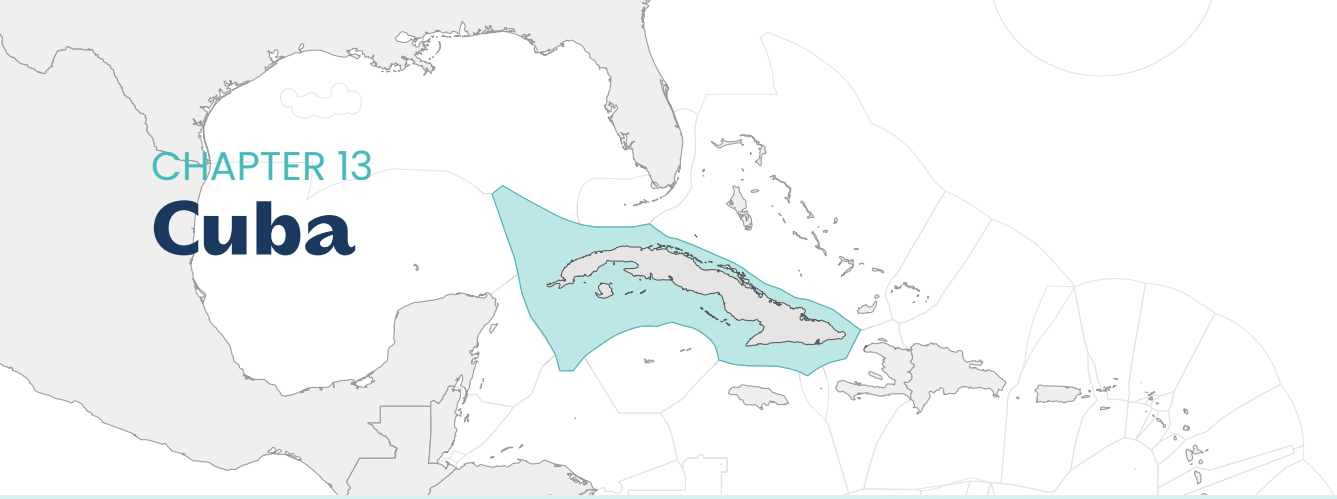


↑ **Figure 2.12.4.** Spatio-temporal distribution of benthic cover monitoring sites across Costa Rica. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.12.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Costa Rica from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Quezada et al. (2023) reported that Caribbean Costa Rican reefs were largely dominated by turf and fleshy macroalgae, particularly *Dictyota* and *Dictyopteris*, which together accounted for roughly a third of benthic cover. Hard corals occupied only about 13% of the substrate, represented by 14 reef-building species. Sites with higher coral cover (>20%) were mainly composed of *Agaricia agaricites*, *Millepora complanata*, and *Acropora palmata*, whereas reefs with moderate to low coral cover were dominated by *Siderastrea siderea,* and *Pseudodiploria* spp.



CHAPTER 13

Cuba

Authors: Patricia González-Díaz, Juliett González Méndez, Dorka Cobián Rojas, Elena de La Guardia Llansó, Susana Perera-Valderrama, Judith Lang, Hansel Caballero-Aragón, Sergio González-Ferrer, Amanda Ramos Romero, Gabriela Aguilera Pérez, Patricia Vicente Castro, Melissa Arbona Fernández, Joan Hernández Albernas, Pedro Chevalier Monteagudo, Raul Igor Corrada Wong, Alain Duran, Andrea Rivera-Sosa.

Data contributors: Gabriela Aguilera Pérez, Pedro Alcolado Mendez†*, Melissa Arbona Fernández, Hansel Caballero-Aragón*, Dorka Cobián Rojas, Tamara García*, Patricia González Díaz, Juliett González Méndez, Elena De La Guardia Llansó*, Sergio González-Ferrer*, Miguel Hernández*, Joan Hernández Albernas, Raul Igor Corrada Wong, Vladimir Kosmynin*, Philip Kramer*, Judith Lang*, Grisel Menéndez-Macía*, Amanda Ramos Romero, Alberto Rodrígues*, Patricia Vicente Castro..

*AGRRA data contributor(s).

INTRODUCTION

Cuba's insular platforms cover 85% of its coast, encompassing 20.4% of the Caribbean's total reef area (4,931 km²). Most of Cuba's coral reefs are distributed across four main archipelagos: Los Colorados, Los Canarreos, Sabana-Camagüey, and Jardines de la Reina (Estrada-Estrada et al., 2023). Local eddies and regional oceanographic currents (*i.e.*, Caribbean Current) maintain high biological connectivity within the platforms and with neighboring countries, making Cuba one of the principal larval sources of coral reef-associated organisms in the region (Schill et al., 2015).

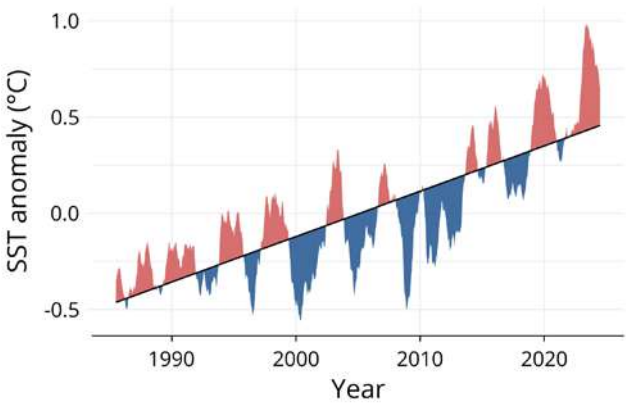
Unfortunately, Cuban reef research faces substantial challenges regarding logistics for field and laboratory work and access to academic resources. As a result, there has been limited published data that prevents robust spatial and temporal analyses. In 2023, the milestone research project, "Bojeo a Cuba", was the first time that most critical reef areas were evaluated using the same methodology. Researchers from multiple national institutes circumnavigated Cuba's shelf and produced the first comprehensive data set that includes benthic and fish communities.

→ **Figure 2.13.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Cuba. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

THREATS

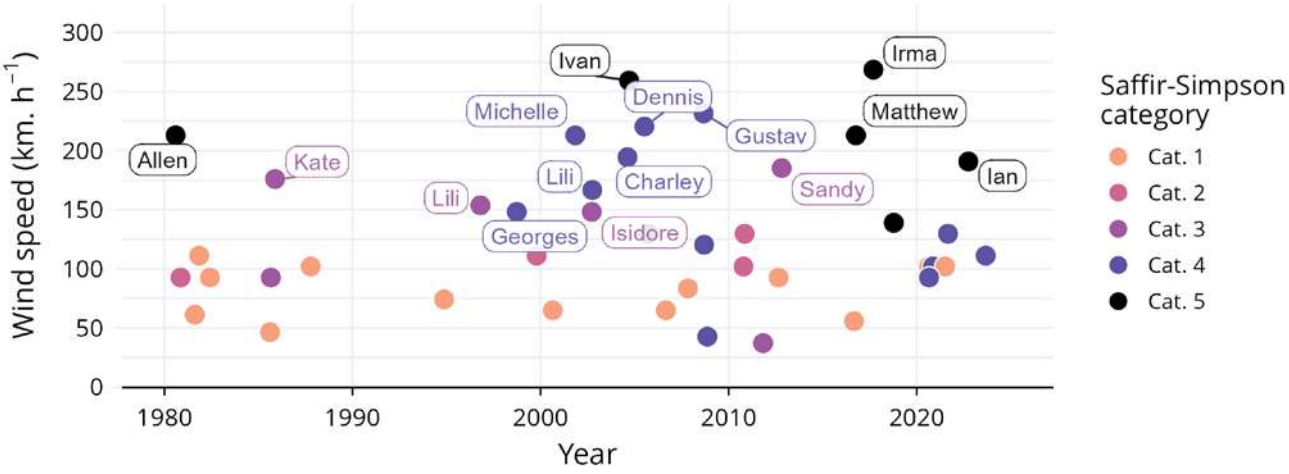
Thermal regime

The long-term average in SST over Cuba coral reefs (1985 - 2024) was 27.8°C increasing by 1.02 °C (**Figure 2.13.1**). The first bleaching event occurred in 1997-1998 in Havana (De la Guardia, 2000), followed by reports of coral loss in the Sabana-Camagüey Archipelago (Alcolado et al., 2000). There were reported bleached corals in 2005 in Jardines de la Reina and Havana without coral mortality, possibly because of the cooling effects of several category 4 and 5 hurricanes in the region (Pina-Amargós et al., 2008; Alcolado et al., 2009; González-Díaz, 2010). Record-high temperatures during July-September 2023 triggered the most severe bleaching reported to date, with rates exceeding 60% on the north and 80% on the south coast in the absence of hurricanes.



Hurricanes

Between 1980 and 2024, a total of 72 tropical storms passed within 100 km distance from a coral reef of Cuba, and of these 28 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.13.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed directly on coral reefs with sustained wind speed of 269 km.h^{-1}. In 2005 and 2008, three major hurricanes (category 4 and 5) hit Cuba between August and November. As a result, the reefs were considered capable of recovering from the bleaching events they were experiencing.



↑ **Figure 2.13.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Cuba. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Cuba are not represented, although they can have had an impact.

Overfishing

Overfishing is the main threat to Cuban reefs (Pina-Amargos et al., 2023). About 20% of fishery resources are fully exploited, 74% overexploited, and 5% collapsed (Baisre, 2018; Puga et al., 2018).

Contamination

Land-based sources such as industry, sewage, settlements near rivers, and poor solid waste management are the primary sources of pollution on reefs, particularly near large cities (Duran et al., 2018; González-Díaz et al., 2018).

Coral diseases

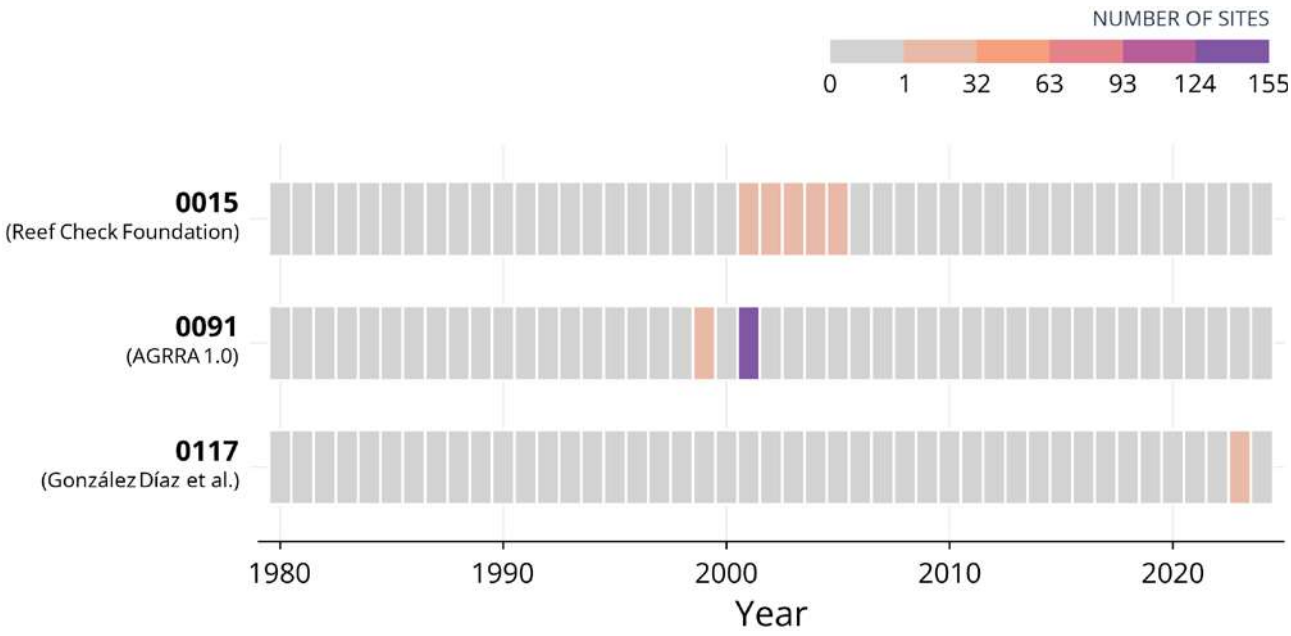
Coral disease incidence in Cuba is generally low (1-5%) (Aguilera-Pérez and González-Díaz, 2024). Stony Coral Tissue Loss Disease (SCTLD) was first detected in August 2023, on the reefs of Camaguey, and its presence was later detected on other reefs. SCTLD has affected chiefly the large, massive "ecosystem engineering" species *Montastraea cavernosa*, *Pseudodiploria strigosa*, *Diploria labyrinthiformis*, *Orbicella* spp., *Colpophyllia natans*, and the iconic *Dendrogyra cylindrus*.

Invasive species

The soft coral *Xenia umbellata* Lamarck, 1816, reported in 2023, is not currently confirmed to be a widespread threat (Espinosa-Saez et al., 2023). However, the lionfish reported in 2007 is now widespread despite control efforts, and its ecological impact remains uncertain (Cobián-Rojas et al., 2018; Chevalier-Monteagudo, 2017).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.13.3**) encompassing 196 monitoring sites. These data were collected through 205 surveys conducted between 1999 and 2023.



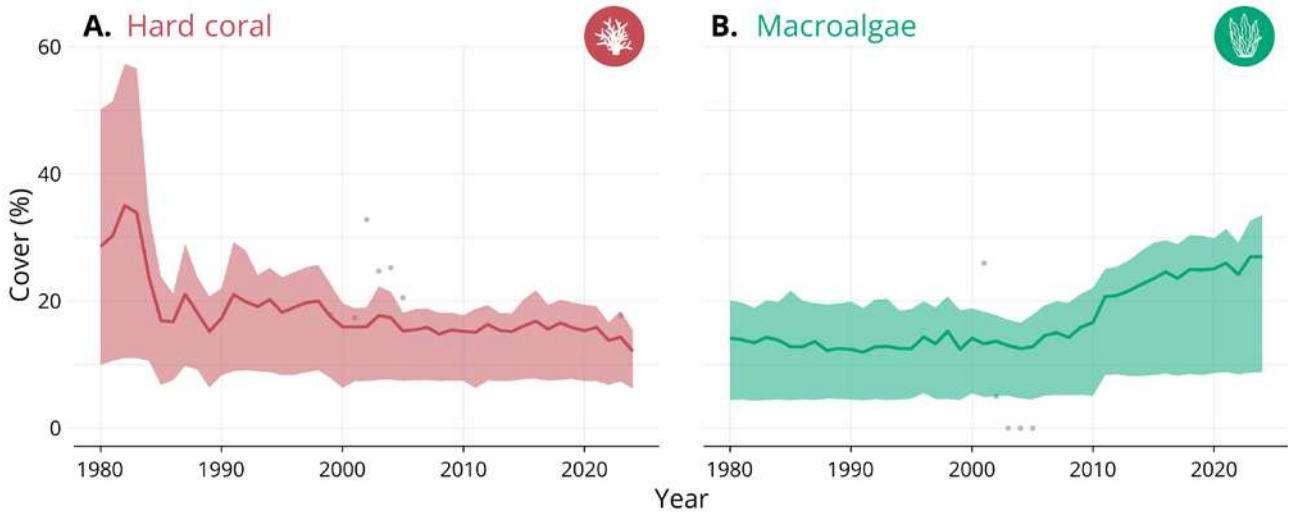
↑ **Figure 2.13.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

↑ **Figure 2.13.4.** Spatio-temporal distribution of benthic cover monitoring sites across Cuba. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

On nearshore reefs, the combined effect of top-down (overfishing) and bottom-up (land-based pollution) controls caused the predominance of algae and other benthic groups (sponges and zoanthids) to the detriment of corals. However, in offshore reefs within marine protected areas, thermal stress, and bleaching are the main impacts. The synergy of the severe bleaching event of 2023 and the appearance of SCTLD in Cuba caused high coral mortality. In the case of bleaching, it affected the genus *Acropora,* while bleaching and SCTLD affected the genera *Agaricia* and *Porites* in addition to the large ecosystem engineering species.and. Although Cuban reefs have followed the regional trend of degradation (Ramos et al., 2024), 2023 marked a critical acceleration of this process. Unlike many other Caribbean areas, there is no evidence of recent *Diadema antillarum* mass mortality on Cuban reefs.

However, population densities are naturally different on the reefs around Cuba, so possible management and rehabilitation measures include relocating individuals to the most affected reefs. Regarding herbivorous fish, large parrotfish are rare in many Cuban reefs (*i.e.*, Havana, Punta Francés, and Cayos de San Felipe), except for Jardines de la Reina National Park. Vicente-Castro et al. (2024) reported for the latter 11 species of parrotfishes with sizes exceeding 100 cm in individuals of *Scarus guacamaia* and *Scarus coelestinus*.

Cuban coral reefs show high spatial variability, with significant deterioration in many crest and fore reefs (Caballero-Aragon et al., 2019). Although there is a general trend toward degradation of Cuba's coral reefs, five Reefs of Hope have recently been identified: Laguna de Maya (Wildlife Refuge, MPA, Matanzas province); Santa Lucía (Camagüey province); Guanahacabibes National Park (MPA, Pinar del Río province); Ciénaga de Zapata National Park (Cazones area, MPA, Matanzas province) and Jardines de la Reina National Park (MPA, Ciego de Avila-Camaguey provinces). These sites are characterized by healthy populations and a high number of colonies of species of the genus *Acropora*, as well as "engineering" species (*Orbicella spp.*, *M. cavernosa*, *Siderastrea siderea*, *P. strigosa*, among others).



↑ **Figure 2.13.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Cuba from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

CHAPTER 14

Curaçao

Authors: Erik H. Meesters, Laëtitia Mathon.

Data contributors: Rolf P.M. Bak†, Andrew W. Bruckner*, Didier M. De Bakker, Caren Eckrich, Andreas F. Haas, Thomas Kemenes Van Uden, Erik H. Meesters, Gerard Nieuwland, Maggy Nugues, Sandin Stuart, Mark Vermeij, Matthijs Van Der Geest.
*AGRRA data contributor(s).

INTRODUCTION

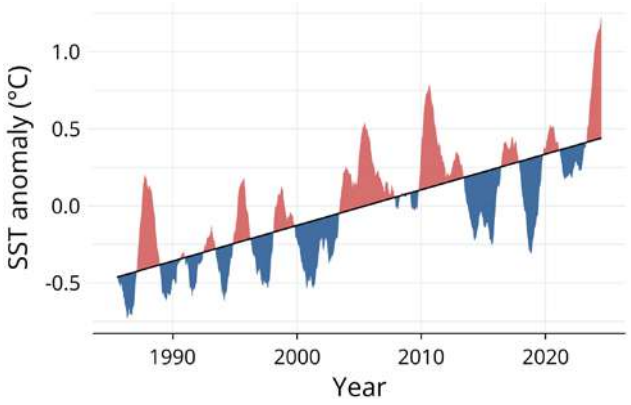
Curaçao lies between latitudes 12°2'80" to 12°23'30"N and longitudes 69°10'00" to 68°44'30"W. The island is 61 km long and 12 km wide at its widest point. The island has a surface of 444 km² and hosts a population of 156,000 inhabitants in 2024. The reefs of Curaçao occur foremost along its leeward coast and the windward side of a small uninhabited island Klein Curacao. Reefs along the islands' windward shores are less developed except for some local, very well developed coral communities. Around Curaçao large inland bays are found harbouring mangrove and seagrass communities serving as nursery habitats for many reef fish species. From the shore, a 50-200m wide shallow reef flat with only scattered patchy coral communities gradually slopes to a drop-off starting between 7-15m depth (Bak, 1975). From here the fore reef slopes steeply downward at 45-90° to a second drop-off around 80-90m. In most locations, the first drop-off often turns into a sandy terrace around 50-60m that gently slopes to deeper water until the second drop-off begins. In general, coral cover and diversity increase from the high-water mark towards the first drop-off, whereby the habitat changes from a sandy bottom to a consolidated reef of Pleistocene limestone rock covered by living corals. Coral cover and diversity remain high crossing the drop-off down to mesophotic depths (below 30m), after which both decrease to around 80m where coral growth becomes extremely sparse (Frade et al. 2019).

The Curaçao Underwater (marine) Park is located off Curaçao's southeast coast. It stretches from Lijhoek at Janthiel along 21.7 kilometres (13.5 miles) of shoreline from the high-water mark to a distance of 100 m from shore, to and around the most eastern tip of the island at Oostpunt. The park includes 217 hectares of pristine fringing reefs. In addition the fringing reefs around Klein Curacao, near the island's northwestern tip, near Kaap Marie and St. Michiel are protected as part of Curacao's five RAMSAR areas.

Curaçao's marine resources contribute substantially to the island's economy, with marine tourism contributing $373.5 million annually (Wait Institute, 2016).

THREATS

Thermal regime



↑ **Figure 2.14.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Curaçao. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

The long-term average in SST from 1985 to 2024 over coral reefs of Curaçao was 27.42 °C. Over this period, SST increased by 1.04 °C, representing a warming rate of 0.027 °C per year (**Figure 2.14.1**). Major declines in the abundance of reef building corals coincided with coral bleaching events in 1998, 2010 and 2023/24).

Hurricanes

Between 1980 and 2024, a total of 9 tropical storms passed within 100 km distance from coral reefs of Curaçao. Major declines in the abundance of reef building corals coincided with swells of tropical storm Brett in 1993 (Van Veghel and Hoetjes, 1995), and Hurricane Lenny in 1999 (Bries et al., 2004) and tropical storm Omar in 2008. Following Hurricane Lenny, toppling, fragmentation, tissue damage, bleaching, and smothering were observed, with the greater damage along westward-facing shores.

Other threats

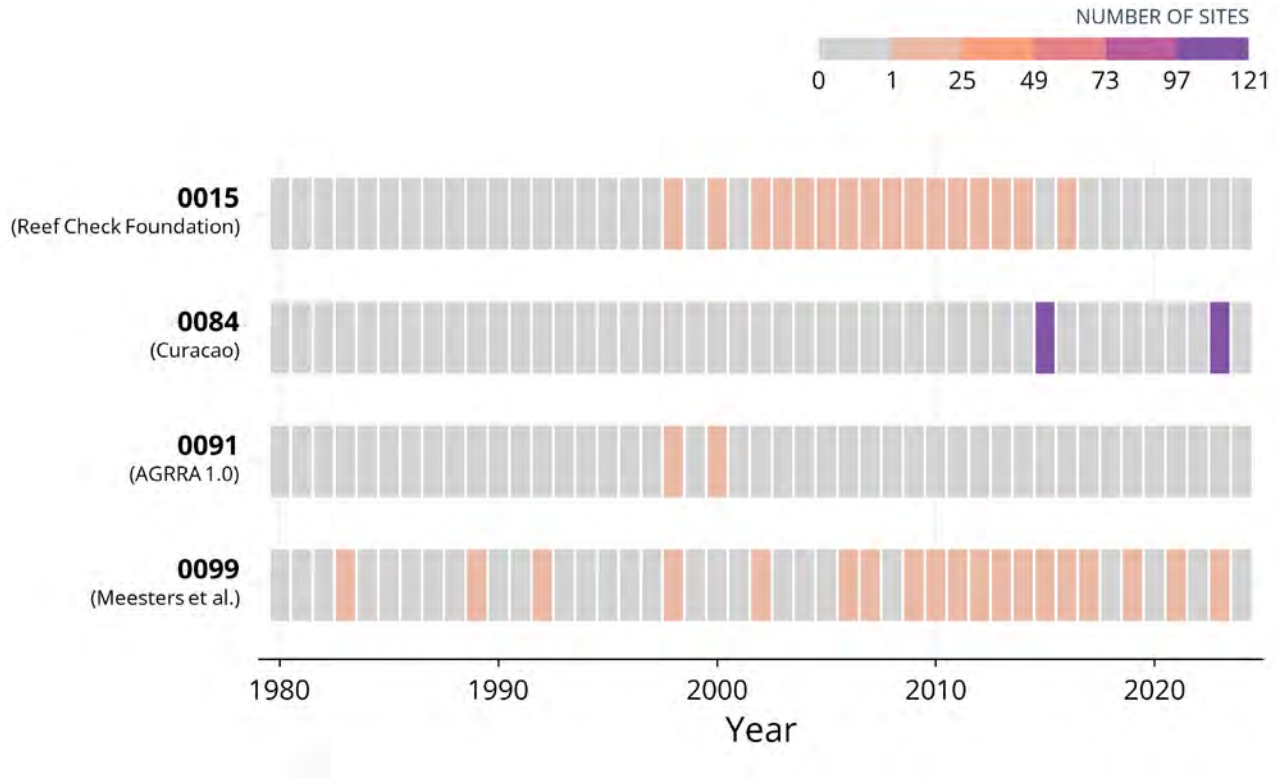
In addition to storm and bleaching effects, major declines in the abundance of reef building corals steps coincided with disease outbreaks (*e.g.*, white-band/white-plague outbreaks in the late-1970s to mid-1980s and SCTLD in 2023-present), Fish communities changed in parallel with the decline in coral abundance. Historical catch analyses that span from 1905 to 2020 reveal long term "sequential over fishing", whereby landings of large reef predators (groupers, jacks, reef sharks) collapsed by the 1950s (Vermeij et al., 2019). Fishing efforts then shifted offshore and toward smaller herbivores and pelagics. Overfishing and habitat destruction are estimated to have equally contributed to the decline in reef fishes around Curacao (Vermeij et al., 2019).

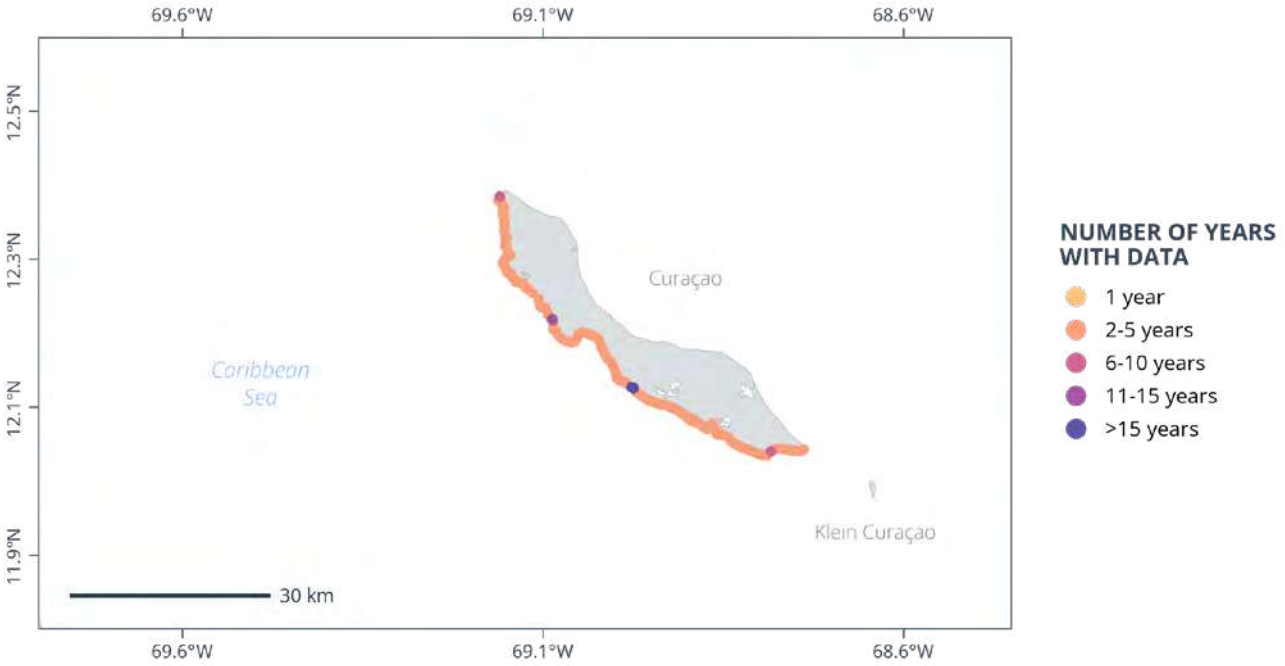
There are many forms of pollution that affect the survival of reef species and the functioning of coral reef communities. While many pollutants (*e.g.*, sewage, nutrients, chemicals and heavy metals) originate in densely populated or industrialized areas (such as the general Willemstad area) west flowing currents transport such pollutants within a day towards the island's most western tip (Bertoncelj et al., 2025). Run-off, especially during rain, groundwater flow towards the ocean and direct release (*e.g.*, sewage overflows) are the prime mechanisms by which pollutants move from land to the ocean. There are 60 sewage overflow locations in Willemstad dumping raw sewage directly into coastal waters used for recreation etc. Only 30 % of households connected to the island's sewer system and municipal sewage treatment plants treat <50 % of incoming flow (Erdogan, 2021). The release of untreated sewage and terrestrial run-off during rain events also promotes the growth and abundance of (pathogenic) microbes in the island's coastal waters. The general increase in microbe abundances in Curacao coincides with increased disease occurrences in fishes (*e.g.*, de Wit and Johnson, 2024), corals (*e.g.*, Cróquer et al., 2021) and people (*e.g.*, Steingrover et al., 2016). Land derived organic pollutants such as biocides, drinking water chemicals, drugs of abuse, food contact chemicals, human metabolites, human neurotoxins, indoor environment substances, natural toxins, persistent, mobile and toxic substances, personal care products, pharmaceuticals, plant protection products, plastic additives, smoke compounds, surfactants are not locally confined to specific discharge locations, but propagated around the entire island making xenobiotic pollutants a concern across broader spatial scale than their specific discharge point or location of origin (NIOZ, unpubl. data).

TRENDS IN BENTHIC COVER

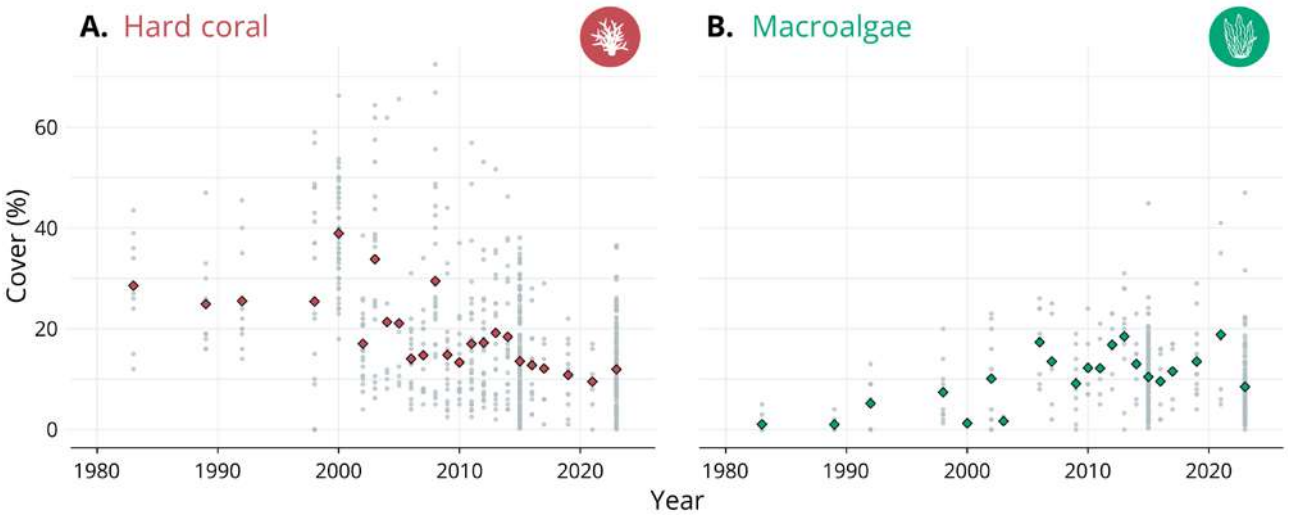
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated with GCRMN's model using data from 4 datasets (**Figure 2.14.2**) encompassing 148 monitoring sites (**Figure 2.14.3**). These data were collected through 438 surveys conducted between 1973 and 2023.

↓ **Figure 2.14.2.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.14.3.** Spatio-temporal distribution of benthic cover monitoring sites across Curaçao. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.14.4.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Curaçao from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Systematic reef monitoring around Curaçao began in the early 1970s, when live hard-coral cover averaged 45 – 55 % on leeward fore-reefs at 10 – 20 m depth (Bak, 1977; Bak and Nieuwland, 1995). An island-wide decline in average coral cover occurred to ≈18 % by 2010 followed by a further decline to ~13% in 2015 (Sandin et al., 2022) after which island wide average coral cover remained stable to 2023 (CARMABI, unpubl. data). Locally (*e.g.*, Oostpunt, Klein Curacao) areas with high coral cover (> 25%) remain. Through time, the decline in coral cover was mirrored by an increase in the abundance and average height of fleshy macro and turf algae.

CHAPTER 15

Dominica

Authors: Laëtitia Mathon.
Data contributors: Kristina Byrd*, Michael Jordan*, Jena McNeal*, Sascha Steiner*, Nicholas Wallover*, Rachel Zuercher*.
*AGRRA data contributor(s).

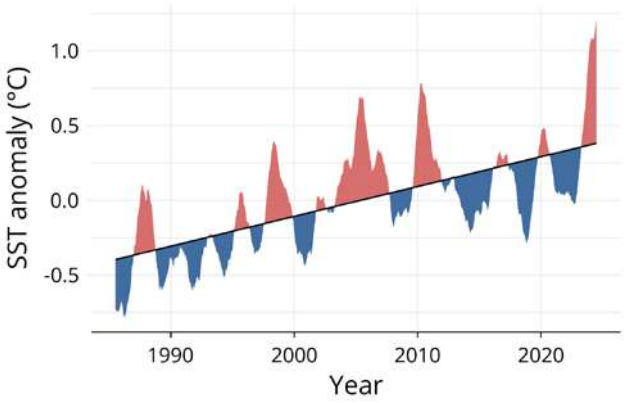
INTRODUCTION

Dominica, a volcanic island of about 750 km² in the Eastern Caribbean, supports roughly 49 km² of coral reefs—just 0.2% of the region's total reef area. Though its human population is relatively small (about 73,000 in 1980 and 66,200 in 2024), most residents live along the coast, historically placing notable fishing pressure on marine ecosystems. The island's benthic habitats extend up to 1.2 km from shore in waters shallower than 50 m, while much of the coastline features steep, near-vertical drop-offs. To safeguard marine biodiversity, Dominica established two marine protected areas: Cabrits National Park in 1987 and Soufrière-Scott's Head Marine Reserve in 1998 (Steiner, 2015).

THREATS

Thermal regime

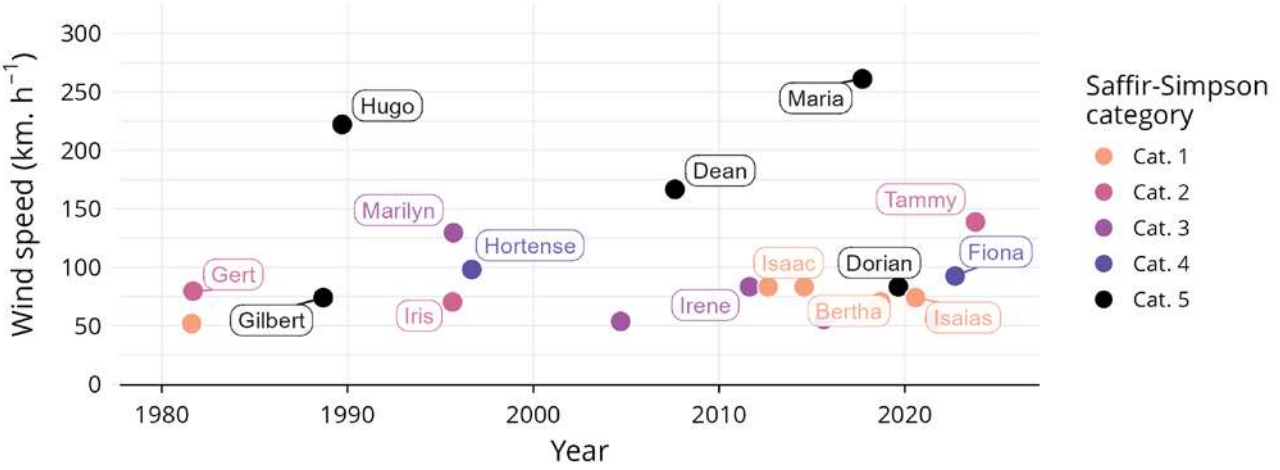
The long-term average in SST from 1985 to 2024 over coral reefs of Dominica was 27.70 °C. Over this period, SST increased by 0.94 °C, representing a warming rate of 0.024 °C per year (**Figure 2.15.1**).



↑ **Figure 2.15.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Dominica. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 29 tropical storms passed within 100 km distance from a coral reef of Dominica, and of these 6 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.15.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Maria, in 2017, which passed 3 km from a coral reef with sustained wind speed of 261 km.h^{-1}.



↑ **Figure 2.15.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Dominica. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Dominica are not represented, although they can have had an impact.

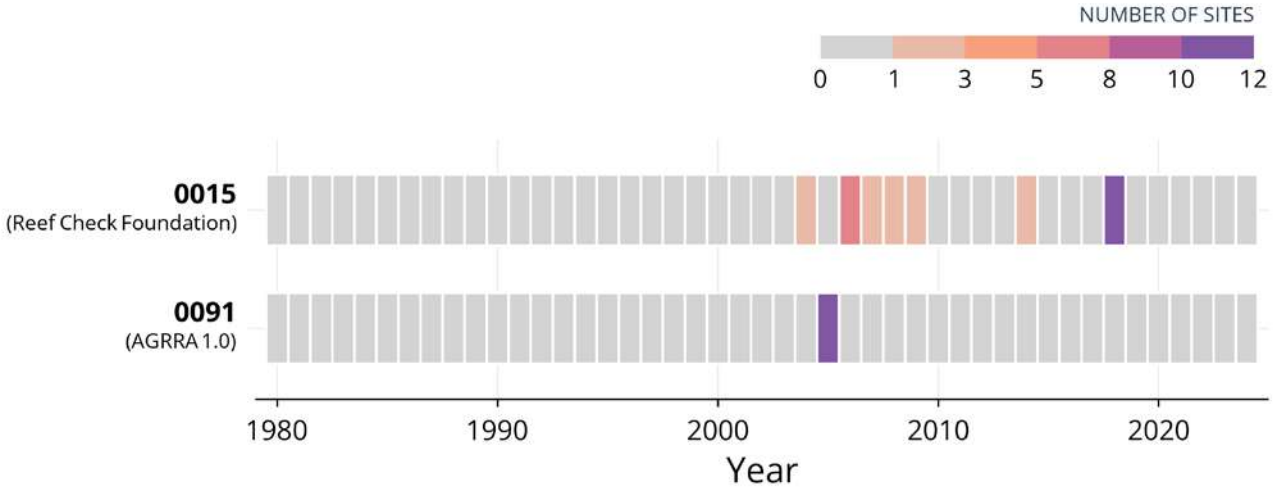
Other threats

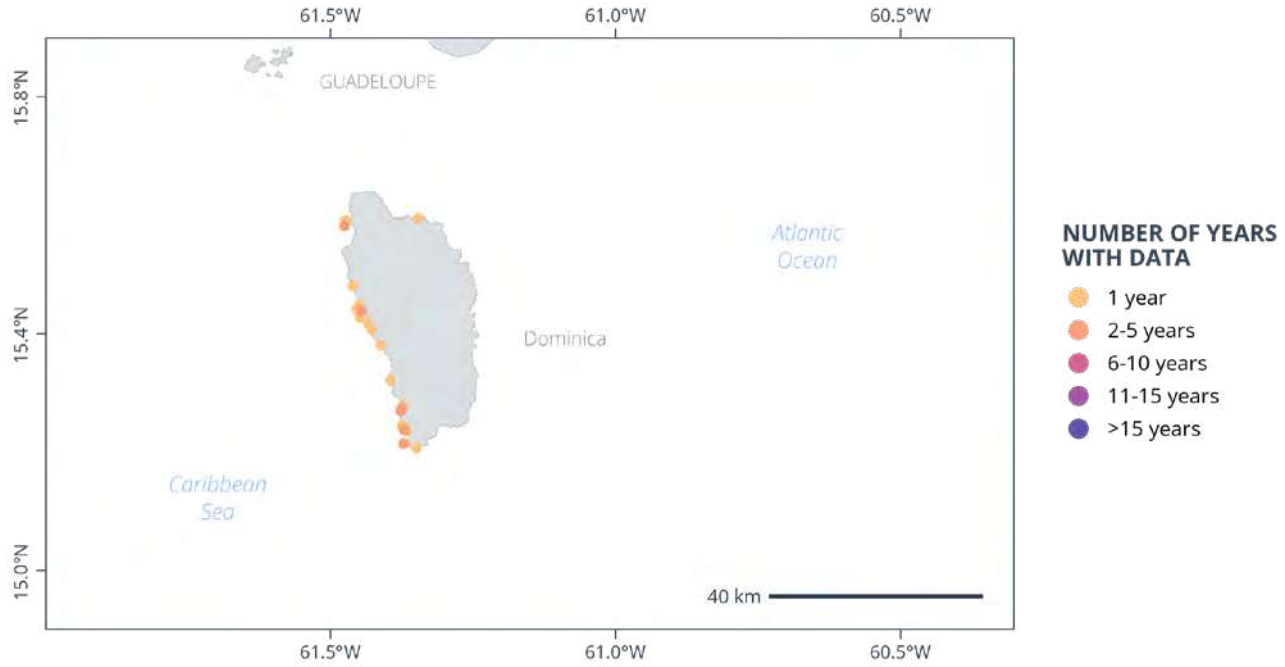
Coral reefs in Dominica face multiple pressures, particularly from land-based and fishing impacts. Heavy rainfall drives sediment runoff into coastal waters, reducing light penetration, altering salinity, and introducing debris and organic matter that limit photosynthesis in corals' symbiotic algae. In addition, insufficient enforcement of coastal management contributes to ongoing habitat degradation. While local fisheries remain small-scale and artisanal, their broad targeting of marine species has placed significant strain on reef ecosystems around the island (Steiner, 2015).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.15.3**) encompassing 31 monitoring sites (**Figure 2.15.4**). These data were collected through 46 surveys conducted between 2004 and 2018.

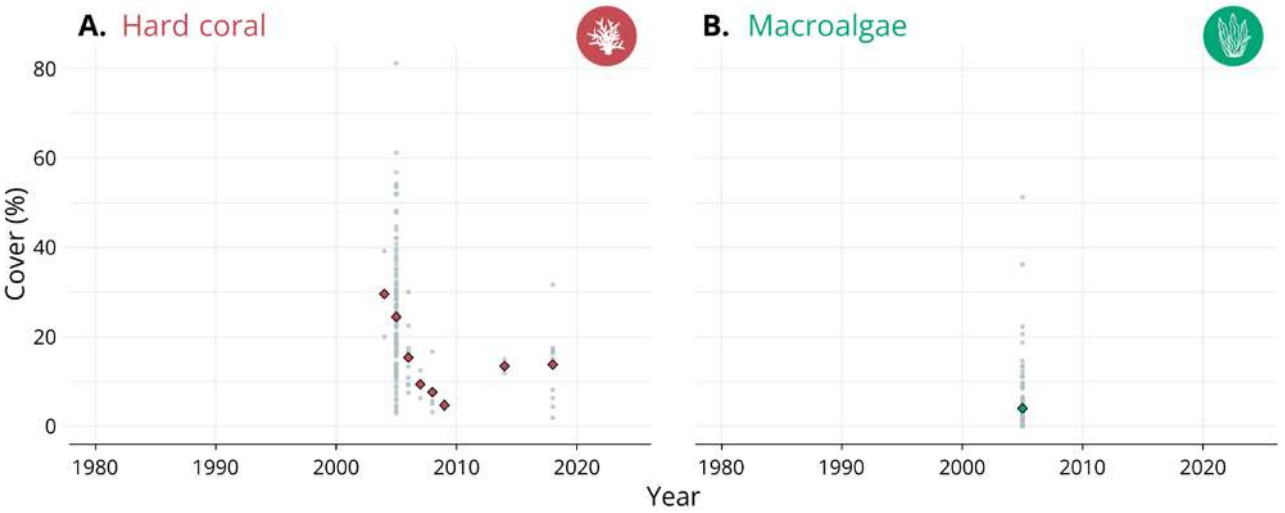
↓ **Figure 2.15.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.15.4.** Spatio-temporal distribution of benthic cover monitoring sites across Dominica. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Dominica's coral reefs have shown clear signs of decline over the past two decades, driven by disease outbreaks and climate-related stressors. Between 2000 and 2002, scleractinian corals were affected by multiple syndromes, including white plague (WP), black band disease (BBD), and dark spots syndrome (DSS), with an overall increase in disease incidence—DSS being the only one with a statistically significant rise (Borger and Steiner, 2005). On the north coast, shallow fringing reefs dominated by *Acropora palmata* now exhibit less than 5% live coral cover, with widespread evidence of past mortality (Steiner, 2015). Both *Orbicella* species have experienced catastrophic tissue loss in the last 15 years, a trend accelerated by the 2005 mass bleaching event (Steiner, 2015).



↑ **Figure 2.15.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Dominica from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 16

Dominican Republic

Authors: Aldo Croquer, Ruben Torres, Robert Steneck, Macarena Blanco-Pimentel, Melina R. González, Maria F. Villalpando, Johanna Calle-Triviño, Ainhoa Leon Zubillaga, Rita I. Sellares-Blasco, Rebecca Garcia Camps, Laëtitia Mathon.

Data contributors: Rebecca Albright*, Shamwari Anseeuw, Maria G. Aponte, Ernesto Arias, Daniel Aurelio*, Macarena Blanco-Pimentel, Marilyn Brandt*, Johanna Calle-Triviño, Rebecca Garcia Camps, Aldo Croquer, Someira Zambrano, Lizeth Cruz, Cara Dickman*, Victor Galván, Rodrigo Garza*, Melina R. González, Sergio D. Guendulain-García, Geury Guerrero, Casey Hermoyian*, Ana Carolina Hernández-Oquet, Iker Irazabal, Samuel King*, Judith Lang*, Carrie Manfrino*, Emy Miyazawa, Bernhard Riegl*, Andreina Rivera, Rita I. Sellares-Blasco, Robert Steneck, Ruben Torres*, Andreina Valdez-Trinidad, Maria F. Villalpando*, Stacey Williams*, Ainhoa Leon Zubillaga, Estefany Vargas-Pérez, Michael A. Del Rosario.

*AGRRA data contributor(s).

INTRODUCTION

The Dominican Republic, spanning 48,671 km² of land and an Exclusive Economic Zone (EEZ) of 255,898 km², has emerged as a regional leader in ocean protection, having already surpassed the 30×30 global target by protecting 30.8% of its marine waters ahead of the 2030 deadline. Within this vast marine jurisdiction, the nation hosts about 750 km² of coral reefs, representing 3.097% of the Caribbean's total reef area.

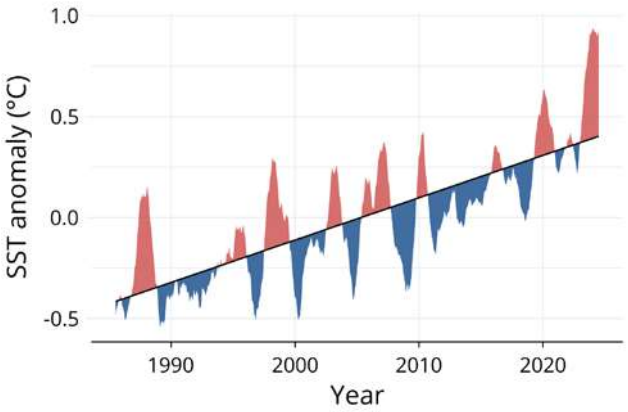
With its population nearly doubling from 5.73 million in 1980 to 11.43 million in 2024, pressures on coastal and marine ecosystems have grown substantially. Coral reefs sustain essential activities such as fisheries and tourism, while delivering critical ecosystem services including shoreline protection, nursery habitats for commercial species, cultural identity, and contributions to climate resilience.

A recent valuation estimates that the country's blue economy ecosystems—coral reefs, mangroves, seagrass meadows, and marshes—generate about US$1,792 million annually, or 1.58% of national GDP. This underscores coral reefs not only as ecological treasures but also as foundations of sustainable development and national prosperity.

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of the Dominican Republic was 27.53 °C. Over this period, SST increased by 0.95 °C, representing a warming rate of 0.02 °C per year (**Figure 2.16.1**), lower than the 1.07°C SST increase averaged over the whole Caribbean region.



↑ **Figure 2.16.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of the Dominican Republic. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Dominican Republic reefs have been repeatedly affected by major bleaching events, notably in 1998, 2005 (Wilkinson and Souter, 2008), and most recently during the 2023-2024 marine heatwave. This heatwave reached unprecedented thermal stress levels (Hoegh-Guldberg et al., 2023), triggering widespread bleaching and mortality across Dominican reefs (*data contributors per. obs*). These recurrent events illustrate the shortening of recovery windows and the compounding effects of repeated disturbances.

As climate change continues to worsen, there are four major lines of action that must be undertaken mitigate the impacts of future extreme climate events: (1) the final and unavoidable reduction of green-house emissions to limit further ocean warming and ocean acidification, (2) efforts to enhance the resilience of marine ecosystems (*e.g.* protecting critical habitats and refugia), (3) fostering adaptive management practices such as monitoring programs that track ecological conditions and ecological indicators to improve decision making and conservation/restoration planning and (4) foster exchange and learning of technological tools for coral adaptation. In the Dominican Republic, future heatwaves may have deep and likely irreversible consequences for coral reef ecosystems that have already undergone rapid and significant changes in less than 5 years.

Hurricanes

Between 1980 and 2024, a total of 40 tropical storms passed within 100 km distance from a coral reef of Dominican Republic, and of these 14 were characterized by sustained wind speed greater than 100 $km.h^{-1}$. The hurricane with the highest sustained wind speed over the studied period was Hurricane Irma, in 2017, which passed 93 km from a coral reef with sustained wind speed of 269 $km.h^{-1}$.

Fishing

Fishing is a vital livelihood in the Dominican Republic, with 14,292 fishers identified in the 2019 National Fisheries Census (MEPyD, ONE, & CODOPESCA, 2020). Coral reefs, though less frequented than sandy, rocky, and muddy bottoms, support small-scale coastal fisheries that capture over 100 species, mainly for local markets and tourism (Betancourt and Herrera-Moreno, 2019). By 2025, 30.8% of Dominican waters were protected, with marine protected areas and fishing closures (CODOPESCA, 2022) aimed at regulating exploitation and restoring key species. However, limited enforcement and ongoing illegal practices undermine these efforts, and greater adoption of sustainable fishing practices remains necessary.

Coral diseases

Coral reef decline in the Dominican Republic is mainly driven by disease and bleaching. Stony Coral Tissue Loss Disease (SCTLD), first reported in 2019 (Irrazabal and Rodriguez, 2019), spread rapidly nationwide, causing unprecedented mortality of adults and recruits of reef-building corals such as *Dendrogyra cylindrus*, *Meandrina meandrites*, and *Colpophyllia natans* in less than 4 years (Croquer et al., 2022, 2024). In 2022, SCTLD reached Bayahibe, where massive coral mortalities overlapped with an epizootic affecting *Diadema antillarum* (Villalpando et al., 2022). However, by June of 2022 high populations of Diadema persisted and/or were not affected by the disease. For example, in Punta Aguila, high densities of this species were recorded (2.8 ind/m^2, Steneck and Torres, 2023). The SCTL disease spread out along the coast and started to be noticed in the southern and southwestern coast of the Dominican Republic between 2023 and 2024, and is now present across the country. By 2023–2024, the fourth massive bleaching event further reduced live coral cover by 60–96%, leaving about 70% of Dominican reefs with <5% cover (Monitorea, unpublished).

Coastal development

During the past decade coastal development in the Dominican Republic has been rapidly expanding. Every year new infrastructure is built along the coastline, particularly, new hotels, residential areas and ports. While the impacts of coastal development have not been quantified at national scale, impacts of sedimentation on reefs is becoming evident, as coral colonies are often smoothed by silty sediments. Beach erosion triggered by coral reef decline and the alteration of dune and sediment deposition coastal dynamics is becoming a common problem in the country (Jaramillo et al., 2025).

Eutrophication

Similar to other countries in the region, eutrophication is a serious problem in the Dominican Republic. During the past decades, there have been efforts to monitor the water quality in the country (ANAMAR, 2025; Castillo-Jáquez et al., 2022), however, these studies either remain in data bases, have been conducted in specific locations for a few years, or have established correlative evidence and not necessarily cause-effect links between eutrophication and coral reef decline. Despite the paucity of long-term studies to show the cause-effect of eutrophication on water quality and coral health in the country, the rapid coastal development and the use of traditional agricultural practices might be reducing water quality and environmental variables that are essential for reefs to keep up.

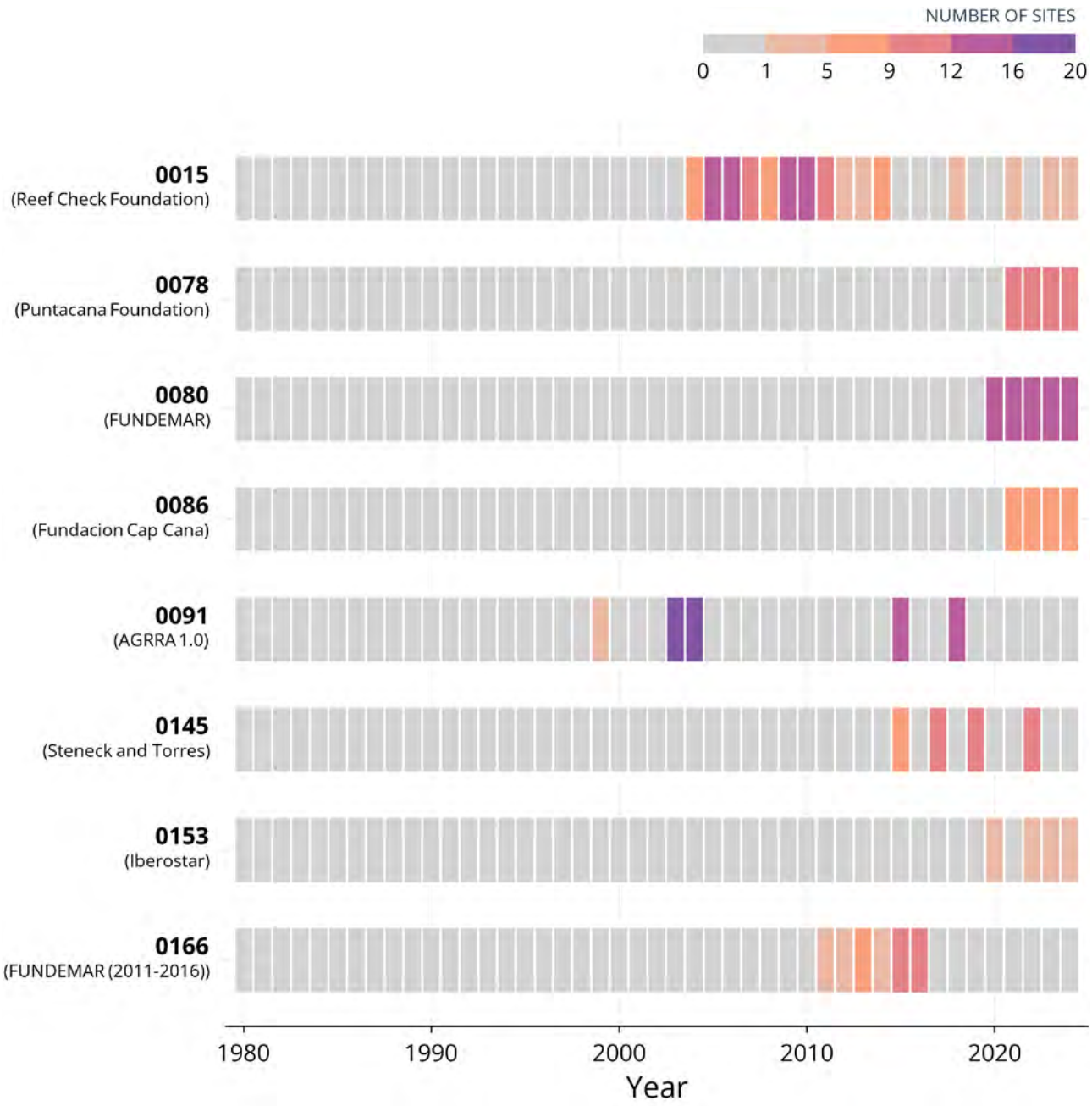
Invasive species

Lionfish (*Pterois volitans*) are widespread across Dominican coasts and reef habitats, though no local studies have formally assessed their impacts. Another emerging threat is the invasion of Red Sea *xenids*, which may expand following recent coral mortalities. In the Caribbean, three *xenid* species have been reported: *Unomia stolonifera*, *Xenia umbellata*, and *Latissimia ningalooensis* (Aguilera-Perez and Gonzales-Diaz, 2024; Toledo-Rodriguez et al., 2025a; 2025b). These invasions have caused severe declines in stony coral cover, inhibited coral recolonization, displaced native species, and reduced biodiversity and ecosystem services, as documented in Venezuela and Puerto Rico (Ruiz-Allais et al., 2021). In anticipation, The Nature Conservancy and the Ministry of Environment have developed a national response plan to address potential xenid invasions in the Dominican Republic.

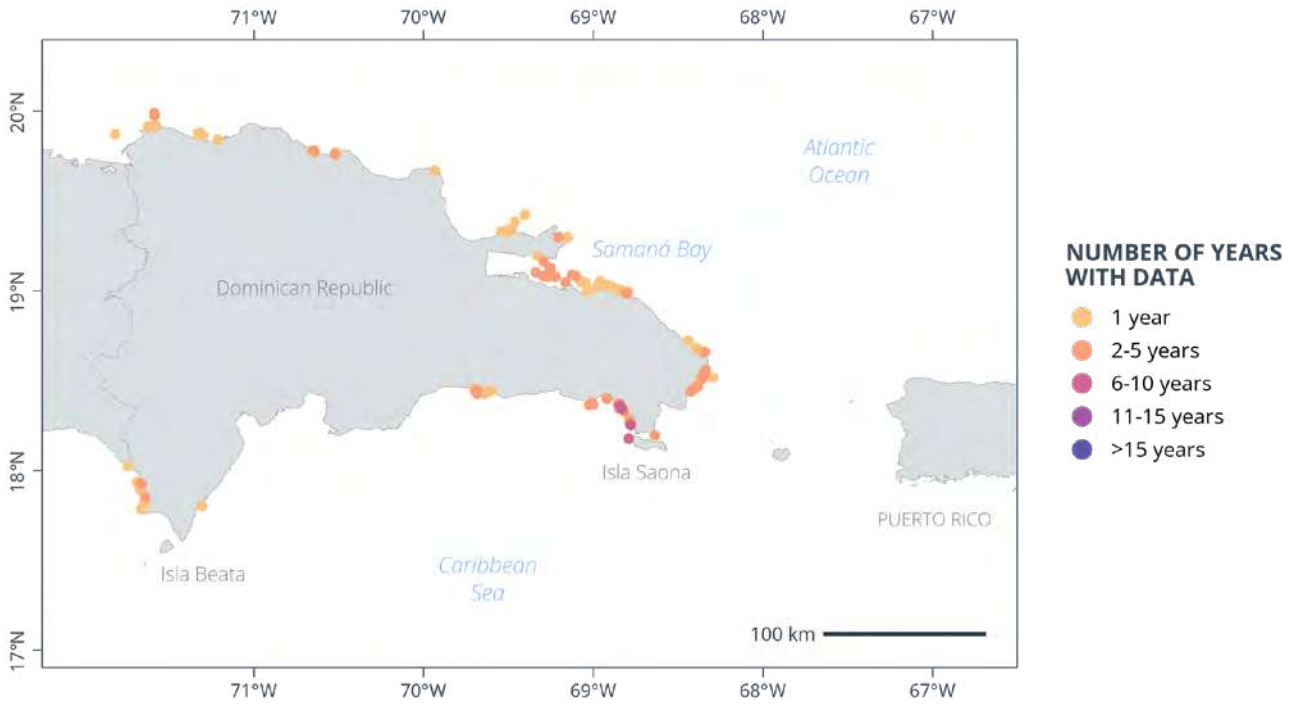
TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 8 datasets (**Figure 2.16.2**) encompassing 178 monitoring sites (**Figure 2.16.3**). These data were collected through 516 surveys conducted between 1999 and 2024.

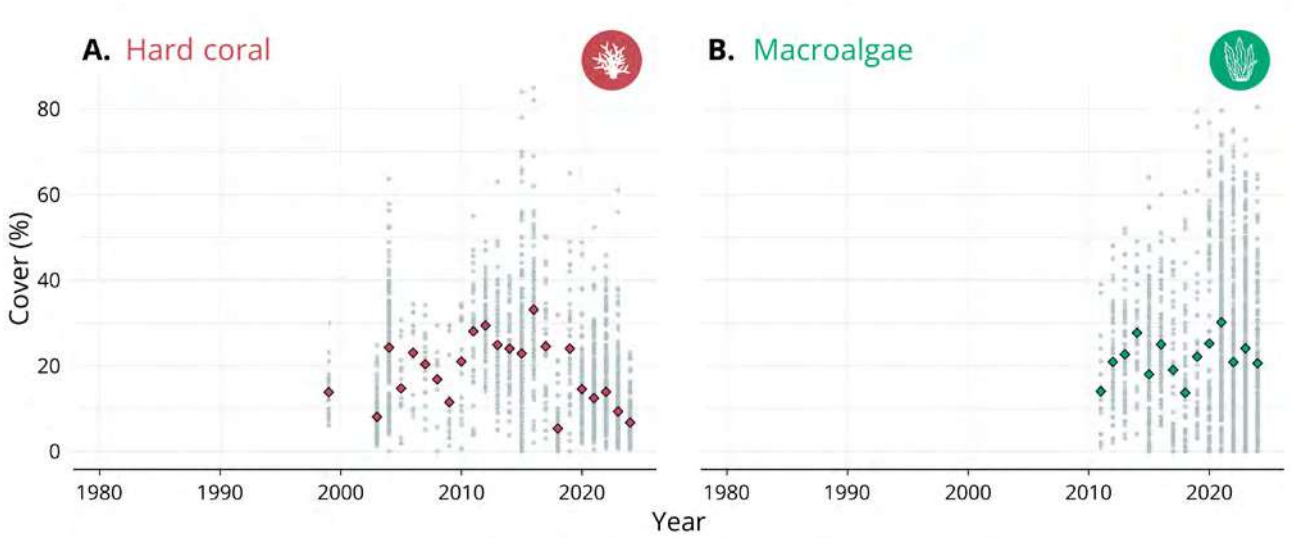
The results show a steadily declining trend in live coral cover, an increasing trend of algal cover, whereas other sessile benthic organisms have remained stable over the past 45 years. Before 2022, across the country, some reefs in the Dominican Republic had live coral cover above modeled cover in the region (*e.g.* Montecristi, La Caleta, Catalina and Coco Reef in Bayahibe) (Cortes-Useche et al., 2019). However, after the 2022 SCTLD epizootics and the 2023-2024 massive bleaching event, live coral cover in these sites was reduced to similar levels predicted and recorded throughout the Caribbean region. The status of coral reefs in the Dominican Republic today is critical, with almost 70% of reefs recently surveyed across the country showing live coral cover below 5% (Monitorea, unpublished). Compared with an earlier national review conducted in 2003 and with former national surveys conducted by ReefCheck RD between 2015-2019 (Steneck and Torres, 2024), coral reefs in the country are showing accelerated declining trends.



↑ **Figure 2.16.2.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.16.3.** Spatio-temporal distribution of benthic cover monitoring sites across Dominican Republic. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.16.4.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Dominican Republic from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Case study nº3

The value of a joint coral-monitoring program to improve the outcome of coral restoration programs at the Southeast Reefs Marine Sanctuary of the Dominican Republic

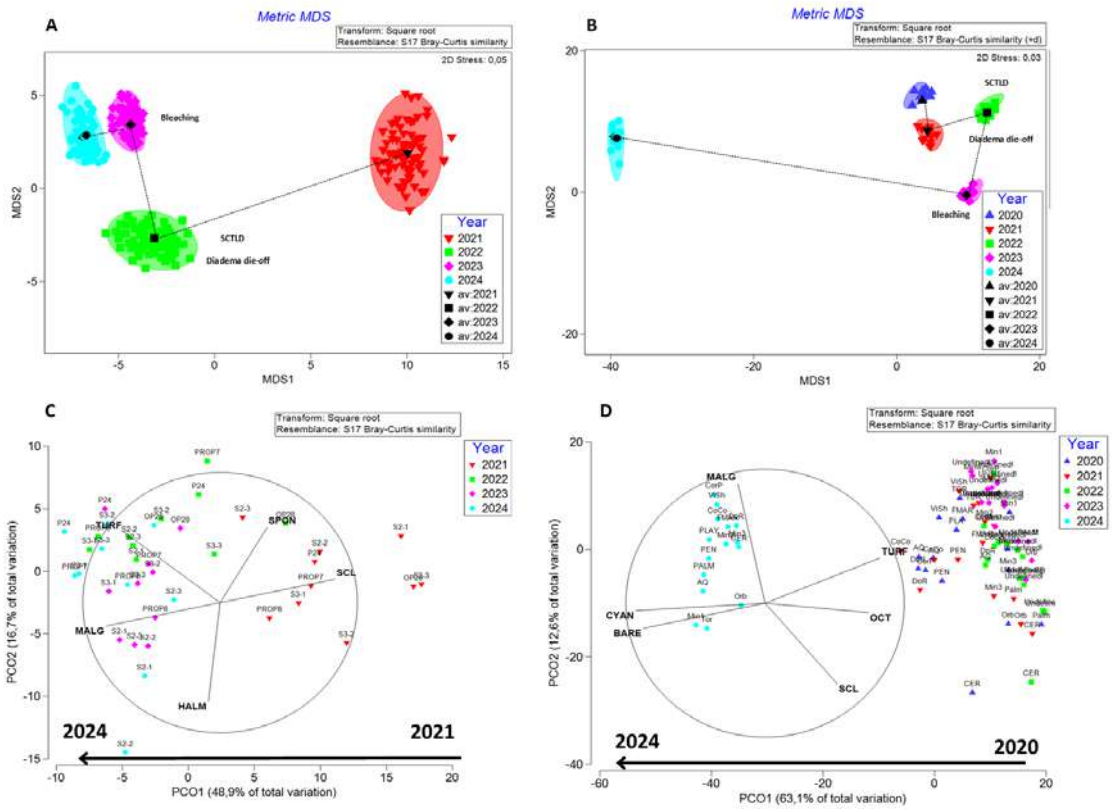
Authors: Aldo Croquer[1], Ainhoa L. Zubillaga[2], Rita I. Sellares Blasco[3], Rebecca Garcia-Camps[2], Andreina Rivera[2], Emy Miyazawa[2], Andreina Valdez-Trinidad[3], Maria F. Villalpando[3], Sergio D. Guendulain-García[3], Ana Carolina Hernández-Oquet[4], Melina R. González[4], Iker Irazabal[5]

[1] The Nature Conservancy
[2] Fundación Puntacana
[3] Fundación Dominicana de Estudios Marinos
[4] Fundación Cap Cana
[5] Red Arrecifal Dominicana

As coral reefs continue to undergo rapid change and their variable trajectories increase uncertainty about the future of these ecosystems, effective actions to preserve what is left and to restore what has been lost must be a global priority (Segaranet al., 2024). While this task is daunting, particularly in the context of climate change scenarios, coral reef monitoring programs can provide insights to advance assertive management actions and to support robust coral restoration strategies based on quantitative data.

During the past four decades, governmental, academic and NGOs institutions have been monitoring coral reefs in the Dominican Republic (Steneck et al., 2023; Creary et al. 2008). These monitoring efforts have been extremely useful for understanding the status and trends of these ecosystems as well as to inform local environmental authorities and the region about the potential drivers of change. Additionally, the Dominican Republic possesses a robust coral restoration program that blends sexual and asexual propagation, particularly at the Southeast Reefs Marine Sanctuary (SAMAR in Spanish), where Puntacana Foundation (PCF) and the Dominican Foundation of Marine Studies (FUNDEMAR) have been leading coral restoration efforts for 15 years. More recently, Cap Cana Foundation (CCF) started a restoration program that is contributing to these efforts.

Encompassing 120 km of the southeastern Dominican Republic coast, SAMAR is an MPA co-managed by the Dominican Ministry of Environment (MIMARENA in Spanish), local and international NGOs, such as FPC, CCF and FUNDEMAR, academic institutions and the private sector.



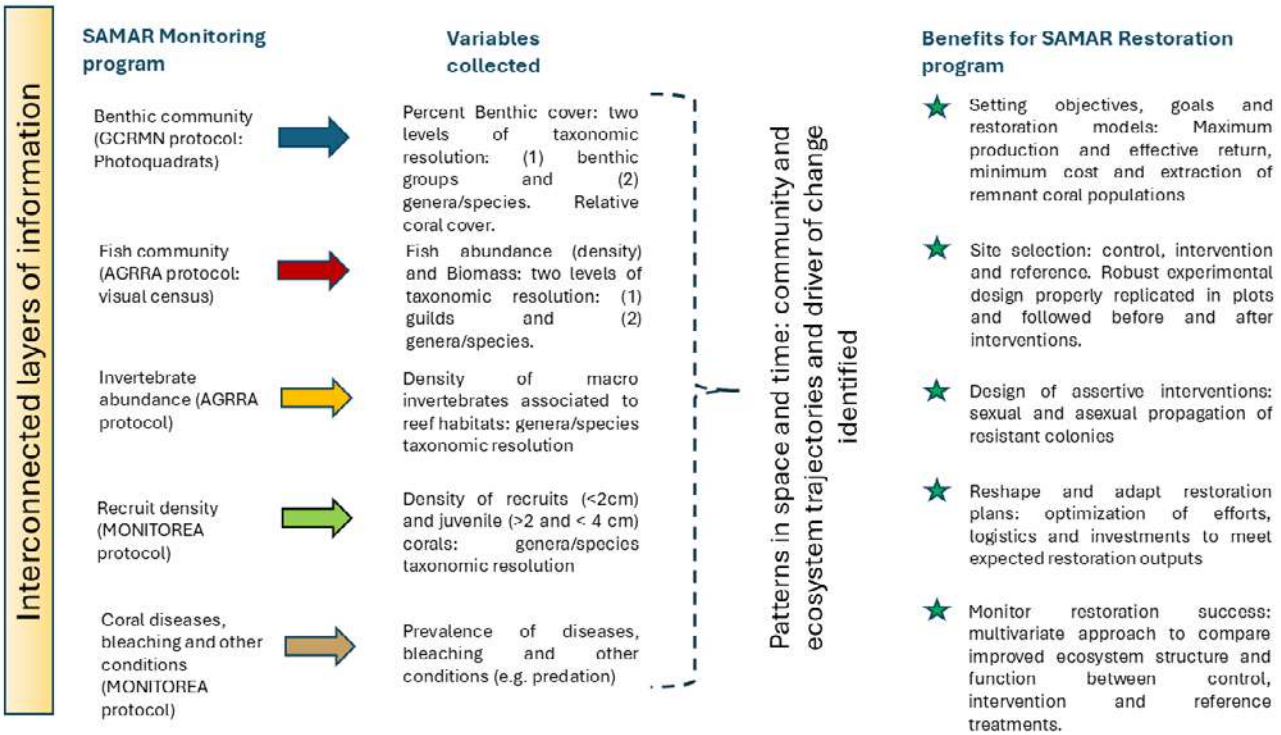
↑ **Figure 1.** Temporal trajectories of benthic communities at SAMAR. Bootstrap metric MDS in a Bray-Curtis ordination space showing shifts and drivers of change in benthic communities at Punta Cana, (A) and Bayahibe (B). Analysis of Principal Coordinates (PCO) upon centroids in a Bray-Curtis Ordination space showing major benthic groups that changed from 2021-2024 in Punta Cana (C) and from 2020-2024 in Bayahibe (D).

In 2018, fifteen co-managers agreed on working together to preserve 1,862.6 km² of marine ecosystem's biodiversity and the myriads of goods and services they provide in three management units: south, center and east. With a Management Plan published in 2024 (Hewson et al., 2023) and expanding for the next 10 years, SAMAR is unique for its co-management model and for having a robust and participative monitoring program designed to guide coral restoration efforts.

The status of coral reefs in the sanctuary is critical, as they have rapidly shifted towards complex pathways characterized by hard coral cover decline and a concomitant increase in macroalgal, octocorals, sponge and benthic cyanobacterial mats cover (**Figure 1.A-D**). Reef-building coral populations normally utilized for coral restoration programs (e.g. *Dendrogyra cylindrus*, *Diploria labyrinthiformis*, *Pseudodiploria strigosa* and *Montastraea cavernosa*, *Acropora palmata* and *A. cervicornis*) have experienced massive mortalities (data not shown). The back-to-back outbreaks of Stony Coral Tissue Loss Disease (SCTLD) and Diadema antillarum sucuticociliatosis (Creary et al., 2023) in 2021 and 2022 and the 2023 massive bleaching events were the drivers of the recent reef declining trajectories recorded at SAMAR (**Figure 1**). SAMAR monitoring program has played a central role to unveil these trends with high precision.

SAMAR monitoring program (SAMAR-MP) uses the MONITOREA protocol (Sellares-Blasco, 2024), which incorporates several methodologies and protocols successfully implemented across the region to describe spatial and temporal trajectories following a spatially nested design replicated twice/three times every year. SAMAR-MP integrates information depicting benthic and fish community structure, density of juvenile corals and abundance of key invertebrates to genera/species taxonomic resolution. In addition, it collects data on prevalence of coral disease, bleaching, predation and other conditions following the same experimental design and taxonomic resolution (**Figure 2**).

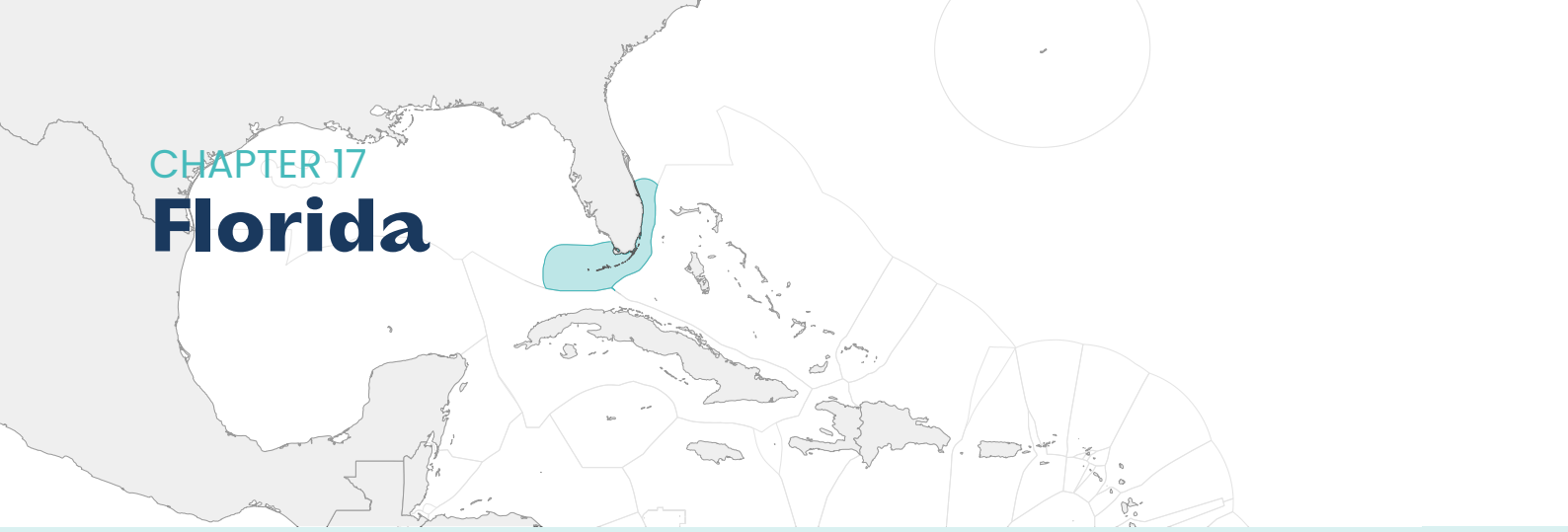
SAMAR-MP strengthens restoration efforts by producing robust multivariate baseline datasets useful for describing spatial and temporal trajectories that reflect the status and progress of coral reefs and by identifying drivers of change (**Figure 2**). The database is open access and available to all stakeholders, co-managers and restoration practitioners within SAMAR. The data helps to set objectives tailored/aligned with the actual status and current trends of SAMAR's coral reefs. SAMAR's monitoring has shaped a coral restoration approach/model that focuses on maximizing coral production and lower costs, while effective return increases at minimum coral extraction practices.



↑ **Figure 2.** Structure of SAMAR monitoring program, variables collected and benefits for SAMAR's coral restoration program.

The integration of multiple layers of information provides quantitative criteria for the appropriate selection of control (*i.e.*, degraded reefs not benefited by restoration), intervention (*i.e.*, degraded reefs benefited by restoration) and reference sites (*i.e.*, natural reefs in better condition compared to the control and intervention sites) (**Figure 2**).

Current restoration efforts at SAMAR entail specific interventions aimed at enhancing key ecological processes such as herbivory and maximizing the survivorship of coral outplants. For example, the acute decline of D. antillarum allowed PCF and FUNDEMAR to understand the need to revitalize herbivory and they started to culture urchins to introduce them into their nurseries and intervention sites. In addition, time series of relative coral cover helped to prioritize which species to harvest in the laboratory using sexual and asexual culture techniques. Lastly, the monitoring program also provides valuable information to decide in advance the best set of strategies to protect corals reared in the laboratory and in the field from stress. In 2022, FUNDEMAR moved about 60 D. cylindrus sexual recruits reared in the laboratory and outplanted in Bayahibe between 2020 and 2022 to FPC's land-based facility to respond to SCTLD outbreak. Furthermore, FUNDEMAR readjusted and increased by an order of magnitude the area to monitor coral spawning and collect gametes bundles to meet production targets of coral spats, whereas FPC has reshaped its asexual propagation program by focusing on those species that survived to SCTLD and bleaching.



CHAPTER 17

Florida

Authors: Erica K. Towle, Rob Ruzicka, Jérémy Wicquart.

Data contributors: Marilyn Brandt*, Michael Colella, Graham J. Edgar, Greg Foster*, Kristi Foster*, Rodrigo Garza*, David Gilliam, Amit Hazra*, Sarah Leinbach, Diego Lirman*, Lizzi Oh, Rob Ruzicka, Stephanie Stinson, Rick D. Stuart-Smith, Erica K. Towle, Joanna Walczak*, Aletta Yniguez*.

*AGRRA data contributor(s).

INTRODUCTION

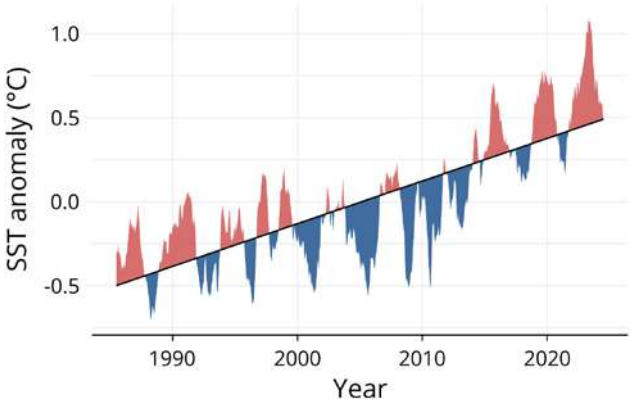
Florida is a continental state of the United States of America, bordered by the Gulf of America on the West and the Atlantic Ocean on the East. Florida hosts around 1,175 km² of coral reefs, which represent 4.8% of the Caribbean's total coral reef extent. Coral reefs are distributed along the Southeast of the peninsula, and all along the Florida Keys, until the Dry Tortugas (**Figure 2.17.4**.). In 2020, around 4.6 million inhabitants were living within 20 km from coral reefs.

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Florida was 26.67 °C. Over this period, SST increased by 1.10 °C, representing a warming rate of 0.028 °C per year (**Figure 2.17.1**).

Several major thermal stress events have affected coral reefs of Florida over the past decades. The 1997–1998 bleaching event represented the first major "shock" to the system, occurring at a time when hard coral cover was still relatively high. As a result, many heat-sensitive species perished, marking a turning point in the degradation of Florida's coral reefs (Ruzicka et al., 2013). Although thermal stress was recorded again in 2005, its impacts in Florida were less severe than in other parts of the Caribbean (Muñiz-Castillo et al., 2019). The next major bleaching episode occurred during the consecutive years of 2014–2015, further weakening coral populations already under stress. More recently, the 2023-2024 marine heatwave produced record-breaking sea temperatures, causing widespread coral mortality. This unprecedented event led to the disappearance of wild *Acropora palmata* populations from approximately two-thirds of the Florida Reef Tract (Stein et al., 2024; Manzello et al., 2025).



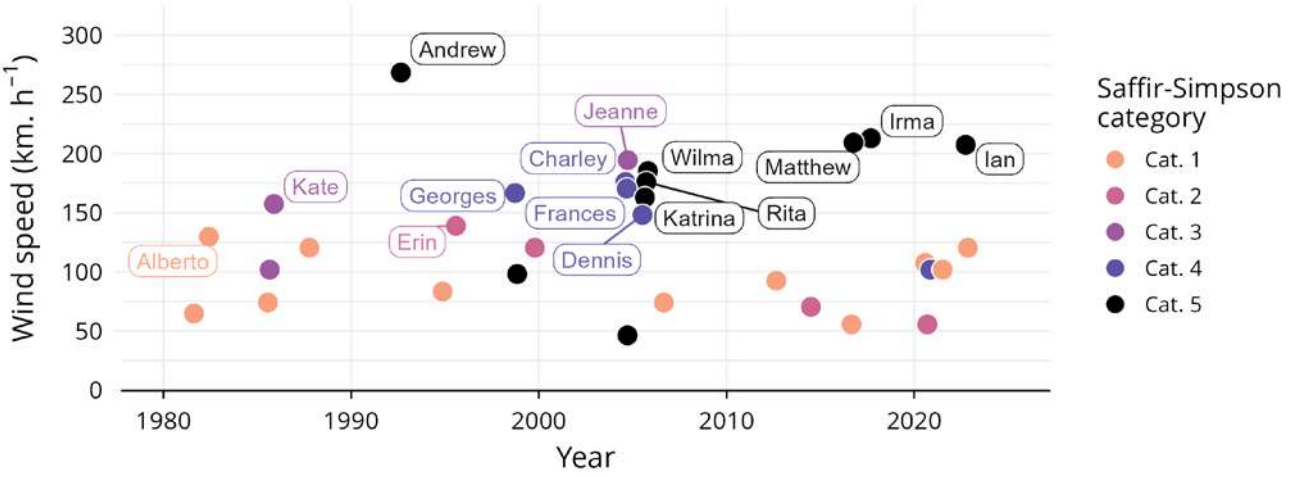
↑ **Figure 2.17.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Florida. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

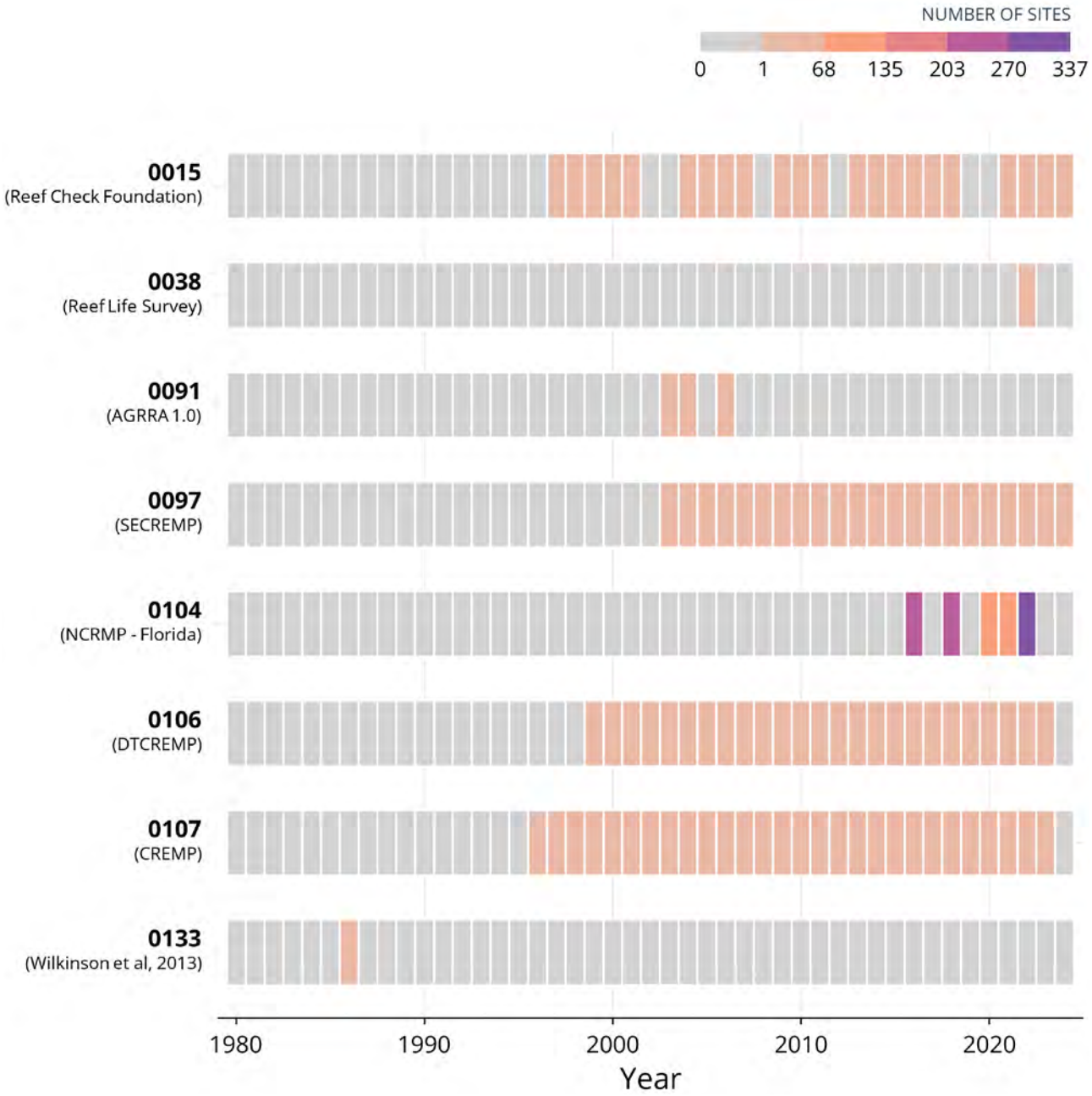
Between 1980 and 2024, a total of 58 tropical storms passed within 100 km distance from a coral reef of Florida, and of these 22 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.17.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Andrew, in 1992, which passed across coral reefs with sustained wind speed of 269 km.h^{-1}.

TRENDS IN BENTHIC COVER

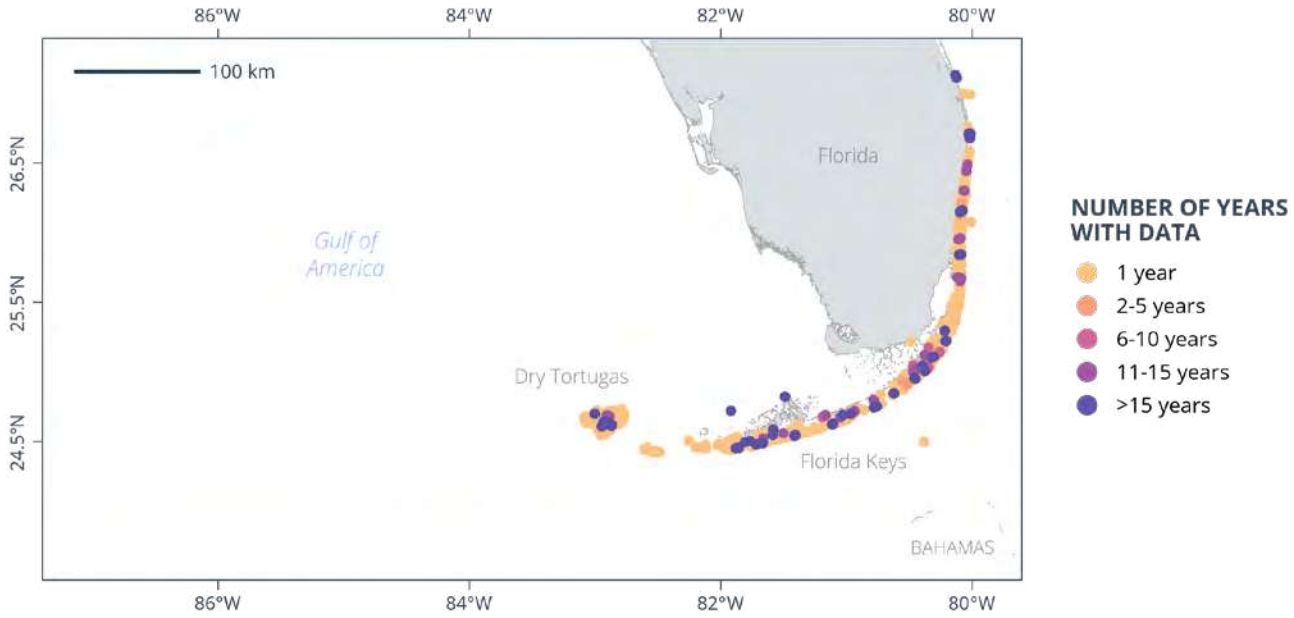
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 8 datasets (**Figure 2.17.3**) encompassing 1,232 monitoring sites. These data were collected through 3,482 surveys conducted between 1986 and 2024.



↑ **Figure 2.17.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Florida. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Florida are not represented, although they can have had an impact.

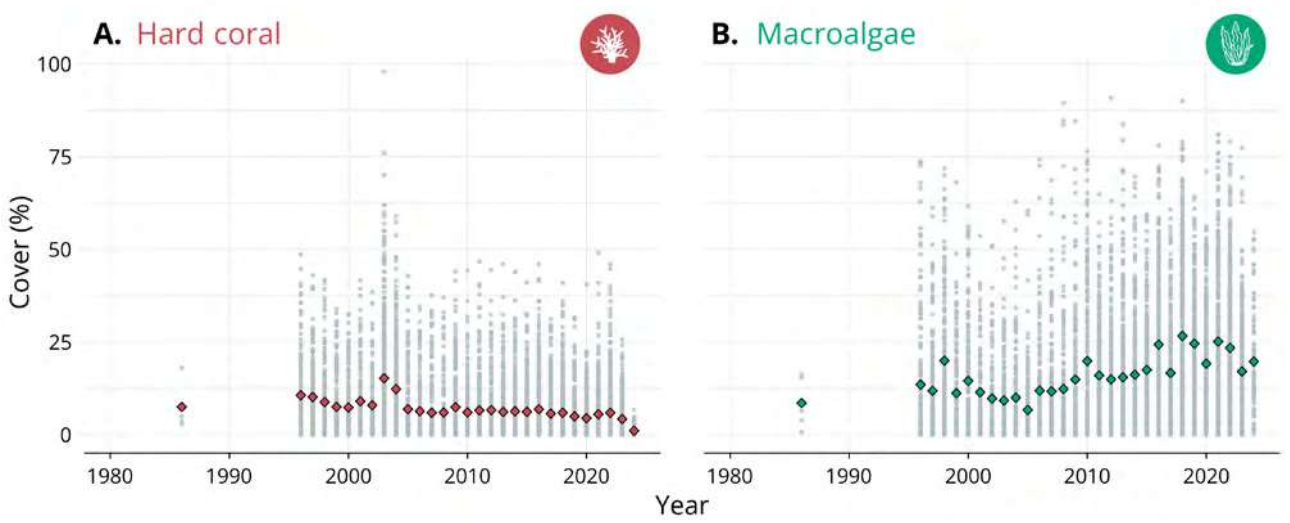


↑ **Figure 2.17.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.17.4.** Spatio-temporal distribution of benthic cover monitoring sites across Florida. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

NCRMP data (included in this analysis) show that coral cover in Florida has continued to decline, particularly in the Florida Keys region. In every Florida region, coral demographics data show that coral species that are highly susceptible to SCTLD, such as *M. meandrites*, have become increasingly rare and, as large colonies have died, population size distributions have shifted to higher proportion of smaller colonies than larger colonies (Viehman et al., 2023).



↑ **Figure 2.17.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Florida from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 18

East and West Flower Garden Banks

Authors: Kelly O'Connell, Michelle A. Johnston, Marissa Nuttall, Erica K. Towle, Shay Viehman, Jérémy Wicquart.

Data contributors: Michelle A. Johnston, Sarah Bernhardt*, Jacque Cresswell, Olivia Eisenbach, Donavon French, Steve Gittings*, Ryan Hannum, Marissa Nuttall, Kelly O'Connell, George P. Schmahl*, Tom Shyka*, Philip Sollins*, Mitchell Tartt*, Erica K. Towle.

*AGRRA data contributor(s).

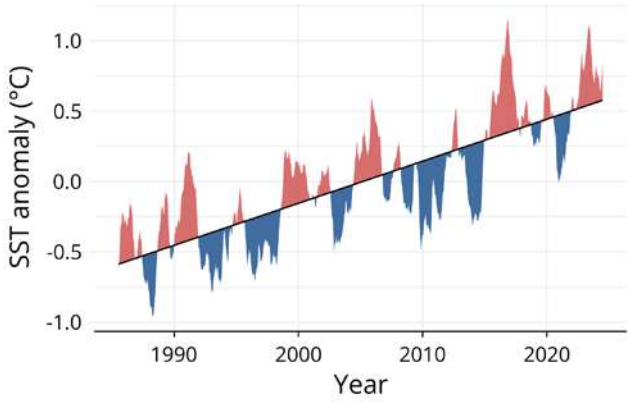
INTRODUCTION

Located within the territorial waters of the United States, Flower Garden Banks National Marine Sanctuary is home to an extensive system of underwater banks about 160 km off the Texas and Louisiana coastlines. Although the Sanctuary encompasses 17 banks in total, only the East and West Flower Garden Banks host significant scleractinian coral communities (ONMS, 2024). Together, these two banks cover approximately 6 km² of coral reefs, representing just 0.02% of the total coral reef area in the Caribbean. These reefs occupy the surface expression of salt dome formations with the shallowest portions of each bank topped by well-developed coral reefs in depths ranging from 16–50 m. Although the coral species found on the reef caps of the banks are the same as those on Caribbean reefs, octocorals are absent in shallow habitats, and scleractinian corals of the genus *Acropora* are exceedingly rare (ONMS, 2024). Due to their remote location and protected status, coral reefs of the Flower Garden Banks are subject to fewer human impacts than many other Caribbean coral reefs.

THREATS

Thermal regime

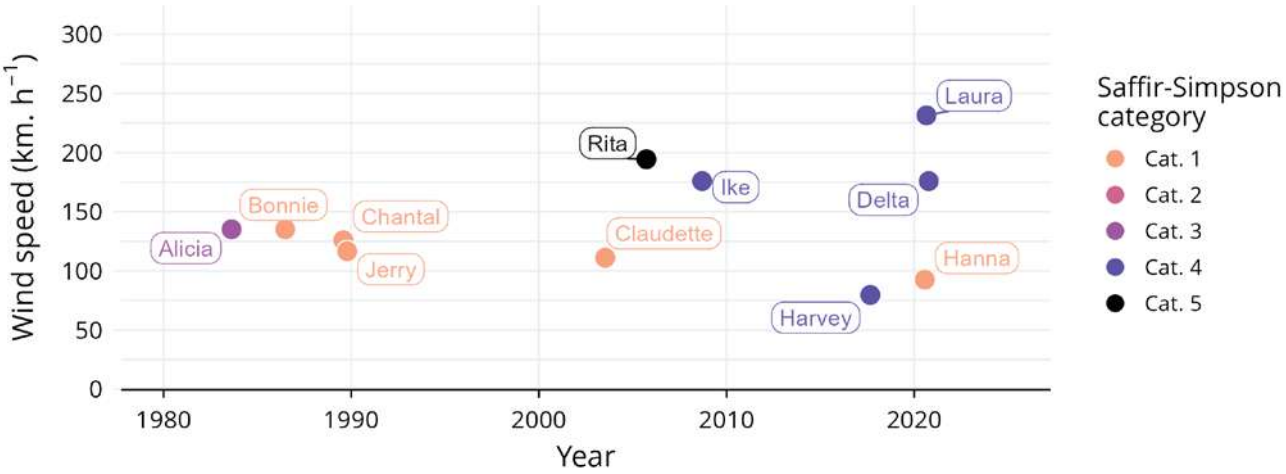
The long-term average in SST from 1985 to 2024 over coral reefs of East and West Flower Garden Banks was 25.27 °C. Over this period, SST increased by 1.39 °C, representing a warming rate of 0.036 °C per year (**Figure 2.18.1**). Elevated ocean temperatures have periodically caused bleaching at the site, though past events were typically short-lived, bleached corals recovered, and overall cover coral remained stable (Johnston et al., 2019). In 2023 and 2024, East and West Flower Garden Banks experienced their first back-to-back bleaching event, signaling a shift toward more frequent thermal stress. While many corals are expected to recover, the recurrence of bleaching in consecutive years highlights increasing pressure on these high-latitude reefs and suggests their historical resilience may be tested more severely under continued thermal stress.



→ **Figure 2.18.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of East and West Flower Garden Banks. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 22 tropical storms passed within 100 km distance from East or West Flower Garden Banks, and nine of these storms were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.18.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Laura, in 2020, which passed 74 km from the banks with sustained wind speed of 232 km.h^{-1}. Increasing storm frequency and intensity raise concern that future hurricanes could cause physical damage to reefs and exacerbate runoff-driven stressors, compounding environmental threats to Flower Garden Banks National Marine Sanctuary (ONMS, 2024).



↑ **Figure 2.18.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 at East and West Flower Garden Banks. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of East and West Flower Garden Banks are not represented, although they can have had an impact.

Other threats

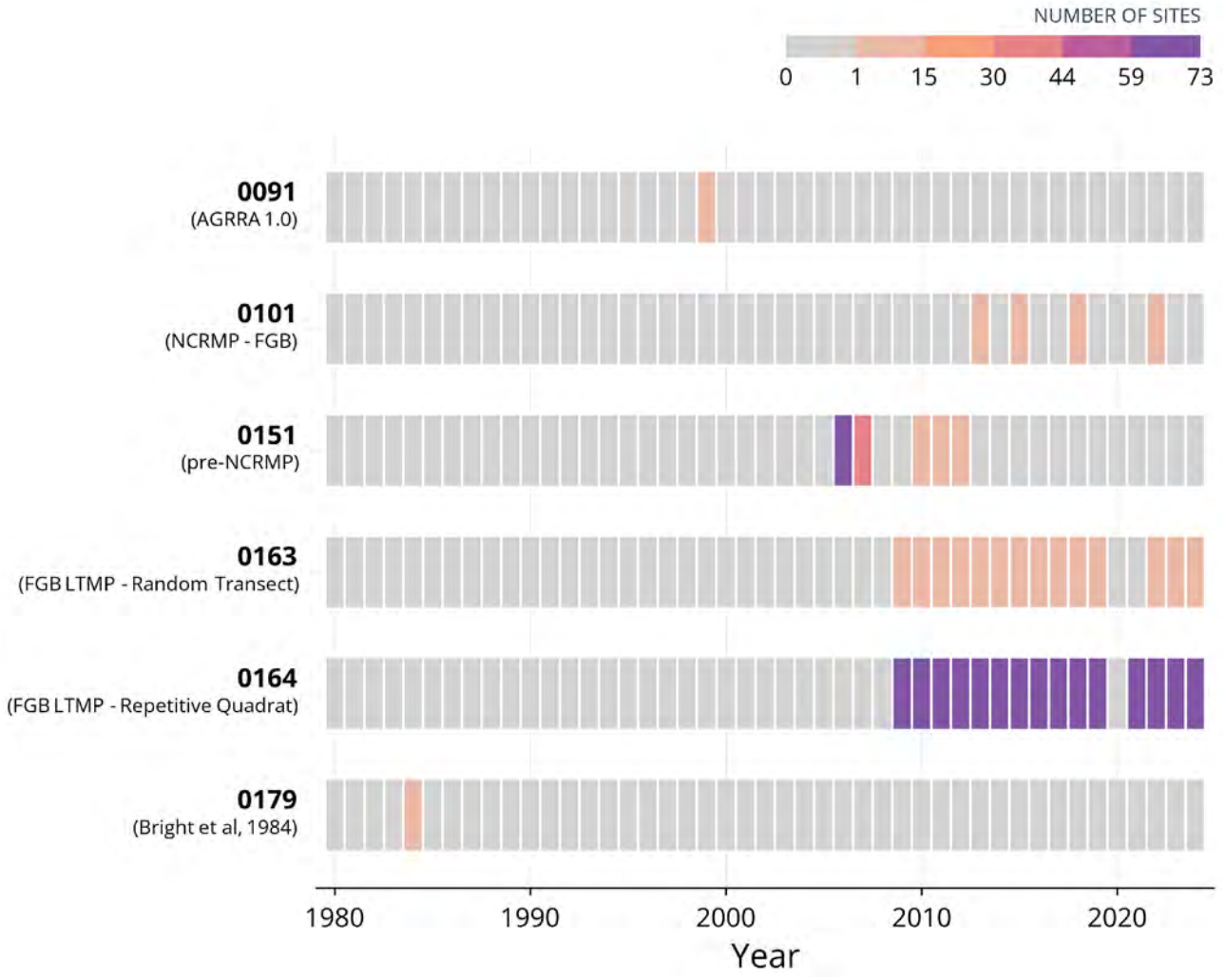
Nonindigenous species represent an emerging concern to the coral reefs at East and West Flower Garden Banks. Orange cup coral (*Tubastraea coccinea*), first reported at East Flower Garden Bank in 2002, is now established across the sanctuary and is abundant on oil and gas platforms throughout the Gulf region. Other introduced species include lionfish (*Pterois volitans*) and the regal demoiselle (*Neopomacentrus cyanomos*), both of which have expanded rapidly across the Gulf over the past decade. While these species are not yet causing measurable degradation within East and West Flower Garden Banks (Blakeway et al., 2022a), their ecological impacts elsewhere, such as predation by lionfish and competition by regal demoiselles, indicate potential risks (Hixon et al., 2016; González-Gándara and de la Cruz-Francisco, 2014). Ballast water discharges from vessels transiting within the sanctuary remain a concern, as they may introduce additional exotic species into sensitive habitats (ONMS, 2024).

While major impacts from oil and gas operations, such as discharges and spills, are rare, they are of great concern to sensitive ecosystems like those in the sanctuary. Located in the northwestern Gulf, East and West Flower Garden Banks are surrounded by extensive offshore energy infrastructure and vessel traffic. A spill or discharge in this region could expose coral reef habitat to hydrocarbons, causing direct damage to benthic organisms and lead to long-term ecosystem damage. Occasional illegal anchoring and cable dragging also threaten localized reef areas (ONMS, 2024).

TRENDS IN BENTHIC COVER

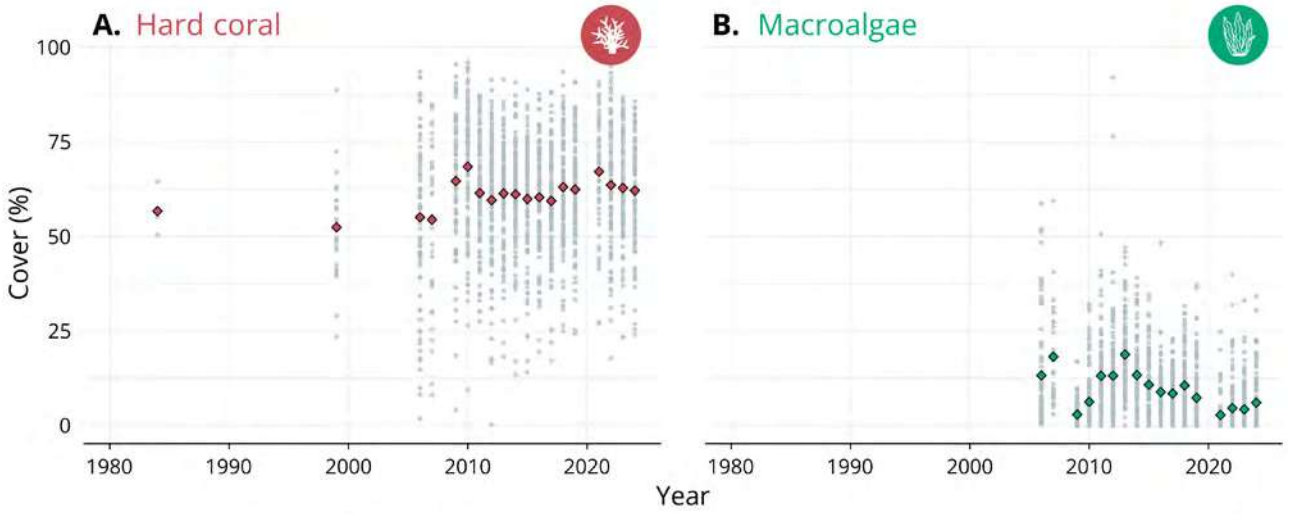
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 6 datasets (**Figure 2.18.3**) encompassing 325 monitoring sites. These data were collected through 1,297 surveys conducted between 1984 and 2024. Early data were collected through limited surveys in the 1980s (*e.g.*, Bright et al., 1984), followed by establishment of the Flower Garden Banks Long-Term Monitoring Program (FGB LTMP) in 1989. This program introduced standardized random transects and repetitive quadrats that have provided continuous, comparable records of benthic cover. Since then, these datasets have been complemented by surveys conducted under the National Coral Reef Monitoring Program (NCRMP), further extending spatial and temporal coverage. The timeline of survey effort shows a steady increase in monitoring intensity, with annual surveys at selected photostations and regular transects, aside from interruptions during the COVID-19 pandemic (**Figure 2.18.3**).

Sanctuary led monitoring has been conducted within permanent 10,000 m² study sites (100 m × 100 m) established on the reef caps of East and West Flower Garden Banks. While these sites provide one of the longest continuous benthic datasets in the wider Caribbean, they represent a relatively small portion of each bank. These surveys offer valuable detail on community dynamics but limited coverage of the broader reef habitats. More recent efforts, including reef-wide stratified random surveys introduced by NCRMP, have helped expand spatial representation.



↑ **Figure 2.18.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

↑ **Figure 2.18.4.** Spatio-temporal distribution of benthic cover monitoring sites across East and West Flower Garden Banks. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.18.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in East and West Flower Garden Banks from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Despite global coral reef declines in recent decades, hard coral cover within East and West Flower Garden Banks has remained consistently high, near or above 50% throughout years of monitoring (**Figure 2.18.5A**, Viehman et al., 2023; Johnston et al., 2024; Eisenbach et al., 2025). Despite disturbances such as bleaching, disease, and a localized mortality event in 2016, hard coral cover has not shown any lasting declines, with evidence of recovery following impacts (Johnston et al., 2018, Johnston et al., 2019, Johnston et al., 2023). This unusual persistence is attributed to the reefs' offshore location, surrounding deep water that buffers environmental fluctuations, and management protections that have minimized local human impacts (Eisenbach et al., 2024).

CHAPTER 19

Grenada

Authors: Robert Anderson, Laëtitia Mathon.

Data contributors: Werner Nagel*, Denzel Adams*, Robert Anderson, Robert Balza, Ethan Degner, Jerry Enoe*, Olando Harvey*, James Henkel, Philip Kramer*, Lindy C. Knowles*, Judith Lang*, Courtney Moll, Clare Morrall*, Stephen Nimrod, Jeremy Searles*, Thomas Sweeney*, Katlynd Treiber-Vajda*, Sandy Voegeli*.

*AGRRA data contributor(s).

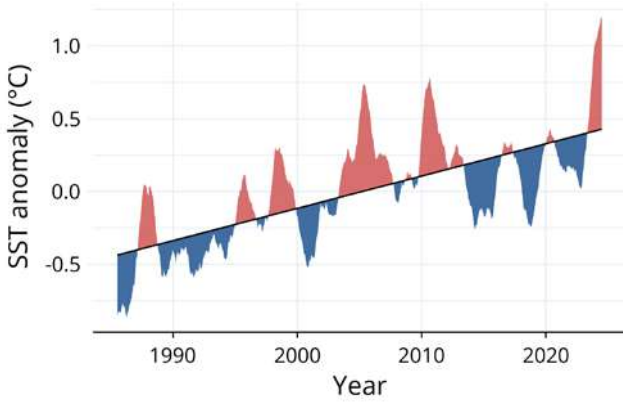
INTRODUCTION

Grenada, a small island nation in the southeastern Caribbean with a land area of 344 km² and an Exclusive Economic Zone spanning 27,426 km², is home to 110 km² of coral reefs—about 0.455% of the region's total reef extent. These reefs are not only central to the island's ecological health, providing vital ecosystem services, but also underpin key economic activities such as fisheries and tourism. Between 2010 and 2016, Grenada's marine ecosystem services were quantified in Eastern Caribbean dollars: fish and shellfish landings ranged from EC$32.5 million to EC$40.4 million annually; coastal protection benefits from reefs were valued stably at EC$1,313.3 million per year; and recreational expenditures amounted to EC$398–419 million yearly (Mengo et al., 2022). Grenada's population has grown from 94,500 in 1980 to 117,200 in 2024, increasing pressures on coastal environments and highlighting the importance of marine protection. In recognition of this, the country has established several marine protected areas, including Molinier-Beauséjour (1999), Sandy Island–Oyster Bay (2009), Woburn–Clarks-Court Bay (1999), and Grand Anse (2018), aimed at safeguarding biodiversity while sustaining livelihoods.

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Grenada was 27.78 °C. Over this period, SST increased by 1.03 °C, representing a warming rate of 0.026 °C per year (**Figure 2.19.1**).



↑ **Figure 2.19.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Grenada. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

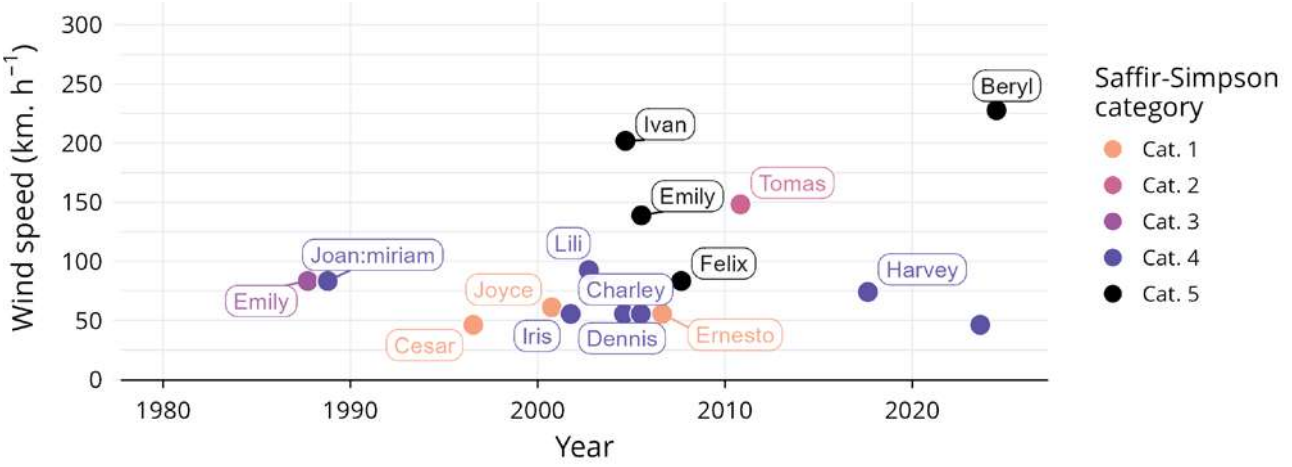
Hurricanes

Between 1980 and 2024, a total of 27 tropical storms passed within 100 km distance from a coral reef of Grenada, and of these 5 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.19.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Beryl, in 2024, which passed 0 km from a coral reef with sustained wind speed of 228 km.h^{-1}.

Other threats

Illegal dumping is a major threat to coral reefs in Grenada (Graham, 2024). About 35,000 cubic feet of waste were estimated from 29 unauthorized dumpsites. The waste ranged from construction debris and vehicles to plastics, medical waste, and treated wood. Such pollutants release toxins and debris that endanger marine life and disrupt reef ecosystems.

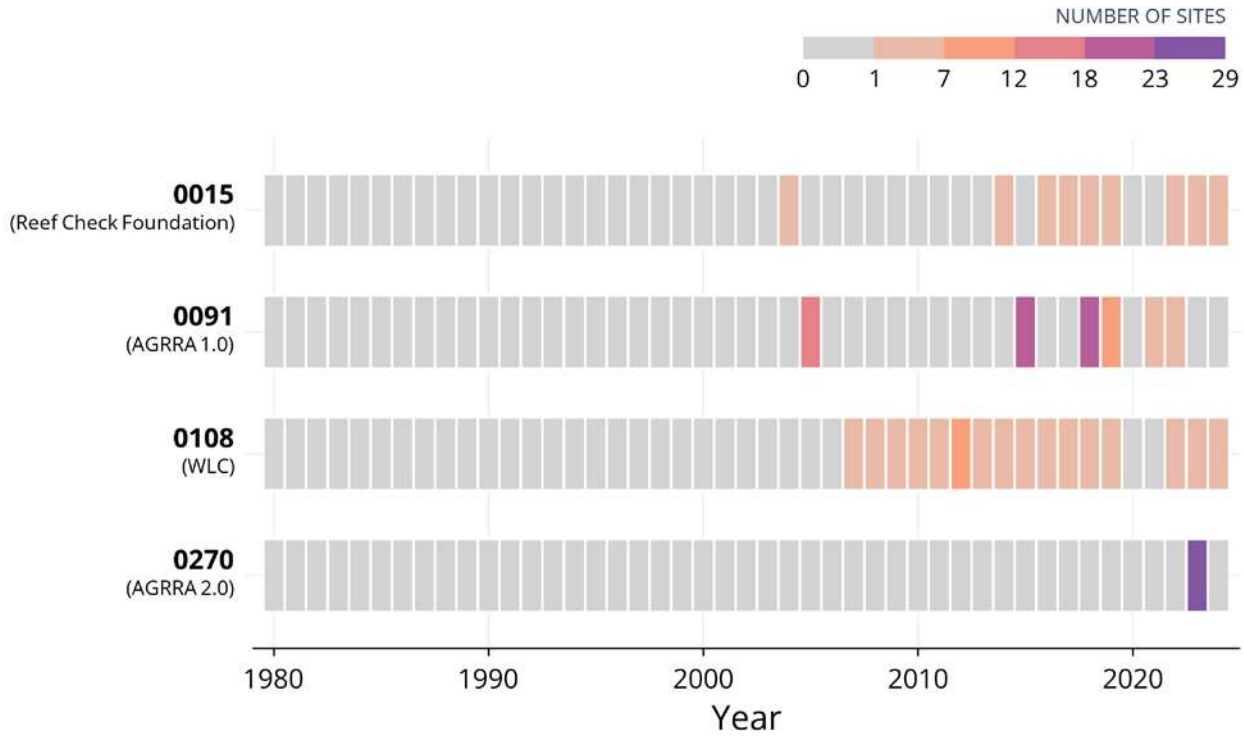
↓ Figure 2.19.2. Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Grenada. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Grenada are not represented, although they can have had an impact.

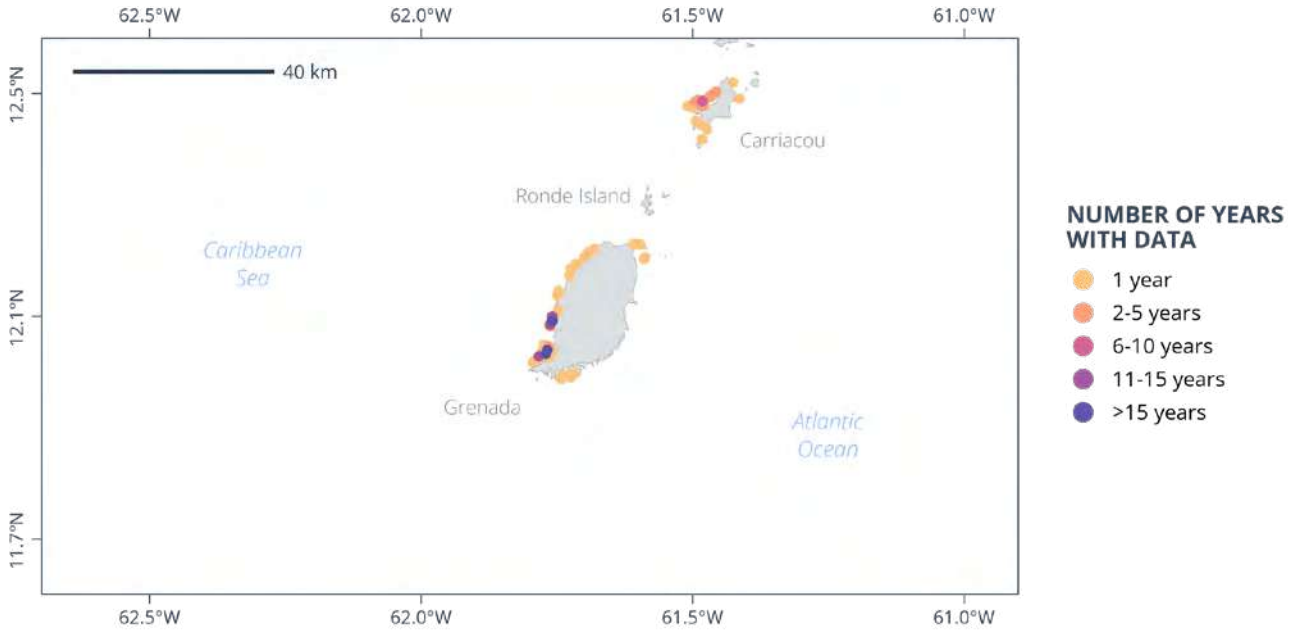


TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 4 datasets (**Figure 2.19.3**) encompassing 99 monitoring sites (**Figure 2.19.4**). These data were collected through 239 surveys conducted between 2004 and 2024.

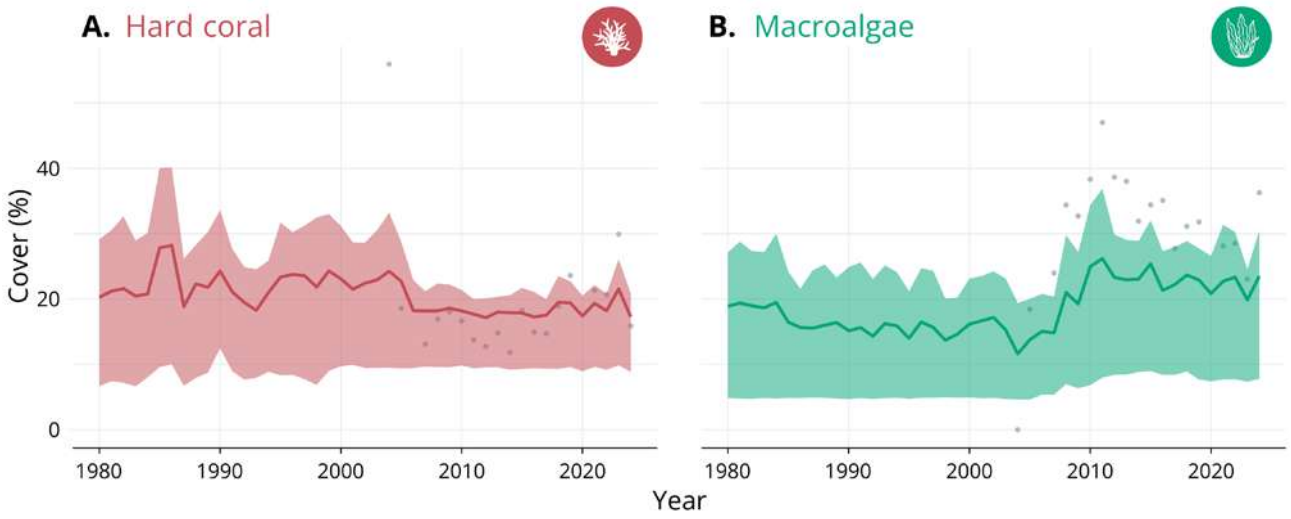
↓ Figure 2.19.3. Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.19.4.** Spatio-temporal distribution of benthic cover monitoring sites across Grenada. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Coral reef studies in Grenada show long-term shifts marked by declining coral cover, rising algal dominance, and changes in fish populations. Hurricanes Ivan and Emily (in 1993 and 2004) and the 2005 coral bleaching event led to significant loss of live coral (Kramer and Dowksza 2005). Surveys from 2006–2007 (Bouchon et al., 2008) found macroalgae as the dominant benthic cover (36–53%), with hard corals making up the second largest component and *Diadema* urchins largely absent. Between 2008 and 2012, research showed increasing algal cover, particularly macroalgae, while live hard coral cover remained relatively low (8–21%), with branching corals, especially *Porites porites* and *P. astreoides*, being the most common (Anderson et al., 2012; 2014). Fish densities declined overall, though wrasses increased in abundance, and *Chromis* spp. consistently dominated surveys. Despite a sea urchin restocking effort in 2011, populations remained very low. By 2014, monitoring at permanent sites within Grenada's Marine Protected Areas revealed spatial variability in reef health (Philips et al., 2015). Coral cover ranged from 22–35% across sites, generally considered good, but fleshy macroalgae cover was often high, particularly in popular tourist areas. Herbivorous and commercially important fish biomass remained low at most sites, while *Diadema* populations were highly variable, ranging from absent to abundant depending on location. More recent monitoring indicated a shift in *P. astreoides* populations, with fewer colonies but larger sizes and coverage, suggesting possible recovery following disturbance (Eagleson et al., 2021). Overall, Grenada's reefs show a trend of algal dominance, reduced coral cover, and limited recovery dynamics shaped by disturbance events, with local variability influenced by protection status and human activity.



↑ **Figure 2.19.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Grenada from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

CHAPTER 20

Guadeloupe

Authors: Claude Bouchon, Yolande Bouchon-Navaro, Charlotte R. Dromard, Mohsen Kayal, Jérémy Wicquart.

Data contributors: Claude Bouchon, Yolande Bouchon-Navaro, Julien Chalifour, Charlotte R. Dromard, Sébastien Greaux, Mohsen Kayal, Florian Labadie, Jean-Claude Lalanne, Sophie Le Loc'h, Raoul Lebrave, Mylène Musquet, Léa Sebesi, Pedro Portillo.

INTRODUCTION

Guadeloupe, a French overseas territory in the Lesser Antilles, lies between Montserrat to the north and Dominica to the south. Guadeloupe is composed of the two main islands of Basse-Terre and Grande-Terre, along with several smaller islands such as Les Saintes, Marie-Galante, Petite-Terre, and La Désirade (**Figure 2.20.4**). Coral reef extent is about 233 km², which represents nearly 1% of the Caribbean's total coral reef extent. Guadeloupe is home to a barrier reef along its northern coast, fringing reefs on its windward shores, and rich, various non-reef-building coral formations on its leeward coast (Bouchon et al., 2008).

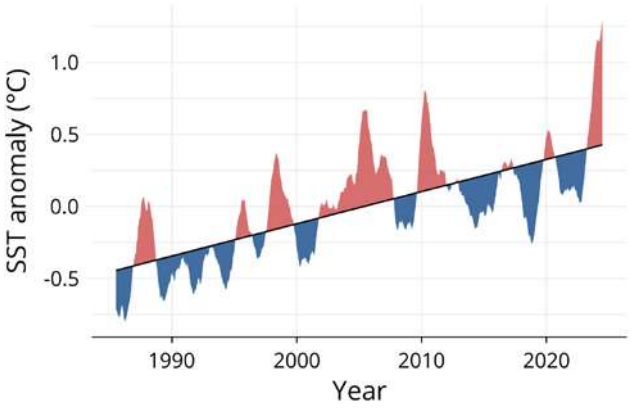
In 2022, Guadeloupe had about 383,500 inhabitants, with a population density of 235 inhabitants per km² (INSEE, 2022). The local economy is primarily based on export-oriented agriculture, particularly bananas and sugarcane, as well as tourism (Bouchon et al., 2008).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Guadeloupe was 27.65 °C. Over this period, SST increased by 1.03 °C, representing a warming rate of 0.027 °C per year (**Figure 2.20.1**).

In the French West Indies, the first significant coral bleaching events were recorded in 1984 and 1987, associated with El Niño phenomena; however, their impacts on coral reef ecosystems were limited.



↑ **Figure 2.20.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Guadeloupe. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

In contrast, the 1998 event affected 56% of coral colonies in Guadeloupe, with post-bleaching mortality estimated between 20% and 30%, depending on the site (Bouchon and Bouchon-Navaro, 1998). In September 1999, a new bleaching episode impacted approximately 50% of corals in the Guadeloupe archipelago. Monitoring efforts were disrupted by damage caused by Hurricane Lenny in November of the same year.

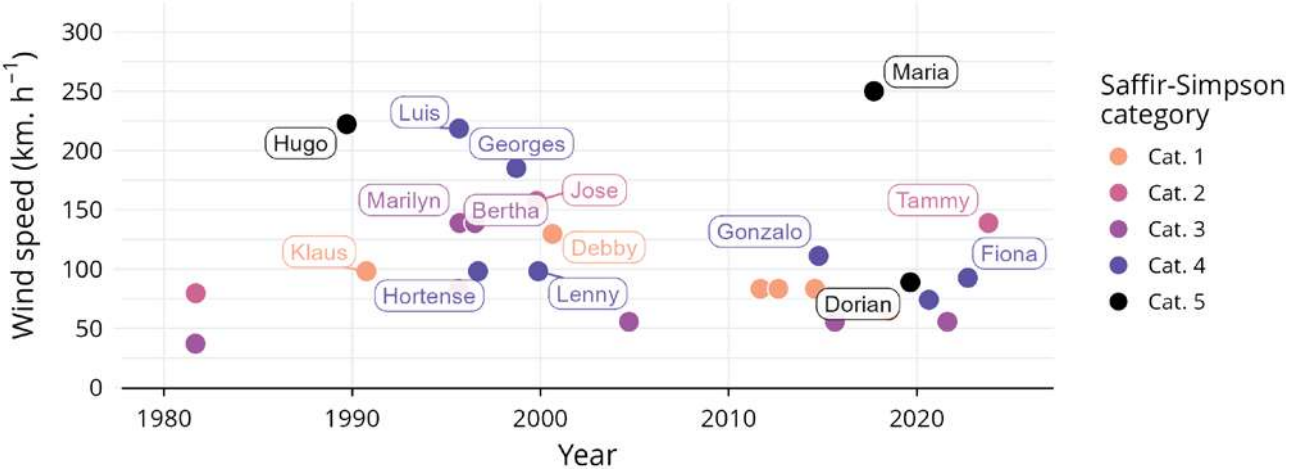
The next major event occurred in 2005, extensively affecting both the Lesser and Greater Antilles, and to a lesser extent, the broader Caribbean region (Eakin, 2010). In the French West Indies, nearly 80% of coral colonies experienced bleaching, and delayed mortality observed in 2006 led to a reduction in coral cover ranging from 40% to 60%, depending on the location (Bouchon et al., 2008b).

In 2019, another bleaching event was documented in Guadeloupe. Although less severe than the 2005 episode, its intensity was comparable to that of the events in 1998 and 1999. Approximately 50% of coral species were affected, with an average bleaching rate of 26% among impacted colonies. Unlike previous events, particularly that of 2005, coral mortality associated with this episode appeared to be low, and no significant decline in coral cover was observed by the end of 2020.

Between April and November 2023, coastal water temperatures in Guadeloupe exceeded the critical threshold of 29 °C, reaching up to 32 °C at specific sites in September (Bouchon et al., 2024a). In 2024, another bleaching event again affected Guadeloupean reefs. Temperatures returned below the critical threshold in December (Bouchon et al., 2024b). These recent episodes were unprecedented in both intensity and duration, marking the first occurrence of two consecutive years of prolonged thermal stress. These events resulted in substantial degradation of coral reef ecosystems, with massive mortality among the most sensitive species, particularly acroporids (90–100%). Among all species, colony mortality approached 35%, resulting in a drastic reduction in coral cover, estimated at approximately 50% between 2022 and 2025 (Bouchon et al., 2025).

Hurricanes

Between 1980 and 2024, a total of 37 tropical storms passed within 100 km distance from a coral reef of Guadeloupe, and of these 10 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.20.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Maria, in 2017, which passed 27 km from a coral reef with sustained wind speed of 250 km.h^{-1}. However, Hurricane Hugo (1989) caused the most severe ecological impact, primarily damaging branching corals from the surface to a depth of 15 m. Post-storm effects consisted in a delayed coral mortality and a bloom of Rhodophyte algae from the genus *Liagora* (Bouchon *et al.*, 1991).



↑ **Figure 2.20.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Guadeloupe. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Guadeloupe are not represented, although they can have had an impact.

Coral diseases

Scleractinian corals in the French Antilles are affected by approximately fifteen known diseases, the most recent being Stony Coral Tissue Loss Disease (SCTLD), first identified in Florida in 2014. SCTLD has spread throughout the Caribbean via marine currents and ship ballast waters, reaching Guadeloupe in May 2020. The disease rapidly impacted all local reefs, affecting 22 of the 60 coral species known in the French Antilles (Bouchon and Laborel, 1990). Highly vulnerable species, such as *Dendrogyra cylindrus* and *Meandrina meandrites*, have nearly disappeared. SCTLD has caused a sharp decline in coral abundance, which has been further exacerbated by widespread bleaching events between 2023 and 2025.

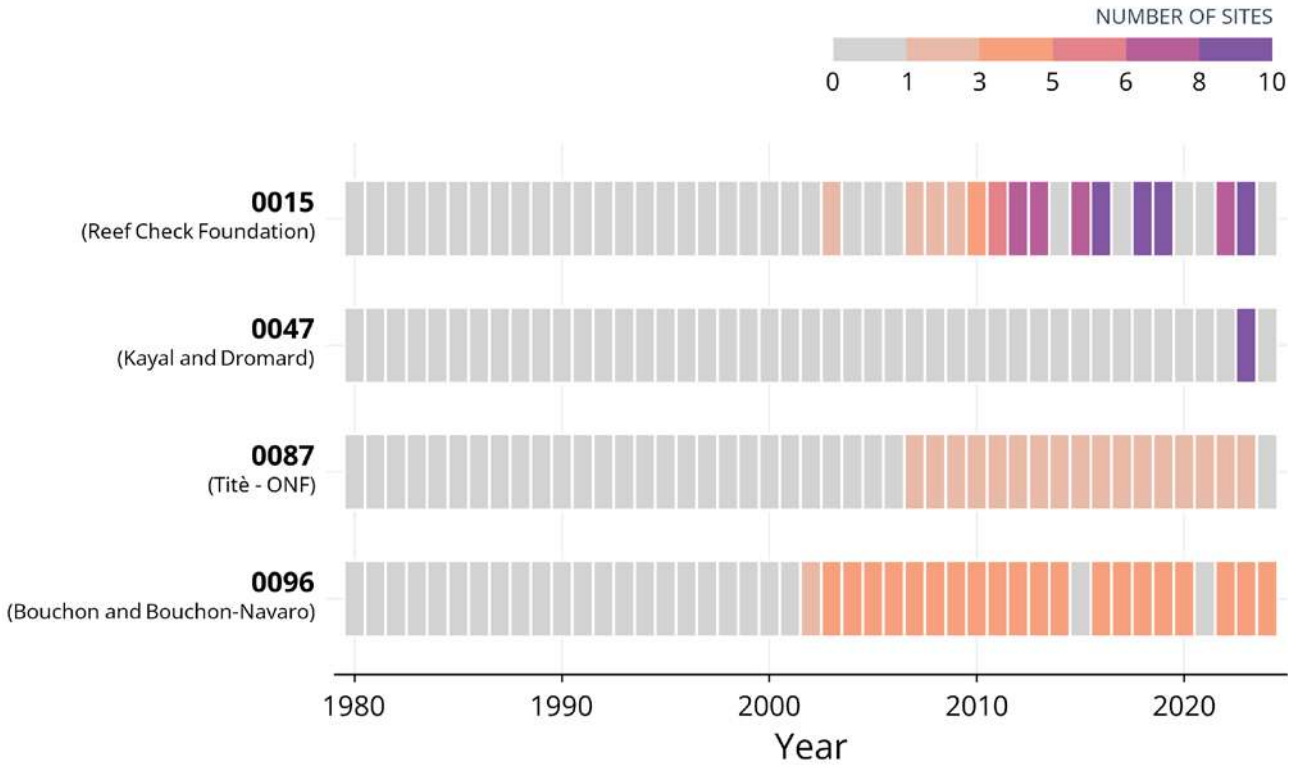
Coastal pollution

Guadeloupe's dense population contributes to significant anthropogenic stress on its marine ecosystems. Agricultural runoff, deforestation, and industrial effluents have led to nitrate enrichment and organic pollution in coastal waters. Despite the presence of ~70 wastewater treatment plants, frequent malfunctions reduce their effectiveness. In 2022, 55% of the water bodies assessed exhibited a poor ecological status, while 45% were classified as having a moderate status, based on biological, physico-chemical, and hydromorphological indicators (Keitz et al., 2022). Since the 1980s, brown macroalgae, especially *Dictyota*, have proliferated, competing with corals and benthic fauna. Overfishing of herbivorous fish (Scaridae, Acanthuridae) intensifies algal dominance.

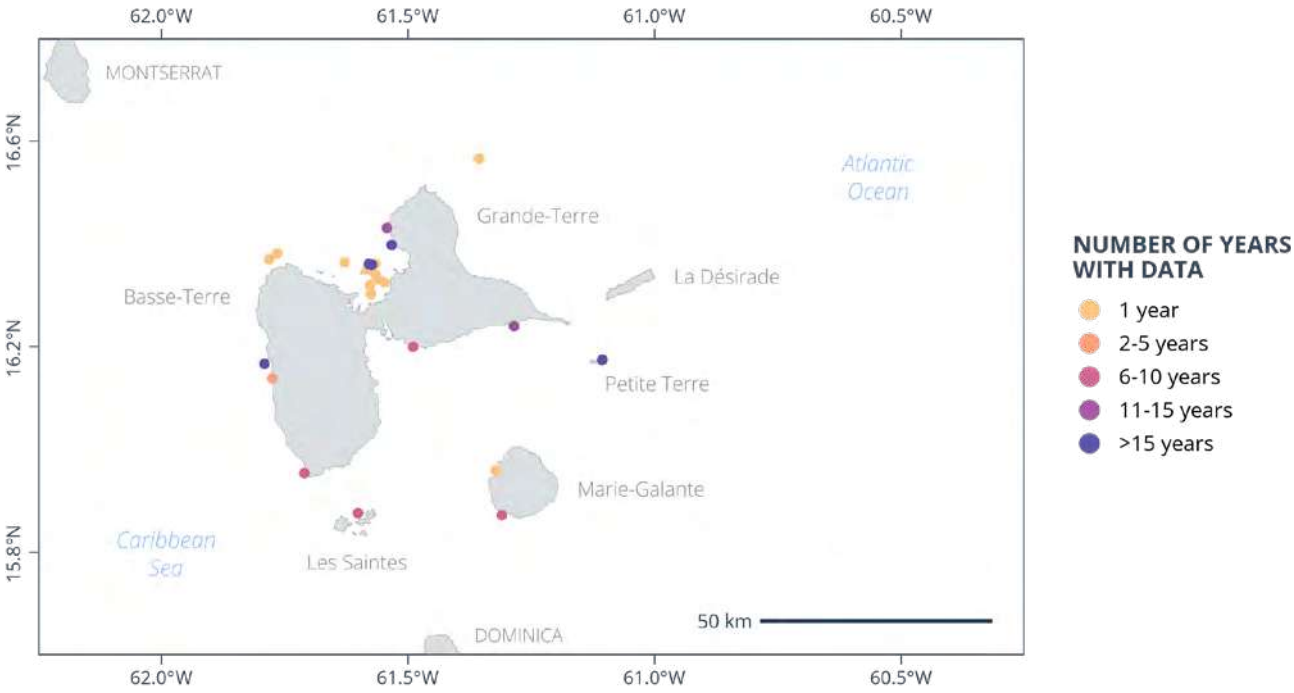
Additionally, heavy metals — including lead, cadmium, vanadium, copper, zinc, and tin — have been detected in marine environments, alongside approximately 2,000 metric tons of annual pesticide imports. These pollutants pose reproductive and developmental risks to marine species, though their full ecological impact remains poorly understood. Furthermore, reef organisms in Guadeloupe are chronically exposed to chlordecone (Dromard et al., 2018a), an organochlorine pesticide used between 1972 and 1993 in banana plantations, which is still highly present in the soil due to its persistence in the environment. The level of contamination by chlordecone remains very high in the food web, even in coral reefs located more than 3 km from the coast (Dromard et al., 2018b). Although the ecotoxicological impact of this molecule has not been tested on all reef organisms, genotoxicity has been demonstrated in a controlled environment (Wecker et al., 2018).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 4 datasets (**Figure 2.20.3**) encompassing 26 monitoring sites. These data were collected through 209 surveys conducted between 2002 and 2024.

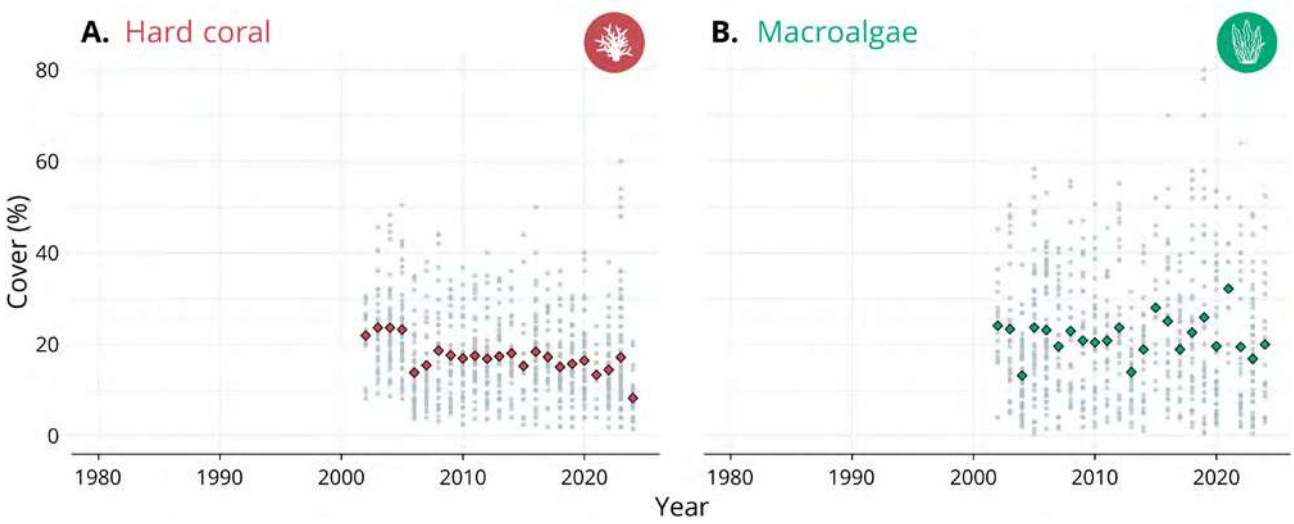


↑ **Figure 2.20.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.20.4.** Spatio-temporal distribution of benthic cover monitoring sites across Guadeloupe. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Quantitative monitoring of coral reef dynamics in Guadeloupe formally began in 2002 with the establishment of the GCRMN network across four sites (**Figure 2.20.3**). However, quantitative surveys using line intercept transects were conducted during various environmental studies throughout the 1980s and 1990s. These surveys provide coral cover estimates for three of the four Guadeloupe's maritime facades: ranging from 26% to 40% at Pigeon Islet, 46% on the outer slope of a reef along the Atlantic coast, and between 15% and 20% on reef flats of the same coastal sector. Additionally, coral cover values of approximately 24% to 28% were recorded on the outer slope of the barrier reef in the Grand Cul-de-Sac Marin Bay and in the Passe-à-Colas (northern facade of Guadeloupe), 14% on the barrier reef flat, and between 27% and 36% on coral formations within the lagoon. Between 2000 and 2020, data from the GCRMN indicate a gradual decline in coral cover across the reefs of Guadeloupe (**Figure 2.20.5**). This trend was sharply exacerbated in subsequent years due to significant coral mortality events triggered by the SCTLD disease, followed by bleaching episodes in 2005 and 2023-2024. By the end of 2024, coral cover rates ranged between 6% and 9%, depending on the reef site.



↑ **Figure 2.20.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Guadeloupe from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

CHAPTER 21

Guatemala

Authors: Ana Giro, Andrea Rivera-Sosa.

Data contributors: Ana Giro*, Ángela Mojica*, Alerick Pacay*, Hazel Araujo*, Marieta Marroquin*, Ana Beatriz Rivas*, Andrés B. Alegría*, Tanya Barona*, Ana C. Fonseca*, Michelle Fernández*, Maria Jose Gonzalez Bernat*, Melanie McField*, Helder Pérez*, Alicia Rodríguez*.

*AGRRA data contributor(s).

INTRODUCTION

Guatemala's Caribbean coastline extends for approximately 148 kilometers (Rosales-Melgar et al., 2024) and is home to 24 coastal communities whose livelihoods, identity, and food security are deeply connected to the sea. Along this coast lies a remarkable mosaic of interconnected ecosystems, wetlands, mangroves, seagrass meadows, and coral reefs, that support diverse marine life and provide essential services such as coastal protection, fisheries, and tourism. Together, these ecosystems sustain not only biodiversity but also the cultural and economic foundations that shape community well-being and resilience along Guatemala's Caribbean shore.

Guatemala hosts approximately 92 km² of coral reefs, representing about 0.378% of the total coral reef area in the Caribbean. These reefs are primarily located within two regions: the Punta de Manabique Wildlife Refuge and the offshore Cayman Crown Reef, at depths of roughly 10–22 meters. Nearshore reefs are strongly influenced by high sedimentation and turbidity resulting from the outflow of three major rivers, the Río Dulce, Sarstún, and Motagua, which deliver large volumes of freshwater and sediments into the Caribbean. Despite these natural and human-driven stressors, Guatemala's reefs remain ecologically significant, sustaining fisheries, supporting biodiversity, and buffering coastal communities. Yet, in the past decade, they have shown signs of degradation, particularly increases in fleshy macroalgae and declines in herbivorous and commercially important fish biomass (Giró, 2020).

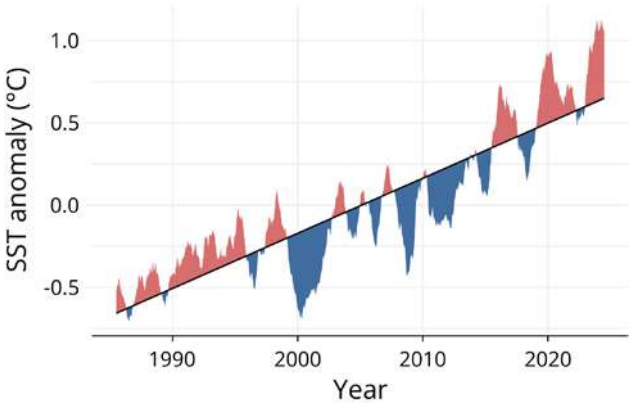
THREATS

Thermal regime

Guatemala's coral reefs have experienced significant impacts from several major bleaching events. The long-term average SST from 1985 to 2024 across Guatemala's coral reef areas was 28.18 °C. Over this period, SST increased by 1.39 °C, corresponding to a warming rate of 0.036 °C per year (**Figure 2.21.1**). This rate is higher than the average for the wider Caribbean region and approaches the upper range of warming observed across Caribbean countries and territories. Although coral communities in this region appear to exhibit some degree of adaptation or acclimatization to naturally high and variable temperatures with some colonies showing partial recovery between bleaching events, the continued rise in SST is narrowing the thermal margin for these reefs.

Bleaching severity on Guatemala's offshore reefs has varied over time. In 2016, approximately 44.9% of corals exhibited bleaching, increasing to 54.3% in 2017. During 2019, elevated SST conditions corresponded with a sharp rise in bleaching prevalence, with 76.8% of corals affected. By 2021, bleaching levels declined to around 30%, with most colonies observed in a pale (20%) or partially bleached (10%) condition (Giró and Mojica, 2022). These fluctuations reflect the dynamic response of coral communities to recurring thermal stress, as well as their variable capacity for recovery between events.

The 2023–2024 bleaching event has been particularly severe. Prolonged periods of elevated SST were recorded both by satellite (**Figure 2.21.1**) and in situ at sentinel sites such as the Cayman Crown reef, where continuous monitoring with HOBO temperature loggers confirmed sustained thermal stress. Field observations documented widespread coral paling and, in some cases, complete bleaching. These recurrent thermal stress events not only reduce live coral cover but also disrupt reef community structure, critically affecting associated fish populations in the country. The cumulative effects of repeated bleaching underscore the vulnerability of these reefs and highlight the urgent need for continued monitoring and management strategies to mitigate the impacts of climate-driven warming.

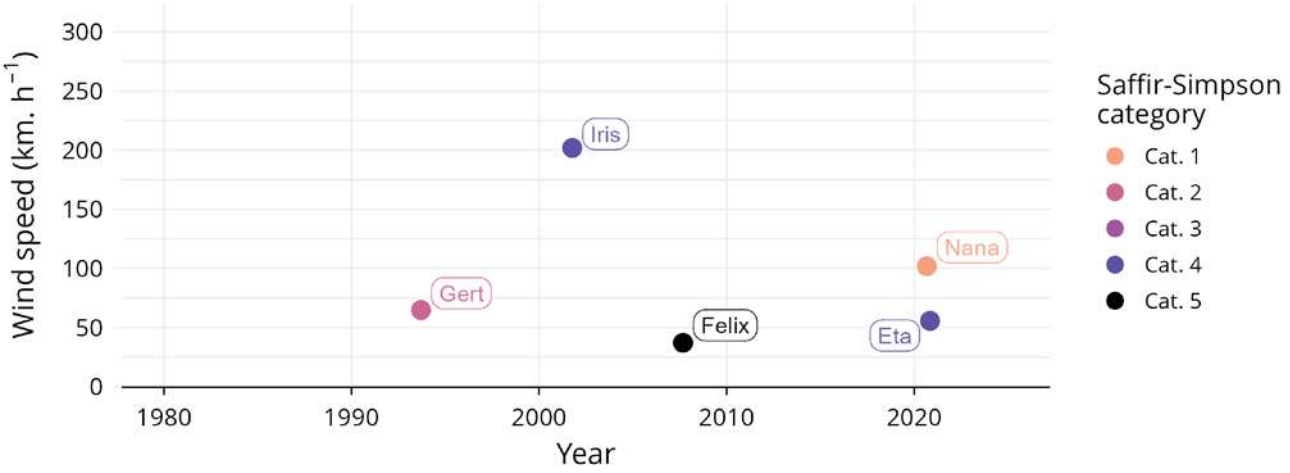


↑ **Figure 2.21.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Guatemala. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 13 tropical storms passed within 100 km distance from a coral reef of Guatemala, and of these 2 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.21.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Iris, in 2001, which passed 18 km from a coral reef with sustained wind speed of 202 km.h^{-1}.

Although hurricanes have rarely made a direct hit on Guatemala's coast, the reefs in this area are nonetheless affected by indirect impacts such as Hurricane Mitch in 1998. River runoff and coastal flooding associated with these storms carry sediments, nutrients, and debris into the ocean, which can smother corals, reduce water quality, and disrupt reef ecosystems. Therefore, while wind and wave damage may be limited, the effects of hurricanes on Guatemalan reefs are real and largely mediated through terrestrial inputs into the marine environment.



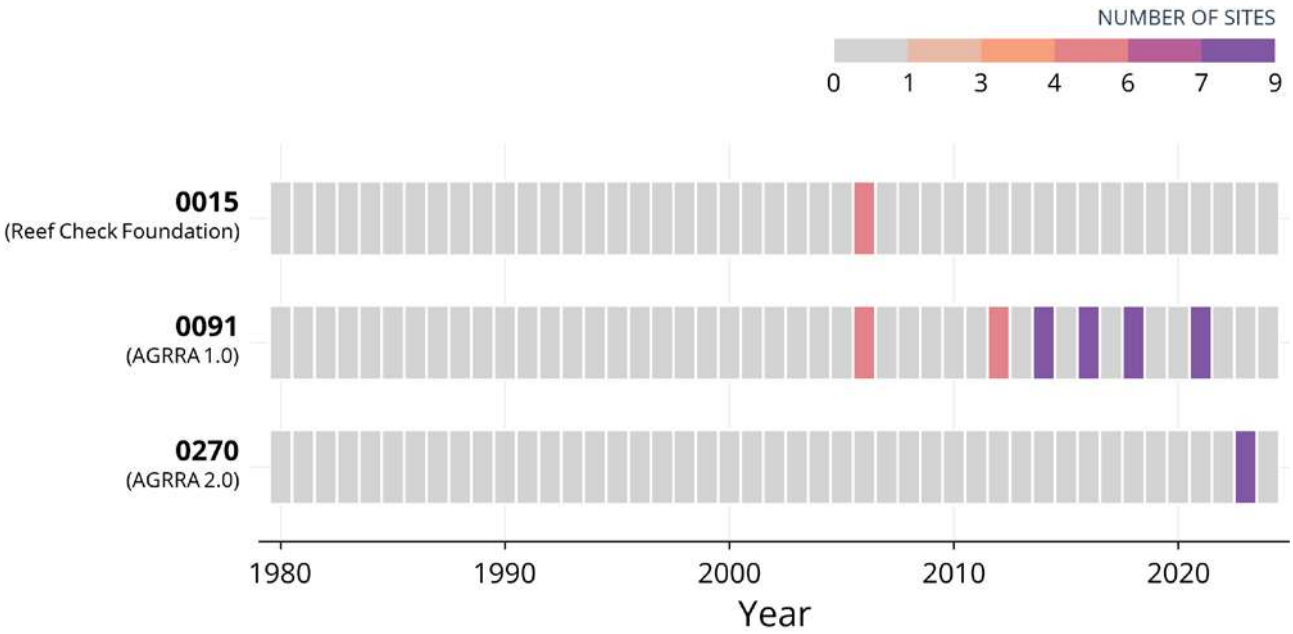
↑ **Figure 2.21.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Guatemala. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Guatemala are not represented, although they can have had an impact (*e.g.* hurricane Mitch in 1998).

Other threats

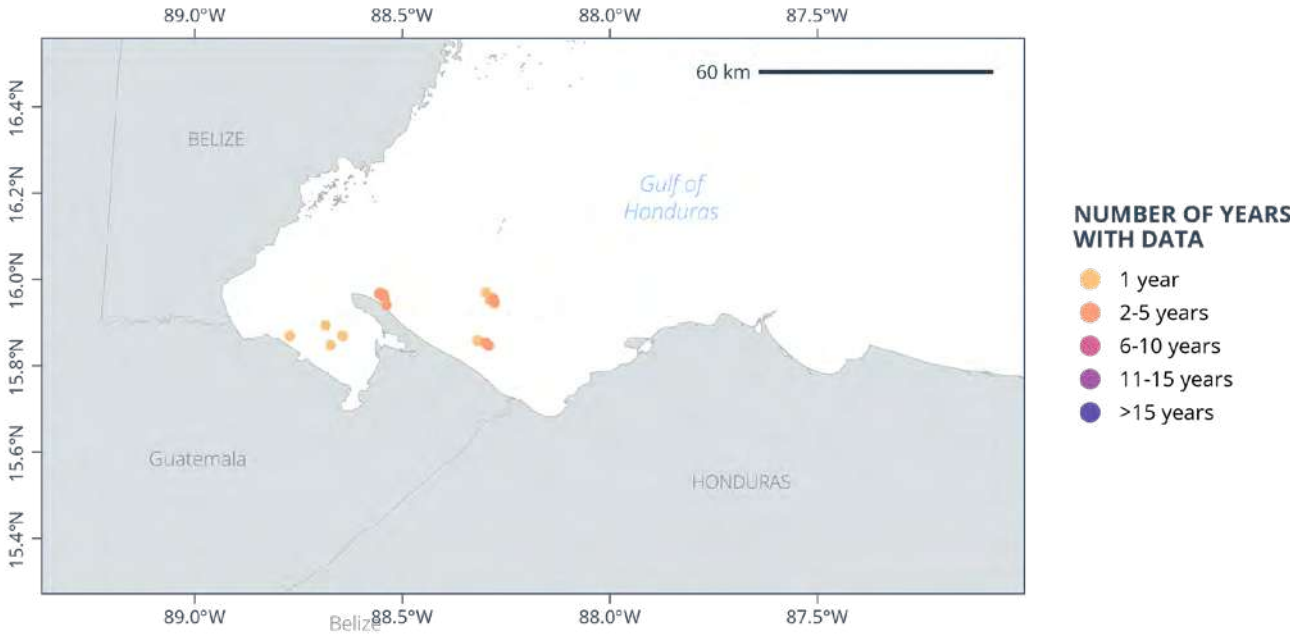
In addition to natural disturbances such as hurricanes, Guatemala's coral reefs face a range of human-induced threats that compromise their health and resilience. Poor water quality, resulting from inadequate or absent integrated watershed management, untreated wastewater, agrochemical runoff, and solid waste, introduces pathogens and other contaminants into coastal waters. These inputs not only degrade the marine environment but also promote the proliferation of macroalgae, which can overgrow and smother corals, and increase the risk of disease, reducing coral growth, recruitment, and survival. Unsustainable fishing practices near or directly on the reefs further exacerbate these pressures, while trawling in adjacent bays disrupts marine ecosystems and reduces the abundance of key species that support reef communities. Dredging activities to maintain navigational channels for ships often damage seagrass beds, which serve as critical nursery habitats for commercially important fish species that later migrate to the reefs. Finally, climate change intensifies these impacts, with elevated water temperatures causing sustained bleaching events and significant declines in coral cover over the past four years. Despite these multiple pressures, Guatemala's reefs continue to persist, highlighting their resilience and the importance of ongoing conservation and management efforts to safeguard their future.

TRENDS IN BENTHIC COVER

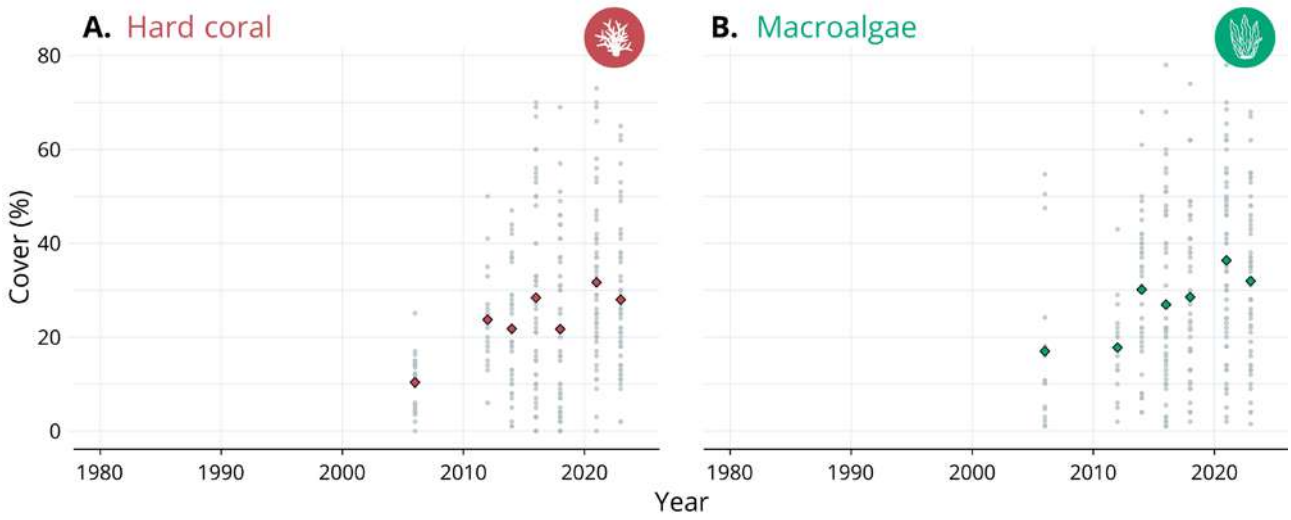
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.21.3**) encompassing 29 monitoring sites. These data were collected through 56 surveys conducted between 2006 and 2023. Trends in benthic cover vary among sites due to differences in reef type and local environmental conditions. Nearshore reefs (1–2 km from the coast), such as those on the area of Cabo Tres Puntas, which are primarily patch reefs where coral cover has generally remained lower over time. The predominant species are *Agaricia spp., Porites astreoides, Siderastrea siderea, Helioseris cucullata, Mycetophyllia spp., Colpophyllia natans and Montastrea cavernosa* (Giro, A. 2020). The benthic composition on these reefs are dominated by macroalgae, reflecting higher nutrient inputs and sedimentation from nearby rivers. In contrast, offshore spur-and-groove reefs like Motaguilla (7 km from the coast) and Cayman Crown (14 km from the coast), situated near coral reef walls, have maintained relatively higher coral cover. The corals that dominate the Cayman Crown reef area are mostly *Agaricia spp., Meandrina meandrites, Porites divaricata, Porites astreoides, Porites porites, Siderastrea siderea, Pseudodiploria strigosa, Orbicella faveolata and Colpophyllia natans* (Giro and Mojica, 2022). Despite this, offshore reefs have experienced periodic declines in coral cover due to bleaching events and localized stressors. Macroalgae cover has fluctuated across all reef types, often increasing following coral loss and potentially inhibiting recovery. Other benthic categories, including sponges and crustose coralline algae, show variable trends depending on site-specific conditions, with some increases in crustose coralline algae seen during 2024 monitoring on offshore reefs likely supporting coral recruitment (A. Giró, personal communication, 2025). Variability in reef structure and species composition drives these differences in benthic trends, emphasizing the need for site-specific monitoring to understand reef dynamics.



↑ **Figure 2.21.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.21.4.** Spatio-temporal distribution of benthic cover monitoring sites across Guatemala. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.21.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Guatemala from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Hard coral cover in Guatemala has remained relatively stable over the past four decades, generally around ~15–25%. In contrast, macroalgal cover has increased since the 1980s and now frequently exceeds ~20%. Notably, fleshy macroalgae declined from 30% in 2021 to 16% in 2023, as reported in the 2024 Mesoamerican Reef Health Report Card, likely reflecting the recovery of herbivorous fish biomass particularly parrotfish and surgeonfish, following their nationwide full protection in 2015, a measure supported by local fishers (McField et al., 2024). However, macroalgae cover still remains high, largely due to ongoing nutrient pollution from watershed discharge and agricultural runoff, which continues to fuel algal growth (McField et al., 2020).

CHAPTER 22

Haiti

Authors: Jean Wiener, Laëtitia Mathon.

Data contributors: Samuel King*, Philip Kramer*, Judith Lang*, Rita I. Sellares-Blasco*, Maria F. Villalpando*, Sandy Voegeli*, Stacey Williams*.

*AGRRA data contributor(s).

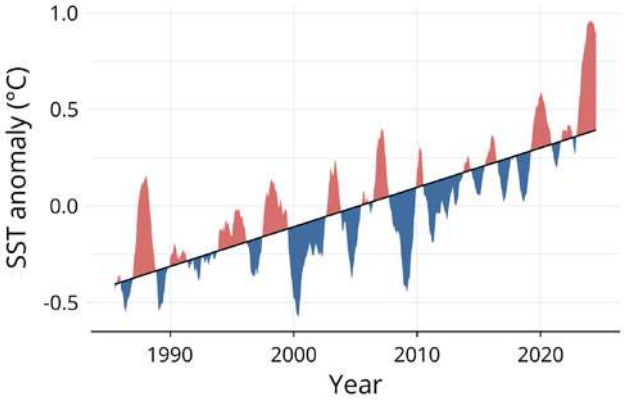
INTRODUCTION

Haiti, occupying the western third of Hispaniola, has a land area of 27,750 km², an Exclusive Economic Zone of 123,525 km², and a population that grew from 5.6 million in 1980 to 11.77 million in 2024. With 1,535 km of coastline and a narrow continental shelf of about 5,000 km², the country hosts 724 km² of coral reefs, representing nearly 3% of the Caribbean's total reef extent (Jones et al., 2004). These reefs are diverse and extensive, including fringing reefs along much of the coast, offshore barrier formations, atoll-like structures, and dramatic coral "walls" at the shelf edge. Although highly productive and ecologically significant, many remain underexplored and are increasingly overexploited by local fisheries. In recognition of the value of its coastal and marine resources, Haiti began the establishment of a network of seven MPAs beginning in 2013 (Henwood et al., 2017). The current extent of Haiti's MPA network is approximately 232,042 ha. These MPAs safeguard a unique mosaic of fringing and barrier reefs, seagrass beds, mangrove forests, and deep offshore waters. The total amount of ecosystem services provided by coral reefs and mangroves within these MPAs is over US$ 9,500,000,000/yr (Wiener, 2013).

THREATS

Thermal regime

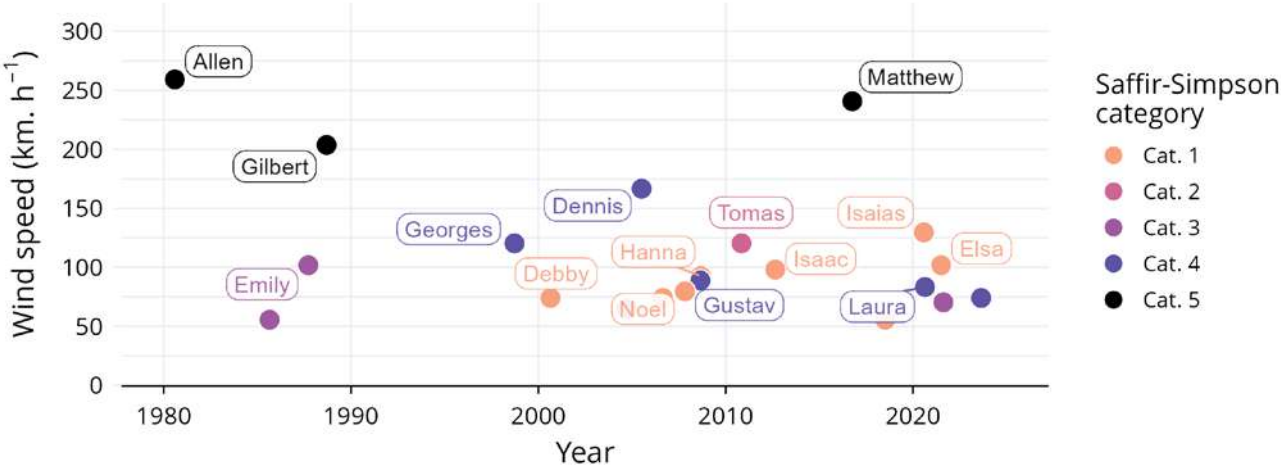
The long-term average in SST from 1985 to 2024 over coral reefs of Haiti was 28.18 °C. Over this period, SST increased by 0.92 °C, representing a warming rate of 0.024 °C per year (**Figure 2.22.1**).



↑ **Figure 2.22.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Haiti. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 32 tropical storms passed within 100 km distance from a coral reef of Haiti, and of these 9 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.22.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Allen, in 1980, which passed 21 km from a coral reef with sustained wind speed of 259 km.h^{-1}.



↑ **Figure 2.22.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Haiti. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Haiti are not represented, although they can have had an impact.

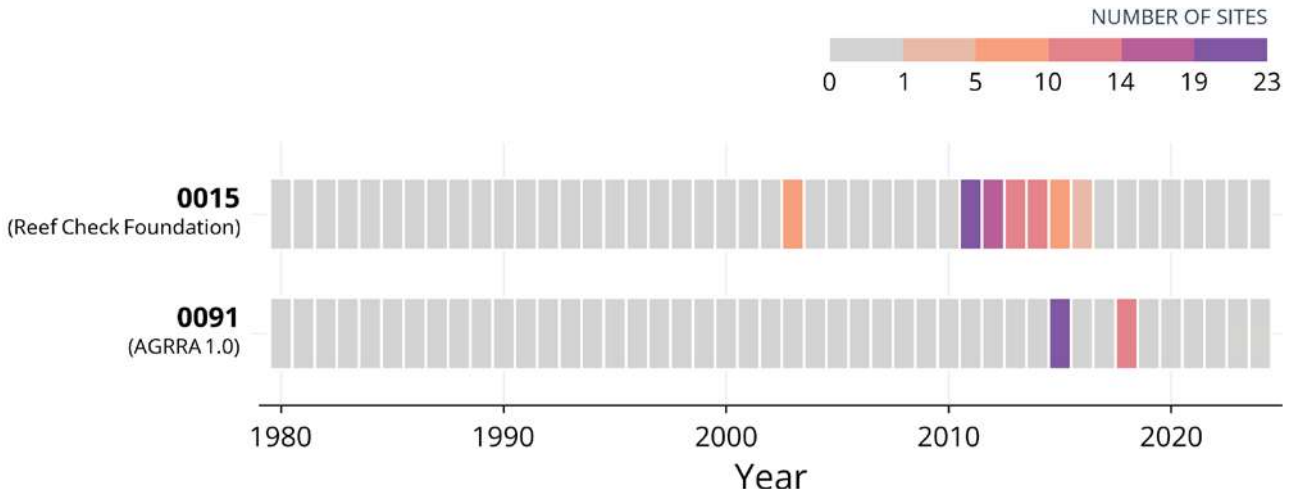
Other threats

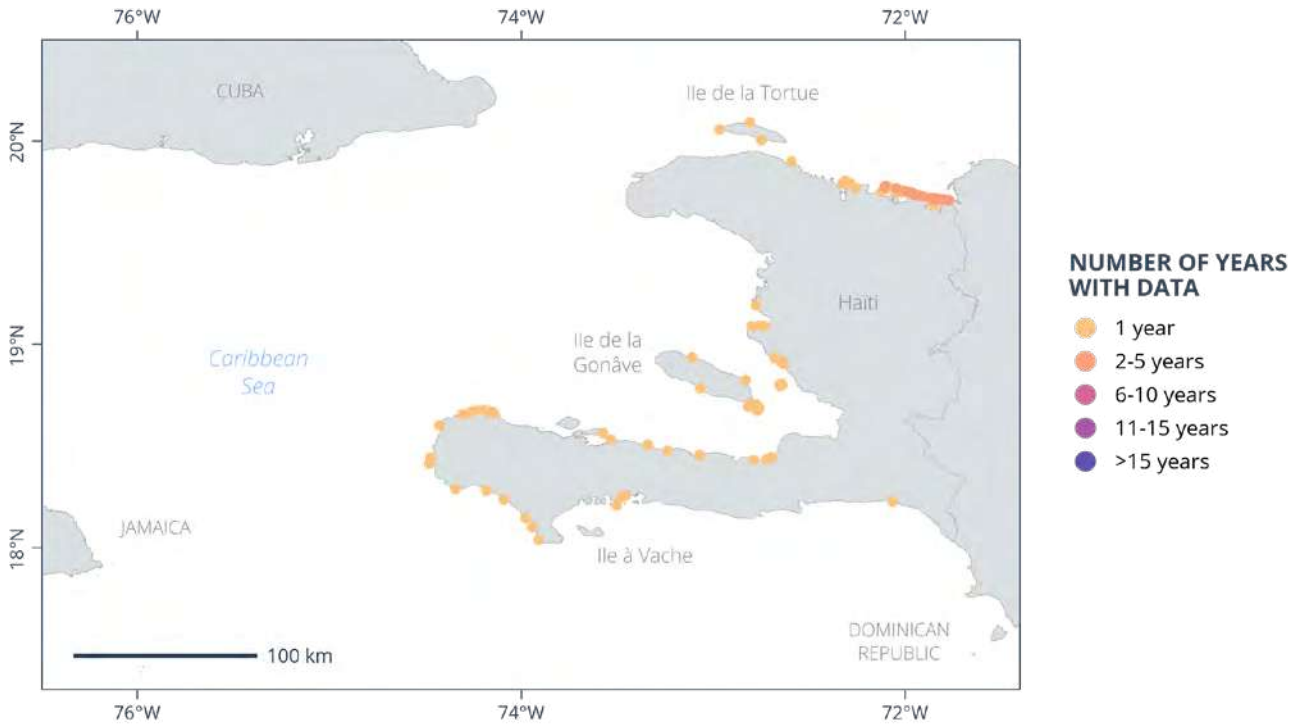
Haiti's coral reefs face multiple, overlapping threats driven by human activity and environmental stressors. Coastal resources are heavily exploited and poorly managed, with overfishing placing intense pressure on reef ecosystems (Jones et al., 2004; Lang et al., 1998). Land-based pollution further compounds these stresses: sediment washed from deforested areas during heavy rains–as seen dramatically during Hurricane Jeanne–carries fertilizers, pesticides, and other pollutants into coastal waters (Jones et al., 2004; Joseph et al., 2019). These inputs lead to sedimentation, eutrophication, and the degradation of habitats critical to reef survival, threatening vulnerable species such as *Montastraea annularis* and *M. faveolata* in Fort Liberté (Joseph et al., 2019). Nutrient enrichment also stimulates the proliferation of fleshy macroalgae, which smother corals and reduce reef resilience, particularly in areas like Port-au-Prince and Cap-Haïtien where pollution from oil, chemicals, and solid waste is especially severe (Lang et al., 1998).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.22.3**) encompassing 96 monitoring sites (**Figure 2.22.4**). These data were collected through 109 surveys conducted between 2003 and 2018.

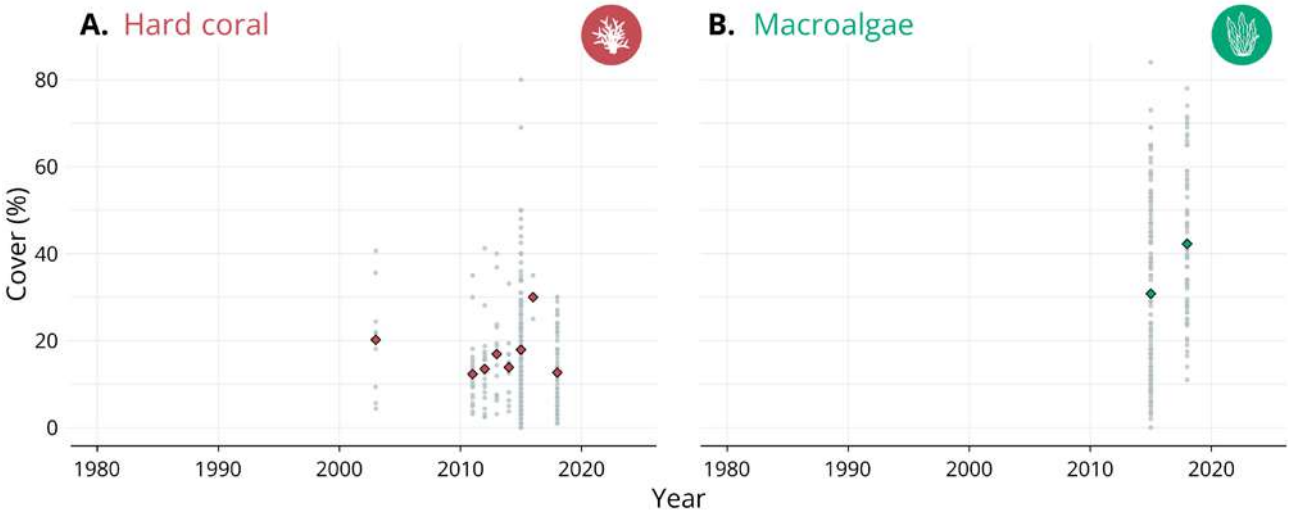
↓ **Figure 2.22.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.22.4.** Spatio-temporal distribution of benthic cover monitoring sites across Haiti. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Coral reef health in Haiti shows signs of long-term decline under mounting pressures. Assessments in the early 2000s indicated that reefs were in reasonable condition overall, with mean coral cover of about 21.8%, though values varied widely from 7% to 50% depending on the site (Jones et al., 2004). Shallow reefs tended to support higher coral cover, but algal overgrowth was already evident, and populations of the herbivorous sea urchin *Diadema* were very low. Reef fish communities were also depleted, with densities of key indicator species ranging from only 0.5 to 5 individuals per 100 m², reflecting high fishing pressure. More recent surveys from 2019–2020 documented further stress, with average coral cover across sites dropping to 16%, and some reefs exhibiting less than 5% live coral (Hodgson and Torres, 2022). Hurricanes, sedimentation, and fast-growing macroalgae contributed to local declines, while coral diseases such as Yellow Band, Black Band, White Plague, and the rapidly spreading Stony Coral Tissue Loss Disease (SCTLD) were also recorded. Together, these findings point to increasingly vulnerable reef ecosystems, characterized by low coral cover, high algal dominance, and depleted fish and urchin populations.



↑ **Figure 2.22.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Haiti from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 23

Honduras

Authors: Juan Carlos Carrasco Navas-Parejo, Alex H O'Brien, Calina Zepeda-Centeno, Belkis Carolina Montalván Torres, Claudia Alejandra Guerrero, Arlene Rodriguez, Helder Perez, Andrés B. Alegría, Damaris Elizabeth Dueñas, Daniela Mejia, Angela Maria Randazzo-Eisemann, Arlene Rodriguez, Sobeida Nazaret Nuñez, Andrea Rivera-Sosa.

Data contributors: Andrés B. Alegría*, Diego Ardon*, Edoardo Antunez*, Ana Betancourth*, John Bodden*, Max Bodmer**, Antal Borcsok*, Gisselle Brady*, Bianca Capua*, Shannon Cameron**, Lisa Carne*, Sophie Carolan**, Veronica Caviedes*, Quentin Cuassin*, Adoni Cubas*, Damaris Dueñas*, Ian Drysdale*, Christi Anne Etches*, Dan Exton**, Michelle Fernández*, Alejandro Flores*, Luis Flores*, Ana C. Fonseca*, Paola Gomez*, Ana Giro*, Erika Gress*, Paolo Guardiola*, Toby Haddon**, Jenny Hickman**, Daniel Hughes*, Andrea Izaguirre*, David Jaen*, Scott Jones*, Jaime Kawas*, Paco Lacoste*, Damaris Landers**, Judith Lang*, Natalie Lubbock**, Sara Madrid*, Oscar Maradiaga*, Melanie McField*, Daniela Mejía*, Astrid Mejía*, Amy Morrison**, Fiama Molina*, Francin Murcia*, Jennifer Myton*, Andrzej Narozanski*, Sobeida Nuñez*, Mayra Núñez*, Gabriela Ochoa*, Alex H. O'Brien**, Pamela Ortega*, Laura Palmese*, Maria Fernanda Pavón*, Helder Pérez*, Cristhian Eduardo Perez*, Diana Phillips*, Roberto Pott*, Cecilia Ramos*, Angela Randazzo*, Melissa Radley**, John Reed*, Charlie Riddock**, Chris Riley**, Andrea Rivera-Sosa*, Arlene Rodriguez*, Alicia Rodríguez*, Andrew Ross*, Hannah Rushton**, Ella Tabares*, Armeid Thompson*, Alejandra Thompson*, Oscar Torres*, Ruben Torres*, Valeria Valladares*, Francin Varela*, Rebecca Wardle*, Trent Williams*, Ross Williamson**, Calina Zepeda*, Emma Ziska*, Zara Zuniga*..

*AGRRA data contributor(s).

**Operation Wallacea data contributor(s)

INTRODUCTION

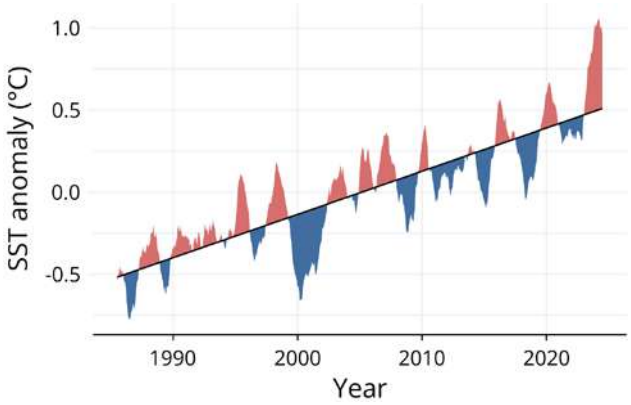
Honduras, a mountainous nation with a vast marine territory (229,501.96 km²) nearly twice its land area (112,492 km²) lies within the Western Atlantic region (Caviedes et al., 2014) and shares boundaries with seven countries through its Caribbean waters and land borders. The country hosts approximately 1,060 km² of coral reefs, representing about 4.4% of the total reef area in the Caribbean. These reefs are primarily found around the Bay Islands, the Mosquitia coast, and the Swan Islands, include fringing, barrier, patch, and aggregate formations that mark the southern extent of the Mesoamerican Reef (MAR). Many northern reefs are underrepresented in satellite imagery due to high turbidity caused by terrestrial runoff from watershed basins, which complicates remote sensing (Purkis et al., 2016; Berger et al.,.2022). These ecosystems support over 60 reef-building coral species, including the critically endangered *Acropora* spp., and sustain more than 500 species of reef fish. The Bay Islands National Marine Park and the Cayos Cochinos Archipelago Natural Monument are among the country's largest protected areas (Rivera et al., 2024), these encompass key coral reef habitats and contribute significantly to marine conservation efforts (Purkis et al., 2015; 2016).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Honduras was 28.05 °C. Over this period, SST increased by 1.14 °C, representing a warming rate of 0.029 °C per year (**Figure 2.23.1**). In Honduras, the first basin-wide severe coral heat-stress event occurred in 1998. The 2005 bleaching impact around Roatán was moderate compared to other Caribbean sites (Rowlands, Purkis and Riegl, 2008: Eakin et al., 2010). During the 2015–2017 marine heatwave, survey data across the MAR recorded over 25% of reef sites in Honduras with more than 15% of colonies fully bleached (Eakin

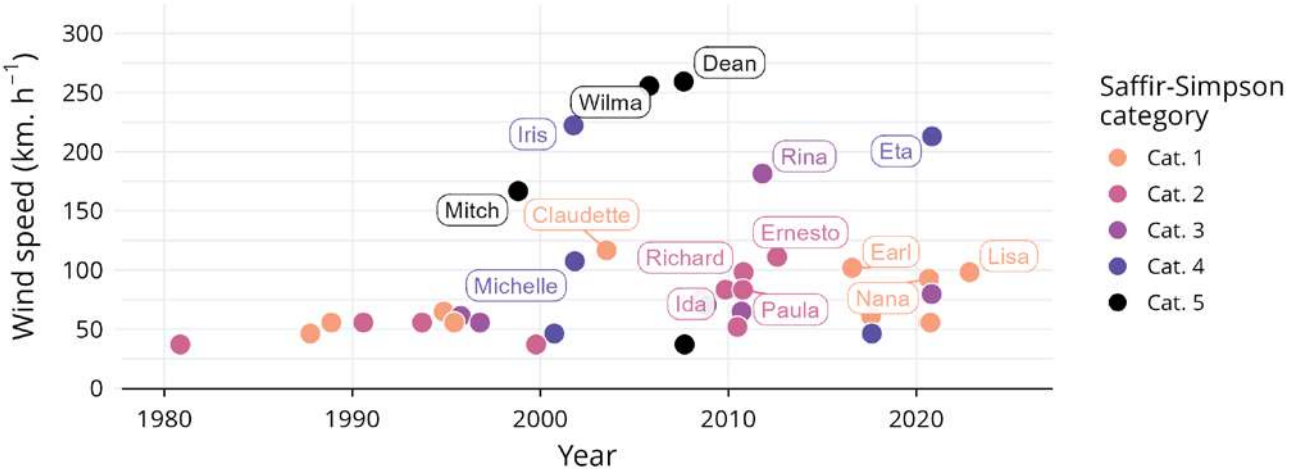
et al., 2019; Muñiz-Castillo et al., 2024). In 2023, an extreme El Niño-driven warming event (with accumulated heat stress ranging from 16–24°C weeks) triggered widespread coral bleaching in Honduras, with documented mass mortality of corals such as *Acropora cervicornis* and *Agaricia tenuifolia* in highly dense areas, such as Cordelia Banks in Roatan, and Tela Bay in the north coast of Honduras respectively.



↑ **Figure 2.23.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Honduras. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 58 tropical storms passed within 100 km of coral reefs in Honduras. Of these, 10 storms were characterized by sustained wind speeds exceeding 100 km.h^{-1} (**Figure 2.23.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Dean in 2007, which passed 71 km from a coral reef with sustained winds reaching 259 km.h^{-1}.



↑ **Figure 2.23.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef in Honduras between 1980 and 2024. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the wind speed values presented correspond specifically to the storm's closest recorded position to a coral reef. As a result, some sustained wind speed values fall below the minimum threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing beyond 100 km from Honduran coral reefs are not included, although they may still have exerted significant impacts.

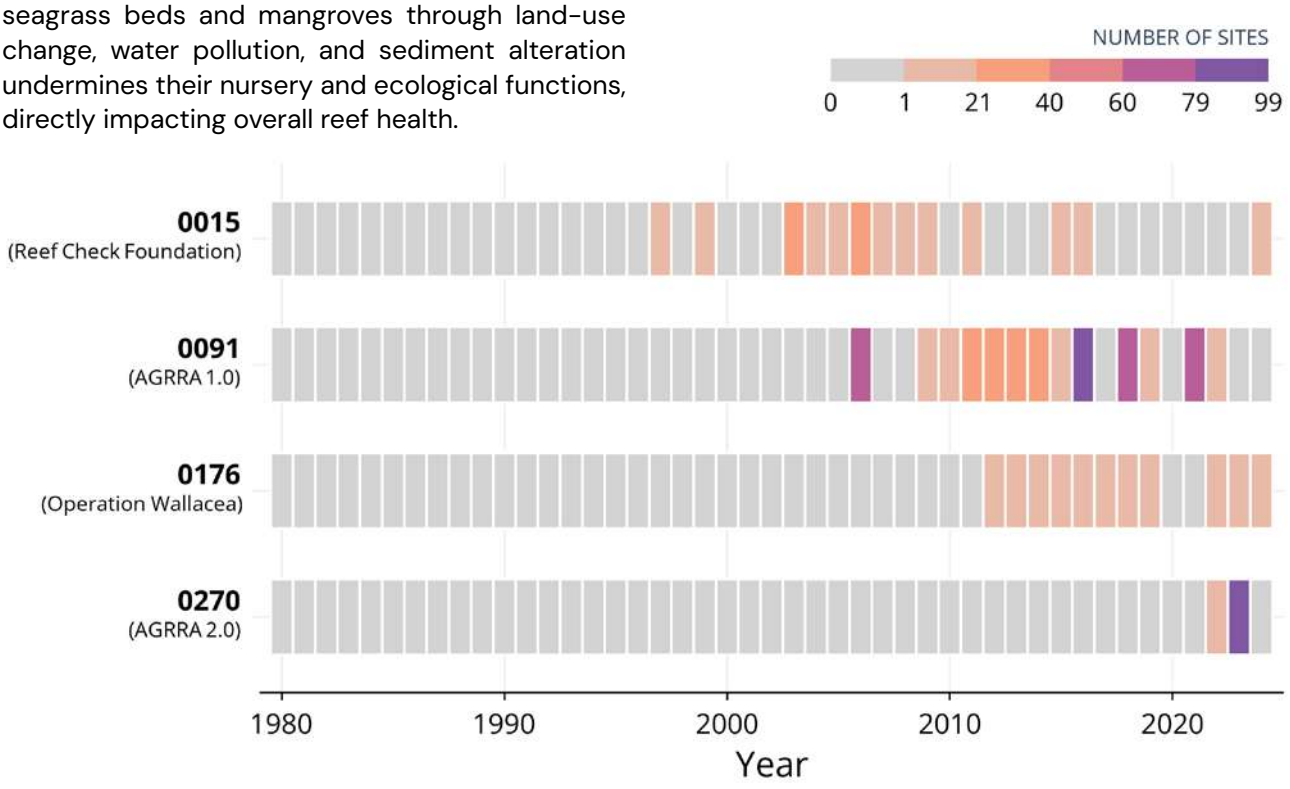
Other threats

Coral reefs in Honduras are under severe stress due to a combination of nutrient pollution, disease, overfishing, macroalgal overgrowth, and plastic debris. Land-based pollution and eutrophication from agriculture, livestock, and deforestation along northern Honduras and the Bay Islands accelerate reef degradation, with runoff from over 190,000 ha of oil-palm plantations and a >2,000% increase in fertilizer use (Carrasco Navas-Parejo *in press*). Catchments shared by Honduras and Guatemala contribute more than 50% of the nitrogen and phosphorus entering the Mesoamerican Reef, aggravated by untreated municipal and industrial effluents, port dredging, and maritime discharges (Berger, et al., 2022). Unsustainable fishing follows a pattern of overexploitation linked to export-driven markets, the use of destructive gears, and fishing down the food web, while incomplete data reporting hinders science-based management (Box and Canty, 2010; Funes et al., 2015; Salas et al., 2007). Plastic pollution from rivers such as the Motagua, Ulúa, Cangrejal, Tinto, and Aguán flows into the Caribbean, especially during the rainy season (Kikaki et al., 2020). In Tela Bay, high

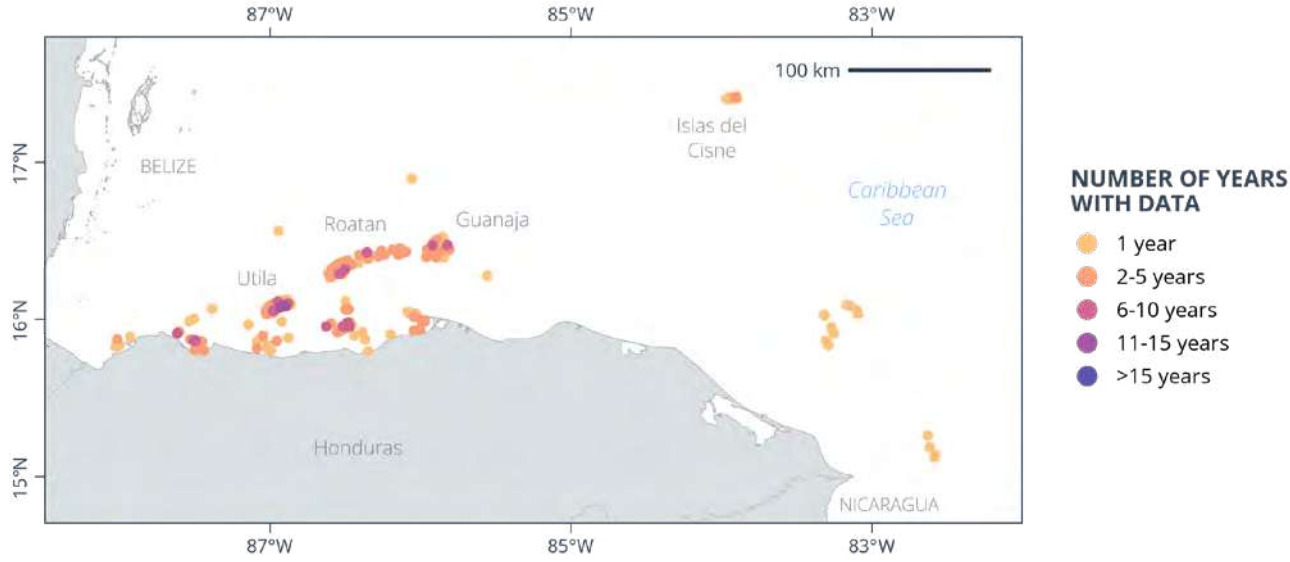
microplastic concentrations originated mainly from Lancetilla River, the Micos Lagoon and the Ulúa River (Carrasco Navas-Parejo et al., 2024). Coral disease, particularly Stony Coral Tissue Loss Disease, has devastated Honduran reefs since 2020, spreading rapidly at rates of up to 155 m/day (Truc et al., 2023). Moreover, the degradation of highly connected ecosystems such as seagrass beds and mangroves through land-use change, water pollution, and sediment alteration undermines their nursery and ecological functions, directly impacting overall reef health.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 4 datasets (**Figure 2.23.3**) encompassing 405 monitoring sites. These data were collected through 848 surveys conducted between 1997 and 2024.

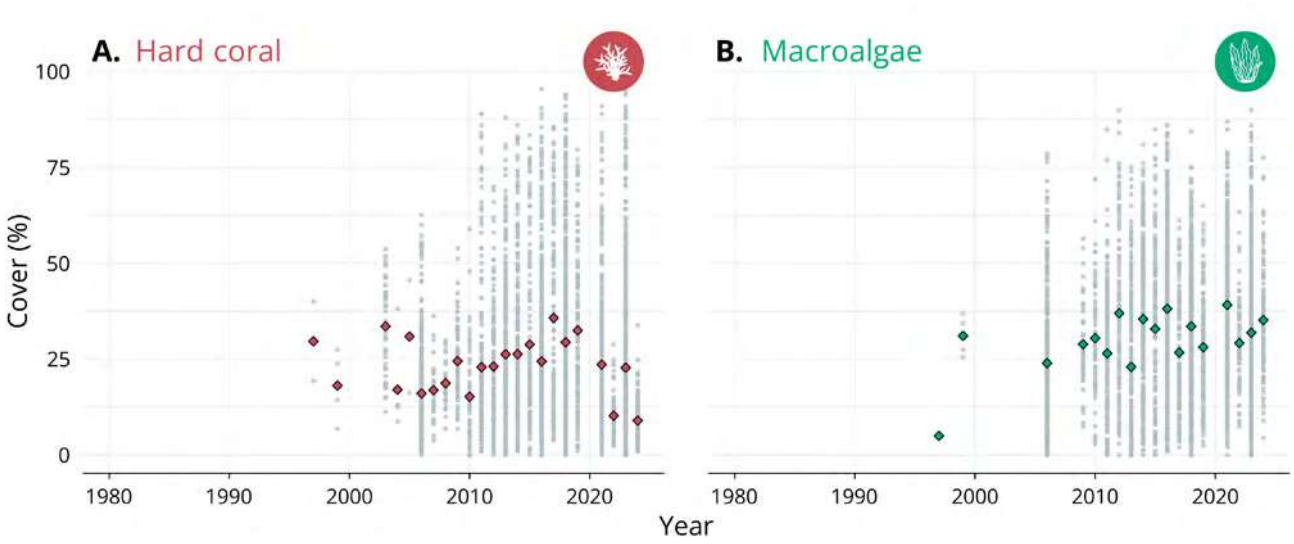


↑ **Figure 2.23.3.** Temporal distribution of surveys from the datasets used to model trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, while the text in parentheses below each number corresponds to the dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.23.4.** Spatio-temporal distribution of benthic cover monitoring sites across Honduras. Sites monitored over a greater number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Since the 1980s, coral reefs along the Honduran coast have shifted from coral-dominated toward macroalgal-dominated assemblages. Early monitoring efforts in the Bay Islands and northern Honduras documented live hard coral cover of ~28% in the early 1990s and relatively low macroalgae (Kramer and Kramer, 2002). However, by the 2000s and 2010s, studies revealed persistent declines in live coral cover and increases in fleshy macro-algae. Although herbivorous fish biomass in Honduras is high, macro-algal cover still remains high indicating poor reef health in the region (McField et al., 2024). A recent study found that between 2012 and 2019, algae and soft coral cover increased, while hard coral cover declined across monitored sites in Honduras (Petit and Allsopp, 2023). Skeletal extension and density of major reef-building corals such as *Siderastrea siderea* are reduced in Honduran-influenced forereefs, correlated with elevated Ba/Ca ratios (proxy for sediment/river discharge) implicating land-based sediment/nutrient runoff (Shaw et al. 2024). In areas like Tela Bay, sheltered reefs initially recorded hard coral cover up to 62% and macroalgal cover ~7% (Bodmer et al., 2015); yet recent marine heatwaves triggered coral bleaching and mortality events, further accelerating the shift toward macroalgal dominance. This transition to macroalgal dominance is consistent with Caribbean-wide patterns where coral decline and macroalgae rise are driven by reduced grazing, nutrient enrichment, and climate related stressors (Gove et al., 2023) and suggests the need for integrated management strategies that address watershed control, herbivore protection, and thermal-stress mitigation.



↑ **Figure 2.23.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Honduras from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 24

Jamaica

Authors: Monique Curtis, Patrice Francis, Jérémy Wicquart.

Data contributors: Ernesto Arias, Brian Beck*, Courtney Black*, Kayla Blake*, Macarena Blanco, Nikki Bramwell*, Johanna Calle-Triviño, Kimberley Coore*, Monique Curtis, Rachel D'silva*, Kristi D. Klomp*, Alexandra Dempsey*, Yannique Ewers*, Beth F. Moses*, Tatum Fisher*, Patrice Francis, Victor Galván, Peter Gayle*, Sean Green*, Hector Guzman*, Loúreene Jones*, Ivana Kenny*, Kimani Kitson-Walters*, Philip Kramer*, Judith Lang*, Andrea Lanigan*, Simone Lee*, Michelle Mcnaught*, Achsah Mitchell*, Tamarah Myles, Suzanne Palmer*, Chanel Raynor, Andrew Ross*, Marjo Vierros*, Sandy Voegeli*, Andrew W. Bruckner*, George Warner*, Nathalie Zenny*.

*AGRRA data contributor(s).

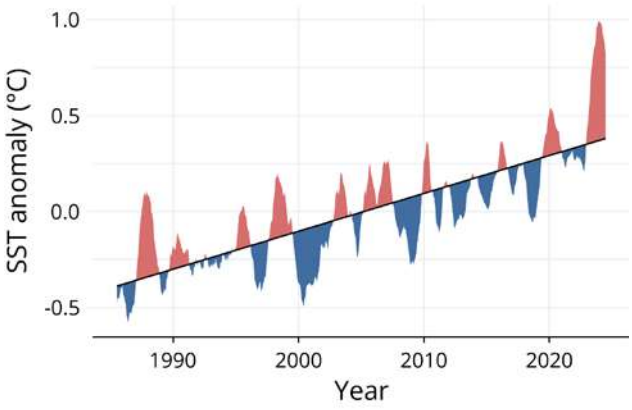
INTRODUCTION

Jamaica, an island nation in the Greater Antilles, is located south of Cuba and west of Haiti. It is the third-largest island in the Caribbean, stretching about 235 km in length and 82 km in width. Coral reef extent is about 734 km², which represents 3% of the Caribbean's total coral reef extent. Fringing reefs occur mostly along the north and east coasts with patchy fringing and barrier reefs on the south coast. There are also corals and reefs along the Formigas and Pedro Banks as well as Morant Cays.

THREATS

Thermal regime

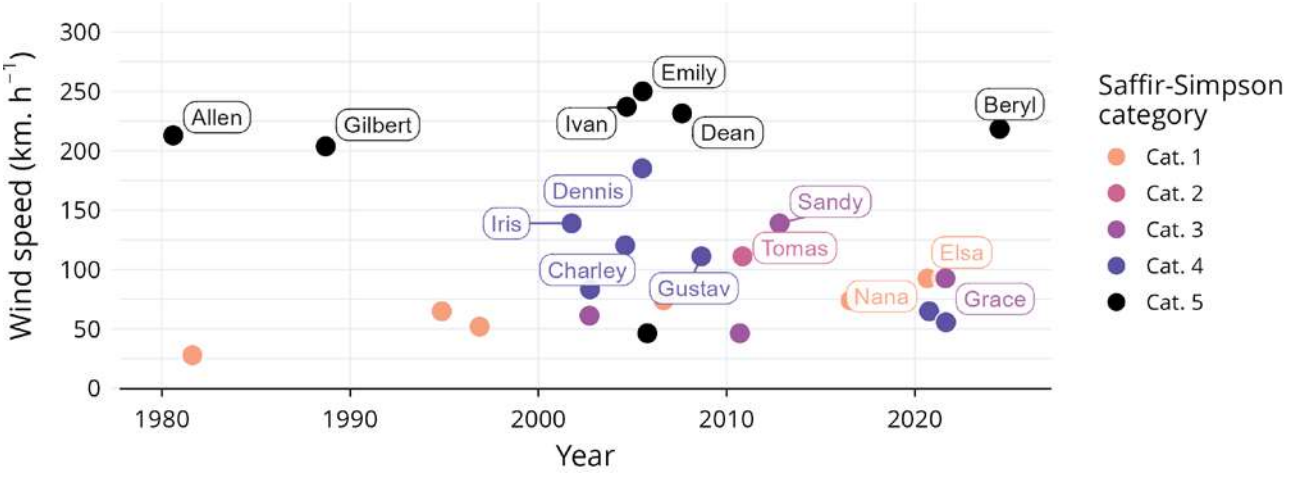
The long-term average in Sea Surface Temperature (SST) from 1985 to 2024 over coral reefs of Jamaica was 28.18 °C. Over this period, SST increased by 0.89 °C, representing a warming rate of 0.023 °C per year (**Figure 2.24.1**).



↑ **Figure 2.24.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Jamaica. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 32 tropical storms passed within 100 km distance from a coral reef of Jamaica, and of these 12 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.24.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Emily, in 2005, which passed 40 km from a coral reef with sustained wind speed of 250 km.h^{-1}.



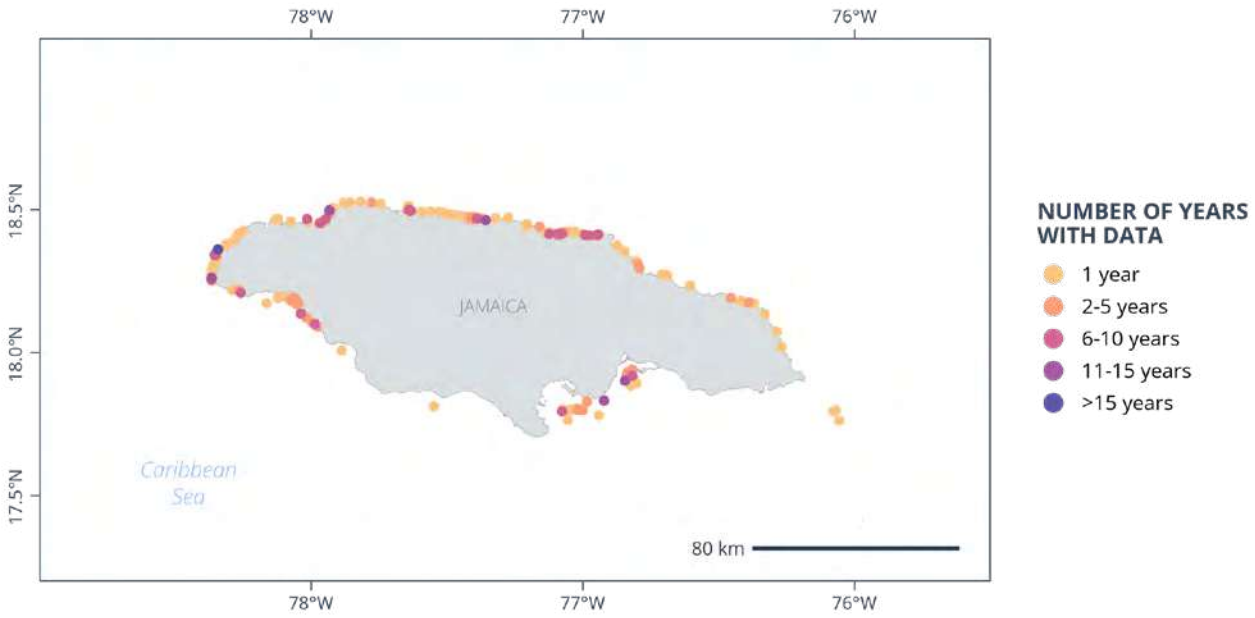
↑ **Figure 2.24.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Jamaica. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Jamaica are not represented, although they can have had an impact.

Other threats

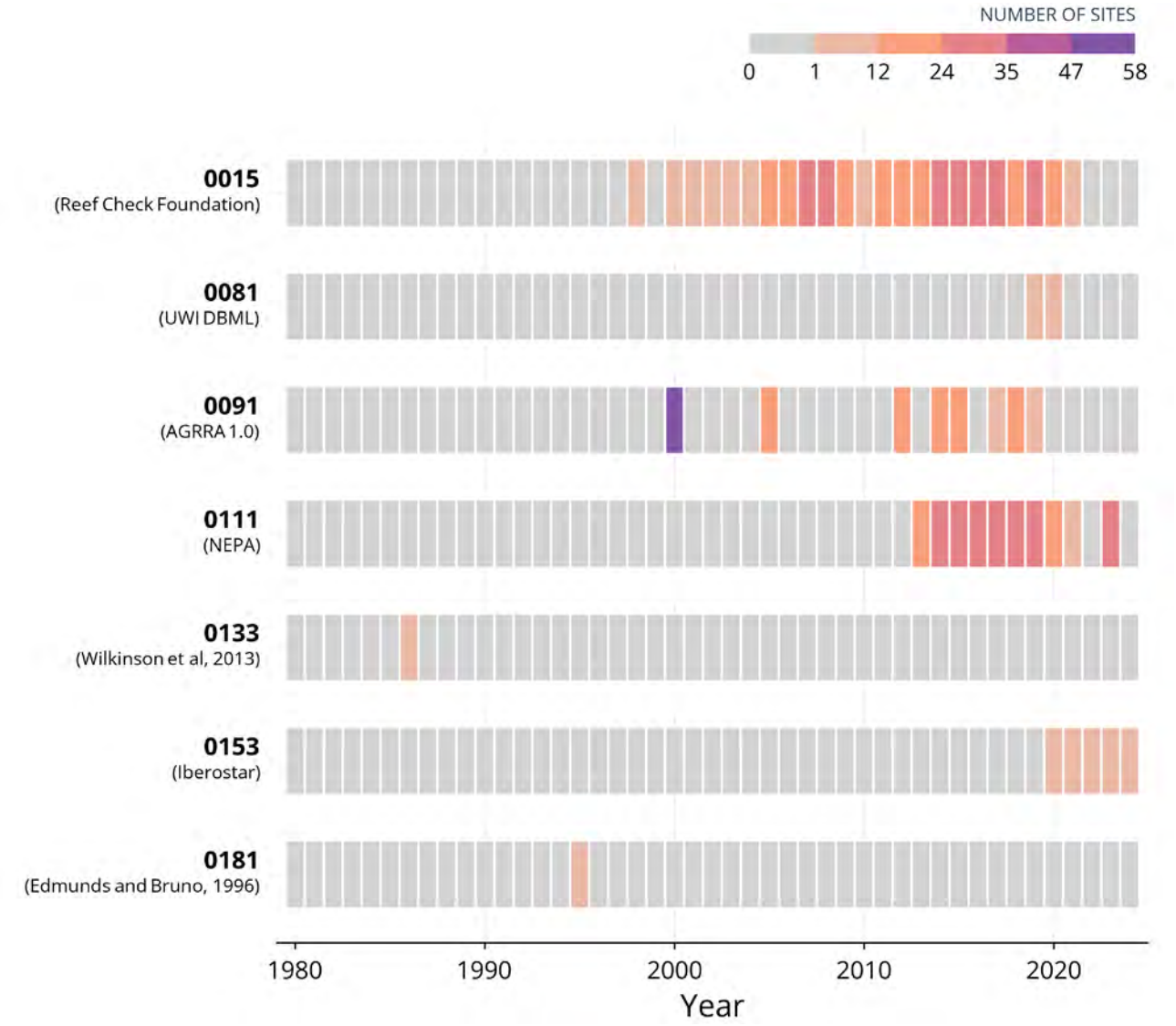
Other natural threats to Jamaica's coral reef ecosystem include coral diseases such white-band disease in the 1980's or Stony Coral Tissue Loss Disease in 2018; along with mass mortality of *Diadema antillarum* in the 1980's and 2022. These natural threats combined with anthropogenic impacts such as overfishing, increased nutrient pollution and coastal development resulted in shift from a coral- dominated to a macro-algal dominated reef (Hughes, 1994; Crabbe 2010).

TRENDS IN BENTHIC COVER

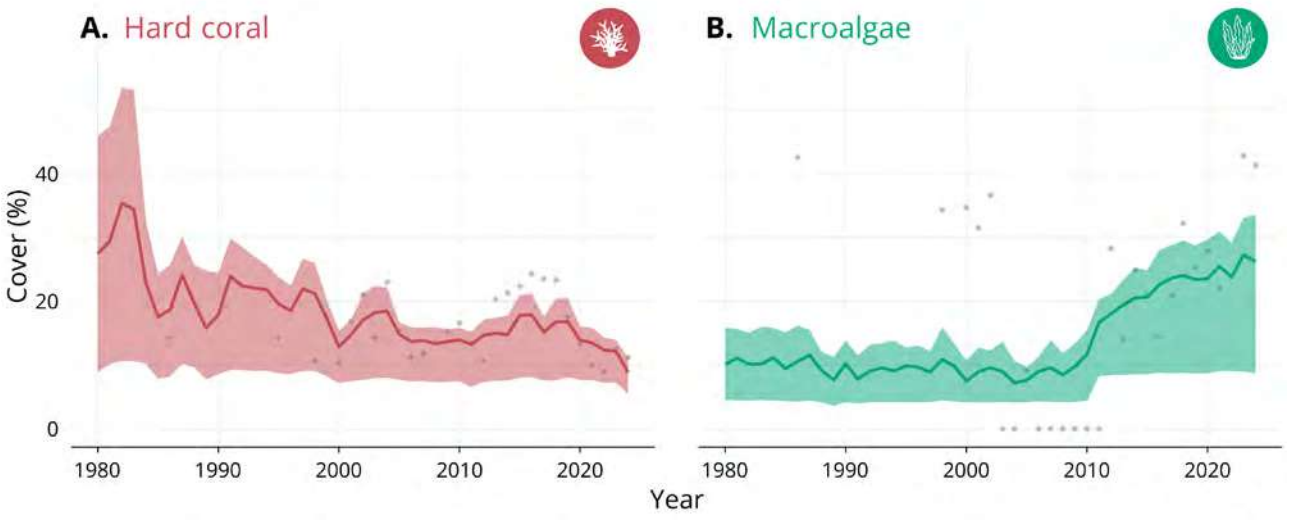
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 7 datasets (**Figure 2.24.3**) encompassing 308 monitoring sites. These data were collected through 794 surveys conducted between 1986 and 2024.



↑ **Figure 2.24.4.** Spatio-temporal distribution of benthic cover monitoring sites across Jamaica. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.24.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.24.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Jamaica from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data

CHAPTER 25

Martinique

Authors: Jean-Philippe Maréchal, Mélanie Bon, Catherine Desrosiers, Claude Bouchon, Yolande Bouchon-Navaro, Jérémy Wicquart.

Data contributors: Mélanie Bon, Claude Bouchon, Yolande Bouchon-Navaro, Sophie Brugneaux, Gilles Delouche, Priscilla Dupont, Laurent Juhel, Hélène Legrand, Max Louis, Josianne Mahieu, Jean-Philippe Maréchal, Cécile Pérez, Pedro Portillo, Bernard Renaudie, Yann Rousseau, Claude Séraline, Victor Tersiguel, Ewan Trégarot.

INTRODUCTION

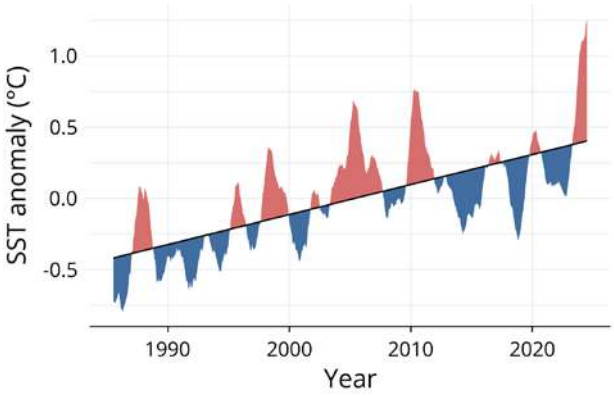
Martinique, a French overseas territory in the Lesser Antilles, lies between Dominica to the north and Saint Lucia to the south. Since 2017, a marine national park covers its entire territorial sea and exclusive economic zone of 48,900 km². Coral reef extent is about 155 km², which represents 0.6% of the Caribbean's total coral reef extent. Using a more precise data source than the one used for this report, another study estimated the total Martinique shallow coral reef extent to be 56 km² (IFRECOR, 2021). These reefs are mainly concentrated along the island's Atlantic and Caribbean southern coasts, as well as at the entrance of Fort-de-France Bay (IFRECOR, 2021). They host 57 species of Scleractinia corals, four species of *Millepora* and one *Stylaster* (Madibenthos, 2016). Sixteen of these species are locally fully protected.

In 2022, the island was home to around 361,000 inhabitants, with a population density of 320 inhabitants per km² (INSEE, 2022). Coral reefs, seagrass meadows, and mangroves contribute around 169 million euros annually to Martinique local economy including coastal protection, reef fisheries, carbon sequestration, tourism, and leisure activities (IFRECOR, 2021).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Martinique was 27.71 °C. Over this period, SST increased by 0.98 °C, representing a warming rate of 0.025 °C per year (**Figure 2.25.1**).



↑ **Figure 2.25.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Martinique. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

The 2005 marine heatwave caused a marked ecological impact, with an estimated 11% decline in coral cover in Martinique (Wilkinson and Souter, 2008). In the French Caribbean, earlier warm episodes in 1984 and 1987, both associated with El Niño, were recorded but produced only limited damage. By contrast, the 1998 event affected 59% of coral colonies in Martinique, with post-bleaching mortality ranging from 20 to 30% depending on the site (Bouchon and Bouchon-Navaro,

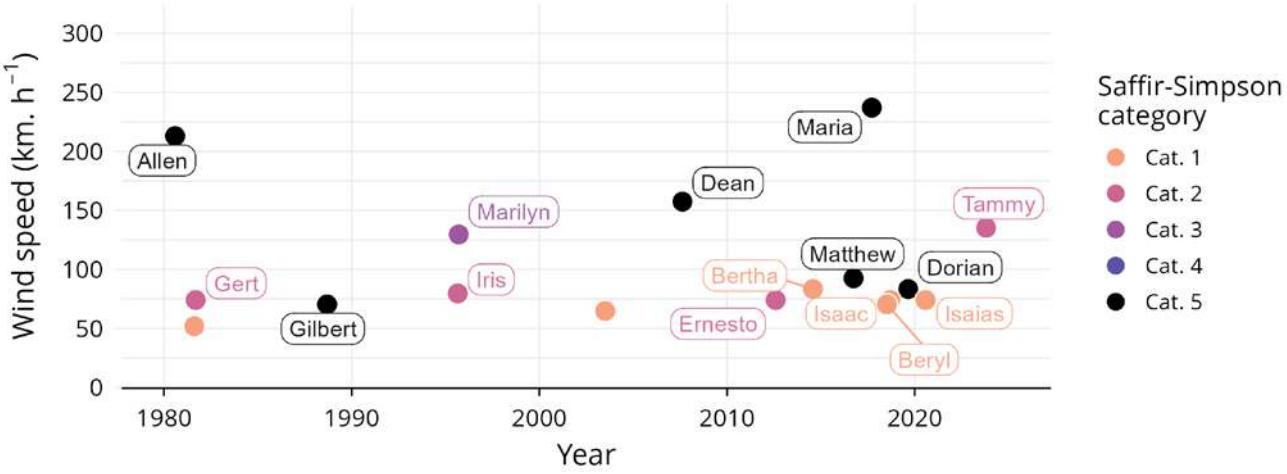
1998). The next major thermal anomaly, in 2005, affected most of the Lesser and Greater Antilles. In Martinique, around 80% of colonies bleached, and delayed mortality recorded in 2006 led to a 27–46% reduction in coral cover, again with strong spatial variability (Bouchon et al., 2008).

An unprecedented situation was documented in 2023. Coastal waters around Martinique reached ~31°C in October 2023, and the thermal stress recorded that year exceeded all previous observations (Bon et al., 2025). This episode caused extensive bleaching, with about 85–86% of colonies affected, and was followed by substantial post-bleaching mortality that translated, at monitored sites, into an overall ~35% decline in live coral cover between October 2023 and February 2025. Mortality was particularly high in thermally sensitive taxa, notably *Agaricia spp.*, *Madracis spp.*, *Acropora spp.*, as well as *Helioseris*, *Mycetophyllia* and *Pseudodiploria* (Impact Mer, 2023; Bon et al., 2025). Coral colonies that survived the 2023 episode were hit again by unusually warm conditions in 2024, limiting recovery. The loss of coral framework rapidly opened free substrate, which was promptly colonized by fleshy algal assemblages, accelerating the shift towards algal-dominated patches.

Hurricanes

Between 1980 and 2024, a total of 28 tropical storms passed within 100 km distance from a coral reef of Martinique, and of these 7 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.25.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Maria, in 2017, which passed 38 km from a coral reef with sustained wind speed of 237 km.h^{-1}.

However, the storm that left the most pronounced ecological signature in the monitoring record was hurricane Dean in August 2007. Surveys conducted in 2008 revealed characteristic mechanical damage on exposed leeward sites, with branching *Madracis mirabilis* populations declining by nearly 75% at some locations, while massive *Montastrea* spp. remained largely unaffected. At other sites, live coral cover dropped abruptly by about 28% between late 2007 and early 2008, creating newly exposed substrate that was rapidly colonized, first by turf algae, then by encrusting coralline algae and macroalgae, indicating a short-term shift in benthic community structure.



↑ **Figure 2.25.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Martinique. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Martinique are not represented, although they can have had an impact.

Coral diseases

Coral diseases have repeatedly amplified the effects of thermal and mechanical disturbances on Martinique's reefs. In the aftermath of the 2005 mass bleaching, the 2006 IFRECOR survey reported an "anomalous" development of tissue necroses and bacterial-like syndromes on weakened colonies, concentrated on the southern Caribbean sites, while northern sites remained almost unaffected. This post-bleaching epizootic added about 15% recent mortality to the 14%

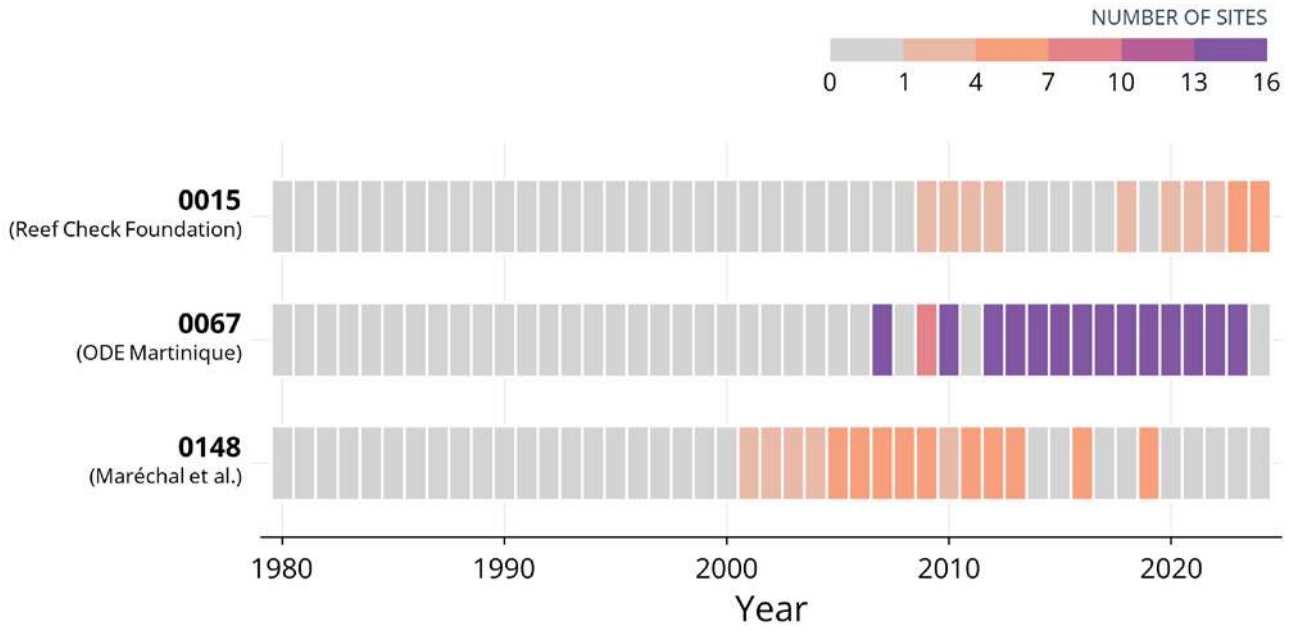
losses directly attributed to bleaching, so that local declines in live coral reached about 30%; at the same time, coral recruitment collapsed, with 3–4 times fewer juveniles in 2006 than in 2005, signaling an immediate constraint on recovery.

A second, more insidious phase of disease emergence was documented 15 years later, marked by the arrival of Stony Coral Tissue Loss Disease (SCTLD) along the Caribbean coast. Between March 2021 and January 2022, ten monitored sites were affected: coral mortality more than doubled, rising from 4% to 9.5% of colonies, while disease prevalence increased from 2.8% to 3.3%, indicating that the outbreak was ongoing. The 2022 survey also revealed a rapid shift in disease impact from initially sensitive taxa to dominant, reef-structuring species. *Porites astreoides*, which represented 50% of the assemblage, saw infection rates increase from 2% in 2021 to 18.5% in early 2022. Other key framework species, including *Siderastrea siderea*, *Orbicella* spp., and *Montastraea cavernosa*, were also affected, while rare or protected species such as *Dendrogyra cylindrus*, *Mussa angulosa*, *Colpophyllia natans*, and *Pseudodiploria clivosa* had already been functionally extirpated from transects. The highest prevalence, reaching 9%, was recorded at several southern sites, confirming that SCTLD can rapidly erode the remaining pool of slow-growing, reef-building species. In some locations, low SCTLD levels coincided with high rates of other necroses, likely linked to local environmental stressors (Créocéan, 2022).

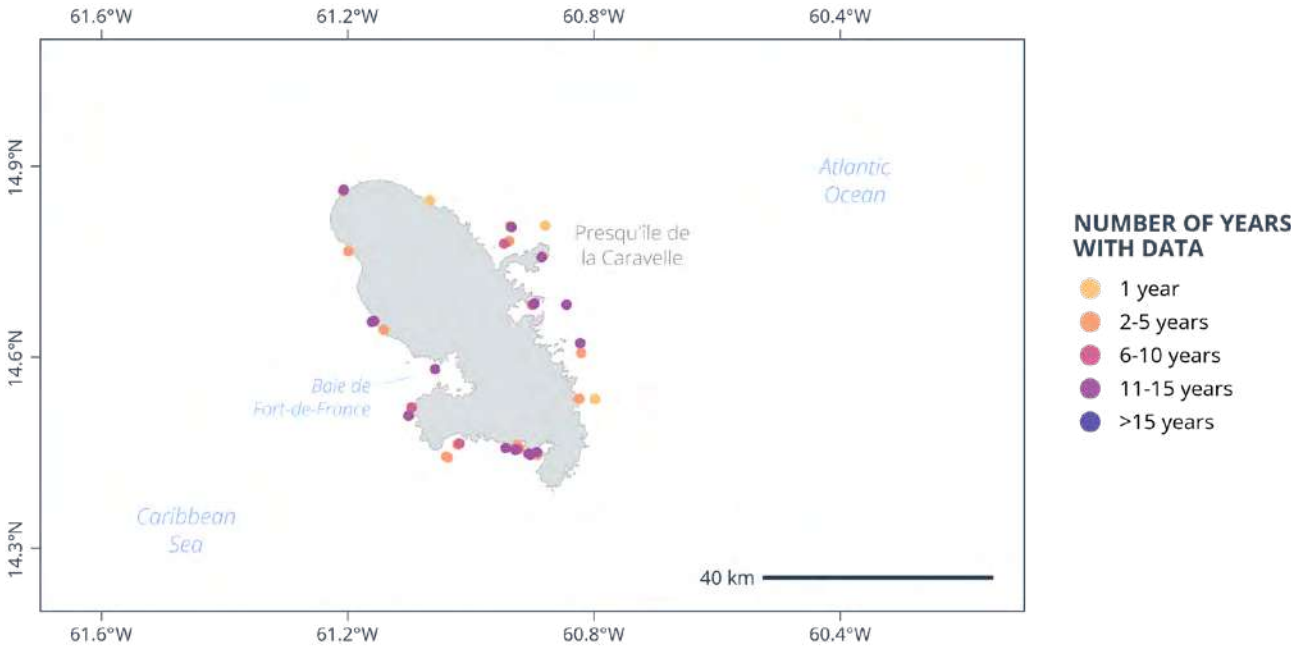
In early 2022, Martinique was affected by the regional mass mortality event of the long-spined sea urchin *Diadema antillarum*, which began in the northern and eastern Caribbean in February and spread southward within weeks. Local alerts issued by DEAL/IFRECOR in April 2022 reported rapid die-offs on several fringing and reef sites, showing the characteristic symptoms observed elsewhere in the region. Once affected, sites often experienced near-total mortality within just a few days. This episode was part of the wider 2022 Caribbean outbreak documented by the AGRRA *Diadema* Response Network and later attributed to a scuticociliate pathogen, which caused 60–90% population losses on many islands. In Martinique, the 2024 GCRMN survey update identified the 2022 *Diadema* collapse as a major event that weakened herbivory just before the 2023 coral bleaching crisis, thereby increasing the risk of algal overgrowth on newly exposed reef surfaces.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.25.3**) encompassing 41 monitoring sites. These data were collected through 322 surveys conducted between 2001 and 2024.



↑ **Figure 2.25.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

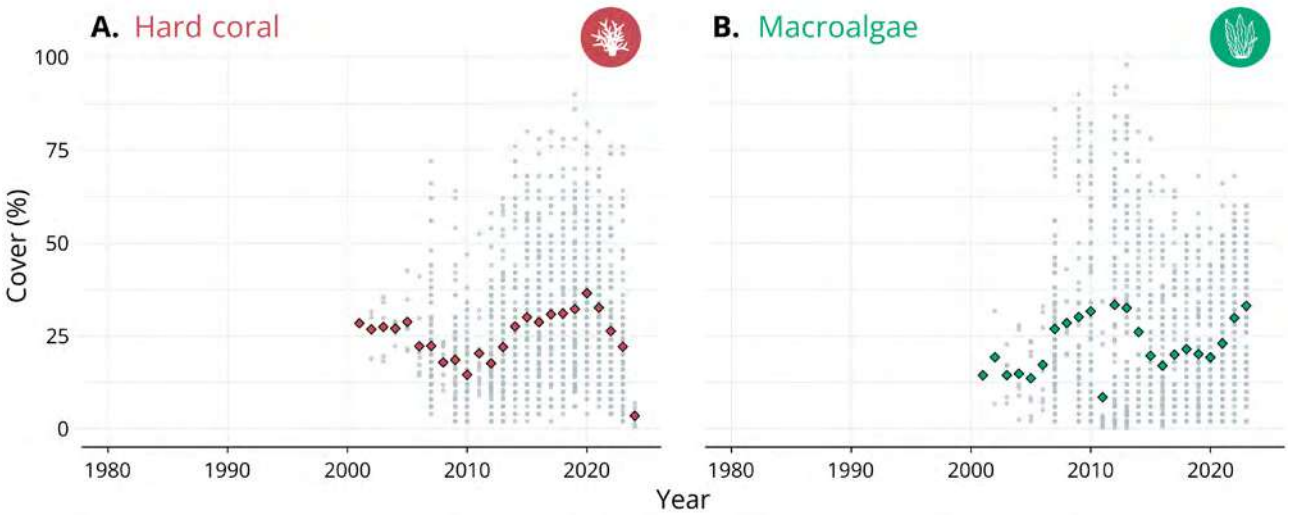


↑ **Figure 2.25.4.** Spatio-temporal distribution of benthic cover monitoring sites across Martinique. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

The first quantitative descriptions of Martinique's coral communities, produced during the CORANTILLES mission in the mid-1980s and later synthesized by Bouchon and Laborel (1986), already revealed a marked contrast between the exposed outer slope at Pointe-Borgnèse, with approximately 40% live coral cover, and the more sediment-influenced inner-bay sites such as Banc Crique, where live coral cover ranged between 15% and 20%. When the GCRMN monitoring network was established in 2001, this spatial pattern was confirmed. At Pointe-Borgnèse, live coral cover generally remained between 30% and 39% until the 2005 mass-bleaching event, whereas at Fond-Boucher it was consistently lower, fluctuating between 18% and 25% depending on the year. Over the same period, macroalgae typically accounted for 20–34% of the benthic cover at Pointe-Borgnèse, while at Fond-Boucher it rarely exceeded 10%, with turf algae being the main benthic competitor.

The island-wide time series (**Figure 2.25.3**) reveals a similar trajectory at a broader scale. Mean hard coral cover was approximately 27–30% between 2001 and 2004, before dropping sharply following the 2005 thermal stress event and the passage of Hurricane Dean in 2007 (Maréchal et al. 2006; Bouchon et al. 2008). From around 2010 onward, most sites showed slow and uneven recovery, with island-wide means rising again to 28–35% between 2015 and 2021 (**Figure 2.25.5**). However, variability among sites remained high, with coral cover values ranging from less than 5% to over 70%, reflecting local differences in exposure, sedimentation, and management practices. This recovery trend was abruptly reversed by the 2023 extreme heat wave event. The 2024 data show a collapse of mean live coral cover to only a few percent, consistent with the 35% loss recorded at permanent stations following the 2023–early 2024 bleaching and mortality sequence (Impact Mer 2023; Bon et al., 2025).

Macroalgae cover exhibited the expected inverse relationship with hard coral cover (**Figure 2.25.5**). In the early 2000s, macroalgae cover was approximately 15%. Between 2007 and 2013, macroalgae cover increased to 25–30%, occupying the space left by the decline in hard coral cover following hurricane Dean and the 2005 bleaching event. During the subsequent phase of coral recovery in the mid-2010s, macroalgae cover declined and then stabilised at 15–20%. From 2020 onwards, macroalgae cover increased again as disturbances led to further reductions in hard coral cover (Maréchal and Trégarot, 2019, 2025; Créocéan, 2022). More recent estimates from 2024 indicate macroalgae cover of around 44% across five monitoring sites (not represented in **Figure 2.25.5**; Maréchal, *pers. comm.*).



↑ **Figure 2.25.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Martinique from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 26

Mexico (Caribbean Sea)

Authors: Ana Molina-Hernández, Nuria Estrada Saldívar, Aaron Israel Muñiz Castillo, Jesus Ernesto Arias-Gonzalez, Alex H. O'Brien, Melina Soto, Miguel Angel Ruiz Zarate, Alejandro Vega Zepeda, Johanna Calle-Triviño, Macarena Blanco, Sandra Mendoza, Andrea Rivera-Sosa.

Data contributors: Lorenzo Alvarez-Filip*, Gabriela Ang*, Miguel Angel Maldonado*, Eric Annis*, Beatriz Arellano*, Ernesto Arias, Elizabeth Arista*, Alejandro Arrivillaga*, Ximena Arvizu*, Macarena Blanco-Pimentel, Hanna Bo*, Frances Budd, Aram Calhoon*, Johanna Calle-Triviño, Andrew Cameron*, Laura Chanona*, Rachel Crouthers*, Israel Cruz*, Kristen Eickhorst*, Alí Espinosa*, Nuria Estrada-Saldívar*, Maria Fernanda Gastelu*, Baruch Figueroa*, Felipe Fonseca Peralta*, Albert Franquesa Rinos*, Sarah Gall*, Victor Galván, Javier Gonzalez*, Alba González*, Matt Gordon*, Nicte Ha Munoz*, Nathaniel Hatwel*, Nayeli Hernández*, Miguel Hernández*, Ernesto Hevia*, Layze Hoogland*, Roberto Ibarra Navarro*, Maria Jose Hernandez Duenas*, Rebeca Klaus*, Philip Kramer*, Nicole Macpherson*, Luis Malpica*, Monique Mancilla*, Isis Martinez*, Tom Martin, Rosa María Loreto-Viruel*, Melanie McField*, Sandra Mendoza*, Ana Molina*, Kayla Moore*, Italia Moreno*, Aaron Israel Muñiz Castillo*, Alex O'Brien, Jose Olivares*, Fernando J. Pardo Urrutia, Luis Parmenio*, Esmeralda Pérez-Cervantes*, Miguel Perez Castro*, David Placencia*, Blanca Quiroga*, Patricia R. Kramer*, Bernhard Riegl*, Verónica Rivera-Monroy*, Miguel Ángel Ruiz-Zárate*, Robert S. Steneck*, Deborah Spayd*, Lara Steneck*, Armeid Thompson*, Anallie Trottier*, Orian Tzadik*, John Vavrinec*, Alejandro Vega-Zepeda*, George Wheldon*, Carl Wilson*, Andrea Rivera-Sosa*, Mélina Soto, Carolina Garza*, Roxana Rodriguez*, Teresa Martin*, Cristina Cortes*, Sara Gutierrez*, Claudia Padilla*, Edgar Guzman*, Itzel Zamora, Mateo Sabido, Geovana Leon*.

*AGRRA data contributor(s).

INTRODUCTION

Mexico (Caribbean Sea) hosts 858 km² of coral reefs, which represent 3.541% of the Caribbean's total coral reef extent. This coastal zone is a key ecological and socio-economic region for Mexico and is part of the Mesoamerican Reef. It contains diverse ecosystems such as coral reefs, mangroves, seagrass meadows, and coastal lagoons. These habitats support high biodiversity and provide essential services, including shoreline stabilisation, carbon storage, and fishery productivity (Rioja-Nieto and Álvarez-Filip, 2019; Herrera-Silveira et al., 2020). The region's karstic geology-with extensive sinkholes and subterranean rivers-connects terrestrial and marine environments, enhancing ecological interdependence.

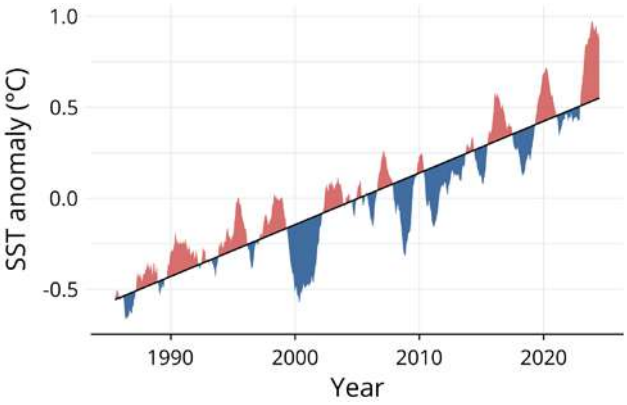
The coastal resources of the Mexican Caribbean sustain the tourism industry, key for its economy which supports thousands of livelihoods. Destinations such as Cancún, Playa del Carmen, Tulum, and Cozumel contribute significantly to the national Gross Domestic Product (14.3% of GDP comes from tourism and this sector represents 8.6% of the national total GDP) (INEGI, 2023). Yet, despite marine protected areas and international conservation designations, the region faces mounting threats from climate change, coral bleaching, *sargassum* influxes, and unregulated coastal development, underscoring the urgent need for resilience-based management.

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Mexico (Caribbean Sea) was 27.83 °C. During this period, SST increased by 1.22 °C, representing a warming rate of 0.031 °C per year (**Figure 2.26.1**). The 1998 basin-wide event marked the first major episode of severe regional heat stress (Muñiz-Castillo et al., 2019). In 2005, bleaching in Mexico was comparatively low to moderate, contrasting with stronger impacts observed elsewhere in the Caribbean (Eakin, 2010).

The 2015–2017 heat-stress event was intense: surveys reported >25% of reefs with over 15% of colonies fully bleached, far exceeding previous localized episodes; impacts were heterogeneous, and recovery trajectories were slow (Eakin et al., 2019; Muñiz-Castillo et al., 2024). In 2023, extreme El Niño related warming (16–24 °C-weeks) triggered widespread bleaching across the Mesoamerican Reef, including the Mexican Caribbean, and mortality was documented in keystone taxa such as *Acropora palmata* (McField et al., 2024; Birkart et al., 2023).



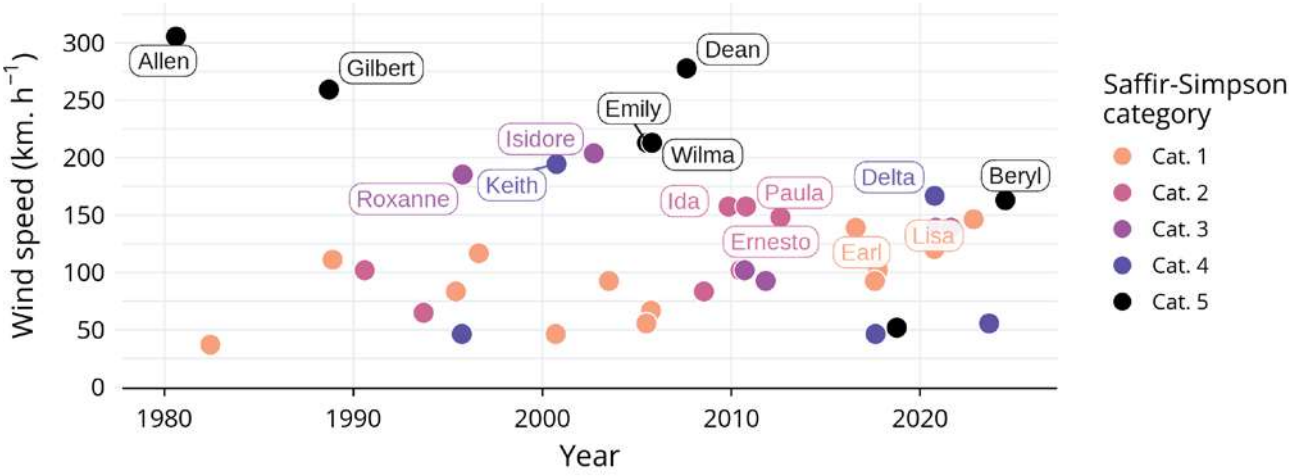
↑ **Figure 2.26.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Mexico (Caribbean Sea). The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, 55 tropical storms passed within 100 km distance from a coral reef in the Mexican Caribbean, 26 of which had sustained winds exceeding 100 km.h^{-1} (**Figure 2.26.2**). The strongest was Hurricane Allen (1980), passing 39 km from a coral reef with sustained winds of 306 km.h^{-1}.

In this region, coral cover losses of up to ~56% have been recorded at some sites (especially in foliose and digitate morphologies), a decline well above averages reported for the Caribbean and the Atlantic, likely due to the back-to-back impact of two hurricanes in 2005 (Gardner, 2005; Álvarez-Filip et al., 2009; González-Barrios et al., 2023).

For this region, the role of reefs as natural breakwaters has been clearly documented: in 2005, after Hurricane Wilma, the reef system of Puerto Morelos dissipated wave energy, altered lagoon circulation, and promoted onshore transport, resulting in beach accretion (≈30 m); in contrast, the exposed beaches of Cancún (without a reef) lost sand and suffered severe erosion (Mariño-Tapia et al., 2014). Given their importance, a parametric insurance scheme for reefs and beaches has been in operation in the Mexican Caribbean since 2019; it has already been triggered to finance post-hurricane repairs (*e.g.*, Delta 2020) a rapid-response tool not yet widely adopted internationally.



↑ **Figure 2.26.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Mexico (Caribbean Sea). Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Mexico (Caribbean Sea) are not represented, although they can have had an impact.

Other threats

Coral reefs in the Mexican Caribbean are affected by multiple stressors, including pollution from wastewater, agricultural runoffs and solid waste (including plastics), overfishing, and poorly regulated recreational water activities. Local nutrient inputs exacerbate *sargassum* blooms, which further contribute to reef degradation, causing coral mortality, a decline in herbivorous organisms (such as fish, sea urchins), and an increase in algal coverage.

Coral diseases

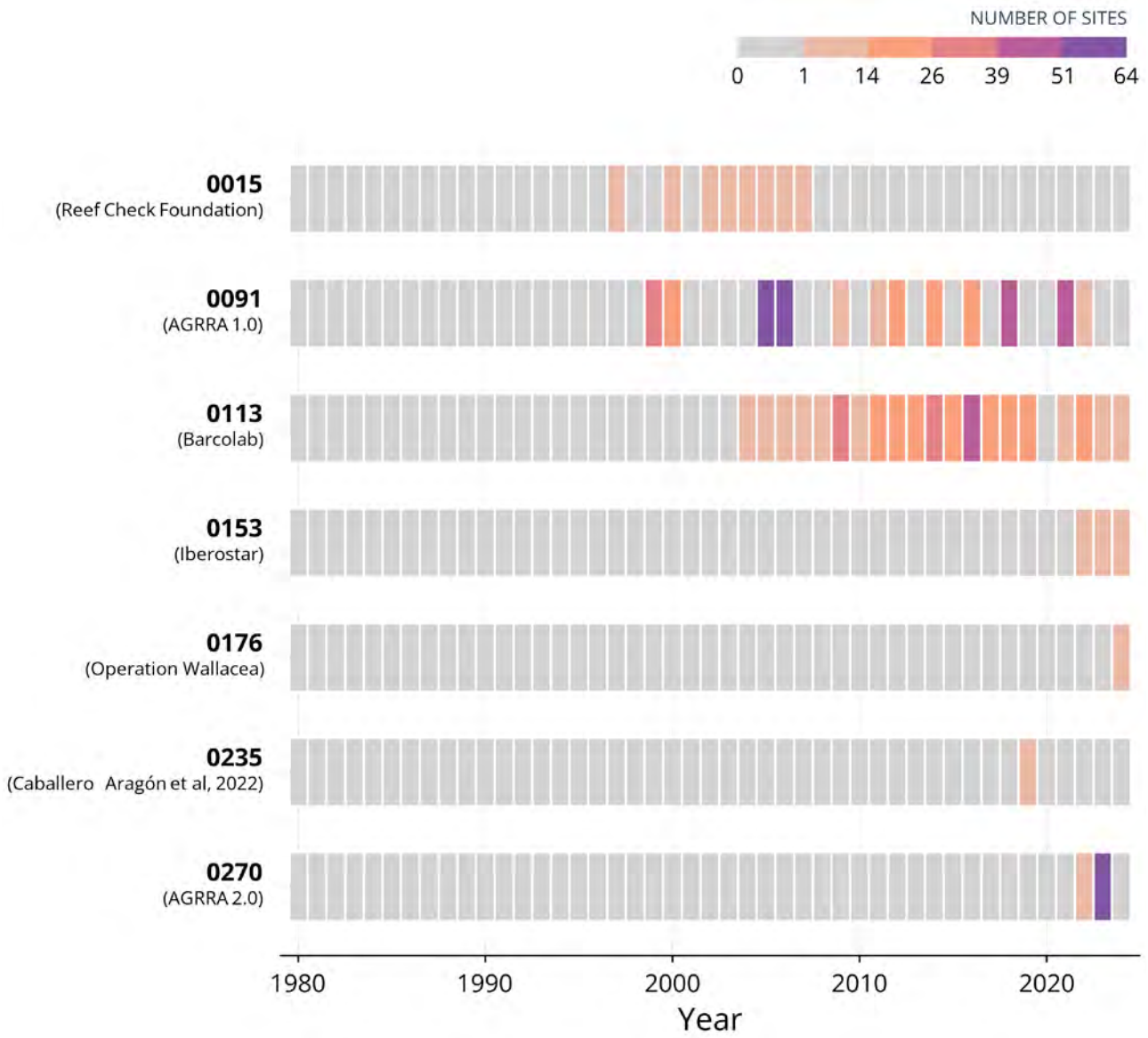
Coral diseases are a major driver of reef degradation in the Mexican Caribbean, threatening the region's ecological integrity (Weil, 2004). Since the 1970s, outbreaks such as White Band Disease (WBD), Yellow Band Disease (YBD), and Stony Coral Tissue Loss Disease (SCTLD) have caused extensive mortality (Harvell et al., 2007; Jordán-Dahlgren et al., 2005; Alvarez-Filip et al., 2022). WBD decimated *Acropora palmata* (Aronson and Precht, 2001), whereas YBD has primarily affected *Orbicella* spp. (Harvell et al., 2007). SCTLD, first reported regionally in 2018, rapidly spread across the region, infecting over 20 coral species, including *Orbicella* spp. and *Pseudodiploria strigosa* (Alvarez-Filip et al., 2019). Its high virulence and broad host range have drastically altered coral community structure and reduced reef functionality (Alvarez-Filip et al., 2022).

Sargassum Blooms and Eutrophication

Since 2011, large masses of pelagic *Sargassum* spp. have proliferated annually across the Caribbean Sea. Although highly variable among years, rafts frequently accumulate along shorelines, and their decomposition produces leachates. In the Mexican Caribbean, reef lagoons are among the most susceptible and affected habitats, particularly in water quality: increases in turbidity, decrease in oxygen, modifications in pH and REDOX potential, particulate organic matter deposition. These conditions impair corals, associated organisms, and elevate nutrients that fuel benthic algal blooms (Cabanillas-Terán et al., 2025, Olguín-Maciel et al., 2022; Rodríguez-Muñoz et al., 2021; van Tussenbroek et al., 2017).

TRENDS IN BENTHIC COVER

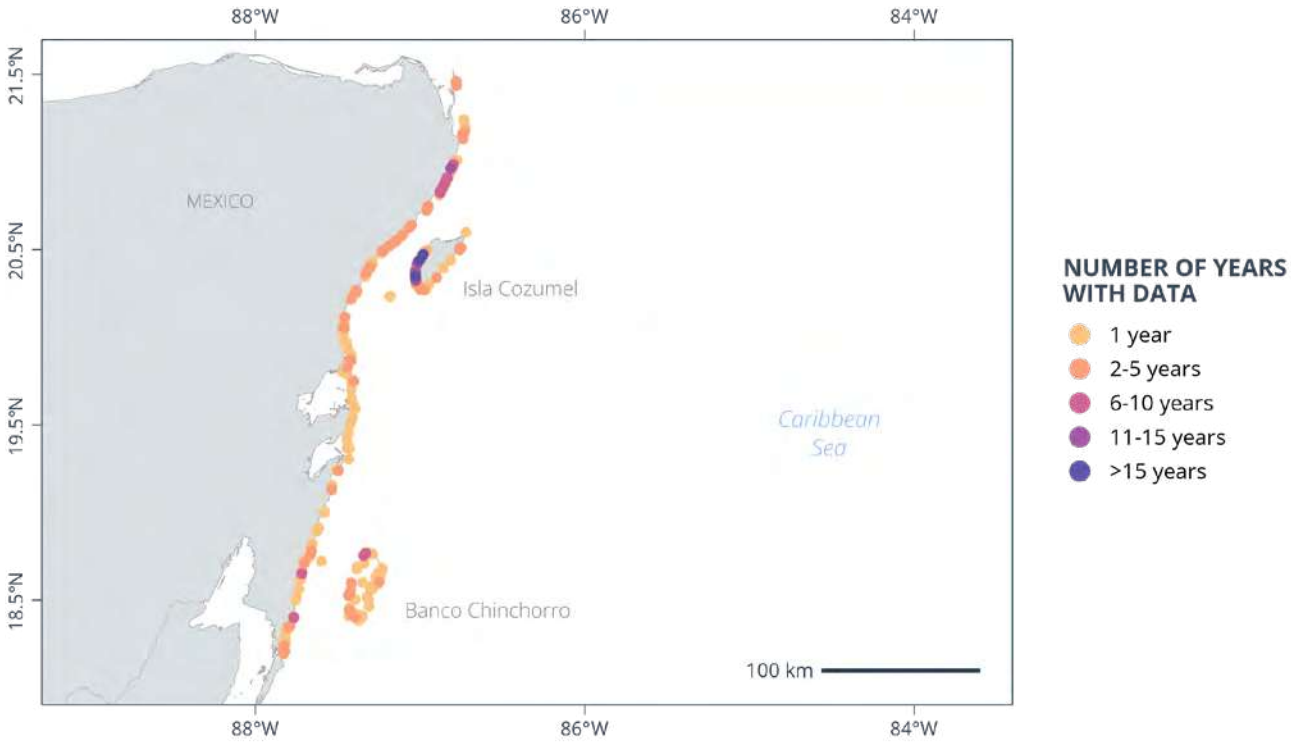
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 7 datasets (**Figure 2.26.3**) encompassing 381 monitoring sites. These data were collected through 934 surveys conducted between 1997 and 2024.



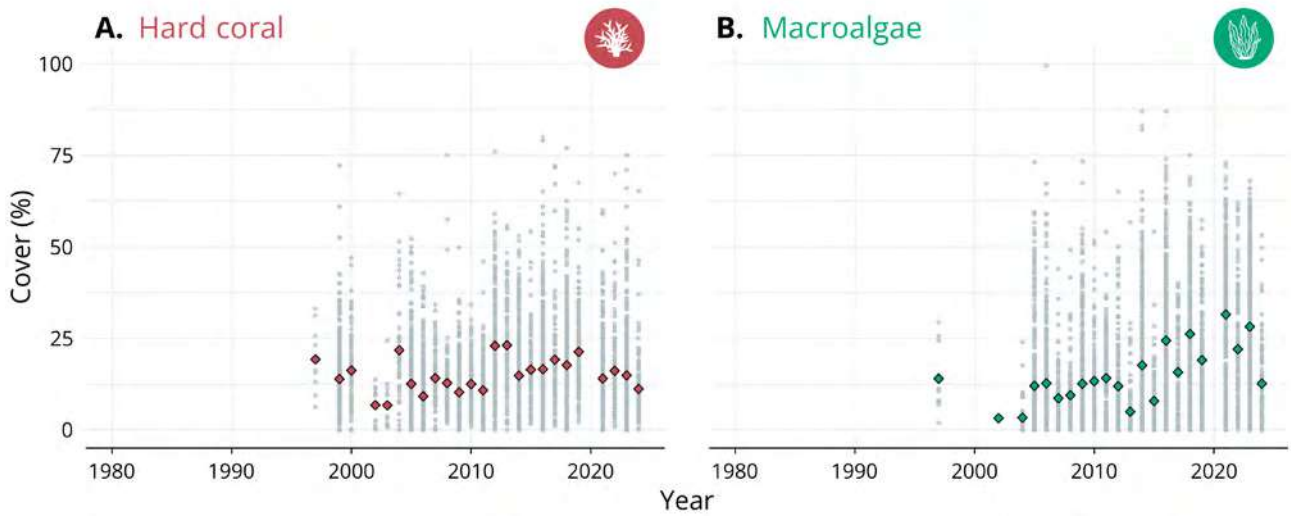
↑ **Figure 2.26.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

The coral and algae cover trend showed in **Figure 2.26.5** follows a similar pattern found in other works: Inside a northern MPA the coral cover trend was from ~38% to ~8%, while algae were from ~39% to 85% in 17 years (1996-2013) (Victoria-Salazar et al. 2017); moreover in a southern MPA coral cover ranged from 6.9 to 8.9% in 2008–2009 to 6.5% in 2021 (Díaz-López et al. 2023). These trends in coral cover have been related to the combined effect of hurricane, diseases, bleaching and currently to the effects of leachates of *Sargassum* spp.

The decline in coral cover has involved a phase shift from coral to macroalgal dominance, with reduced three-dimensional complexity and a turnover from framework-builders to opportunistic taxa (Arias-González et al., 2017; Contreras-Silva et al., 2020). While recovery is limited by nutrient enrichment, weakened herbivory, and other factors, the capacity to produce calcium carbonate, and to sustain reef growth, has declined markedly, threatening their persistence under rising seas and accelerating climate change (Molina-Hernandez et al., 2024). Tools that directly address these root drivers are urgently needed.



↑ **Figure 2.26.4.** Spatio-temporal distribution of benthic cover monitoring sites across Mexico (Caribbean Sea). Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.26.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Mexico (Caribbean Sea) from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

Protecting the Mesoamerican Reef through Blended Finance: The MAR+Invest Case Study

Authors: Amy Jones[1], Judith Morales, Roxana C. Elorriaga

[1] MAR Fund

The Mesoamerican Reef (MAR) is the largest barrier reef in the Western Hemisphere, spanning over Mexico, Belize, Guatemala and Honduras, and is a critical ecosystem for biodiversity, climate resilience, and the livelihoods of over three million people. It provides food, coastal protection, and income from tourism and fisheries. However, like reefs worldwide, it faces mounting threats, the most important ones being: overfishing, untreated wastewater, and climate change1. Conservation efforts in the MAR have traditionally relied on philanthropic or donor funding. While essential, these sources are often short-term and limited in scale. To achieve long-lasting impact, there is a growing need to support scalable business models that deliver both environmental benefits and economic returns.

One approach to unlocking such models is through blended finance, which brings together philanthropic, public, and private capital. In this model, grants or concessional funding are used strategically to reduce risk or offer incentives, making investments in complex areas like ocean conservation more viable and attractive to commercial partners. This mechanism can accelerate the growth of sustainable enterprises while delivering measurable impact for people and reefs.

How MAR+Invest works:

MAR+Invest is a regional blended finance initiative designed to channel capital into "reef-positive" businesses that directly or indirectly contribute to the health of the MAR by addressing its most important threats. Through a combination of grants, concessional capital, and commercial investments, it helps de-risk reef-positive ventures and attract new sources of private capital.

MAR+Invest provides tailored technical assistance, mentorship, business development services, and access to financing. It targets key sectors such as wastewater treatment, sustainable fisheries, regenerative aquaculture, and ecotourism. Impact is measured through a science-based framework with environmental and social indicators.

The initiative is led by a strategic alliance of regional partners: MAR Fund (Technical Assistance Facility), the Mexican Fund for the Conservation of Nature and Sureste Sostenible (Build & Connect mentorship programme), Healthy Reefs for Healthy People (reef health monitoring and evaluation), New Ventures (Business Acceleration Programme and Transactional Services), and Viwala (concessional loans). MAR+Invest is supported by the Global Fund for Coral Reefs (GFCR) and aims to mobilize US$45 million in private capital (through loans, equity, debt and grants) by 2030 to scale reef-positive solutions and strengthen coastal community resilience.

↑ **Figure 1.** Entrepreneurs of the second cohort of the MAR+Invest Acceleration Programme attending the final workshops during the closing event, Puerto Morelos, Mexico, April 2025.

Success Stories from the MAR+Invest initiative:

- **Terminator Water Solutions (TWS) – Tackling water pollution**

TWS is a Mexican wastewater treatment company that developed the WOMBAT 1000 system to treat domestic wastewater at small and medium scales. By reducing nutrient runoff, it helps prevent macroalgae growth that harms coral reefs. In 2024, TWS joined the MAR+Invest Acceleration Programme, receiving mentorship to strengthen their business model and define impact indicators. Recognizing its environmental and financial potential, MAR+Invest provided a concessional loan in 2025, allowing TWS to scale its technology in Quintana Roo, Mexico.

"One key benefit from the 2024 Acceleration Programme was learning how to define and measure our environmental and social indicators so we can demonstrate our positive impact on the Mesoamerican Reef." – *Sarah Connor and José Antonio Cano, Terminator Water Solutions*

↑ **Figure 2.** Terminator Water Solutions demonstrating their new technology, WOMBAT 1000, to donors and partners of MAR+Invest.

- **Maritime Procurement Services (MPS)** – Innovation in Cruise ship Waste Management

MPS is a Mexican company that specializes in the collection, treatment, and recycling of maritime industry waste to reduce ocean pollution and decarbonize the shipping sector. With MAR+Invest's support as financial advisor and capital mobilizer, MPS secured private investment in 2024 to build a new waste treatment facility in Cozumel, Mexico—the largest cruise port in the Caribbean and a critical site close to the Mesoamerican Reef. The new plant will process waste, including hazardous materials, significantly reducing the risk of marine pollution near the reef. MAR+Invest's Transactional Services, which supports companies with financial advice and fundraising to scale impact in the MAR, played a key role for MPS to mobilize capital. It structured its investment proposal and facilitated connections with key regional partners such as Coral Reef Alliance (CORAL).

MPS attracted commercial capital through Viwala due to its strong business model and environmental impact. The investment will help MPS expand capacity, lower costs, and scale its waste-to-energy technology for producing alternative industrial fuels.

↑ **Figure 3.** Waste management plant of the Maritime Procurement Services.

- **Scaling Solutions for Sustainable Fisheries**

Overfishing, often driven by the economic vulnerability of fishers, remains a critical challenge. In the Caribbean coast in Honduras, MAR+Invest, with the support of the BRIDGE Project, is developing innovative financial tools to support small-scale fishers, like small loans and parametric insurance to cover income loss from extreme weather events and tied to conservation agreements. These mechanisms incentivize fishers to follow sustainable practices, such as respecting seasonal closures and no-take zones, aligned with national laws. Although these tools are still under design, they offer a promising model to align financial support with long-term conservation goals.

↑ **Figure 4.** Trainings for fishermen in Honduras.

↑ **Figure 5.** Beach clean-up in Mexico.

- **Looking Forward**

As conservation financing is evolving, MAR+Invest demonstrates how innovative finance can be a catalyst for ecosystem protection. By supporting businesses that combine sustainability with profitability, the initiative redefines conservation as an investable opportunity—moving beyond grants to build resilience through sustainable enterprise.

MAR+Invest aims to expand its portfolio of reef-positive ventures, strengthen public-private partnerships, and unlock new streams of finance to protect the MAR for future generations, by aligning economic incentives with urgent environmental needs and financial sustainability.

Photos credit: Maritime Procurement Services

CHAPTER 27

Mexico (Gulf Of Mexico)

Authors: Nuno Simoes, Ana Molina-Hernández, Rodolfo Rioja-Nieto, Esmeralda Pérez, Andrea Rivera-Sosa.

Data contributors: Gilberto Acosta*, Lorenzo Alvarez-Filip*, Jimmy Arguelles Jimenez*, Guadalupe Barba-Santos*, Rafael Cabral Tena*, Merary Contreras Juarez*, Alejandro Cordova Morales*, Gabriel Garcia Ayala*, Joaquin Rodrigo Garza, Jhoan Gutierrez*, Guillermo Horta-Puga*, Judith Lang*, Karen Macias*, Melissa Mayorga Martinez*, Tatiana Medel Alvarado*, Eduardo Palacios*, Fernando J. Pardo Urrutia, Esmeralda Perez-Cervantes, Angela Randazzo Eisemann, Mariana Reyna, Edgar Tovar*, Angelica Vazquez Machorro*.

*AGRRA data contributor(s).

INTRODUCTION

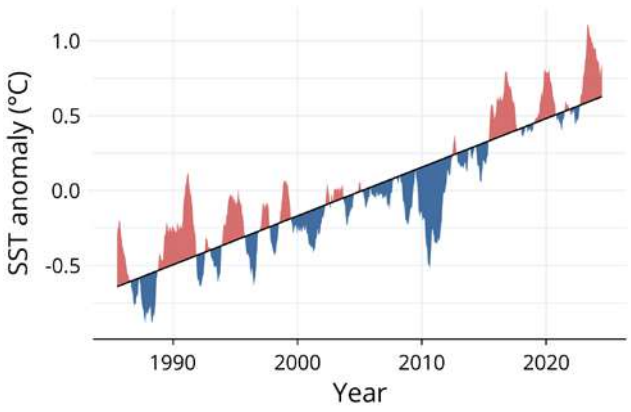
Mexico's Gulf of Mexico shelf (Gulf of Mexico) hosts a series of platform reefs distributed across relatively small areas. It includes thirteen main reef systems: Alacranes in Yucatán; Cayo Arenas, Bajo Nuevo, Bancos Ingleses, Triángulos Oeste, Triángulos Este, Triángulos Sur, Banco Pera, Bajos Obispos, Banco Nuevo, and Cayo Arcas in Campeche; and the Lobos-Tuxpan Reef System and Veracruz Reef System (Jordán-Dahlgren and Rodríguez-Martínez, 1998; Chavez et al., 2010 Zarco-Perelló et al., 2013). This region hosts 582 km² of coral reefs, which represent 2.40% of the Greater Caribbean's total coral reef extent. These reefs are ecologically significant due to their isolation and unique environmental settings, which range from turbid, sediment-rich coastal waters to clear, offshore oceanic conditions (Gil-Agudelo et al., 2020). Historically, these reefs have developed under challenging conditions, such as significant riverine influence and winter storms, which may have fostered greater resilience compared to their Caribbean counterparts. Notably, some reefs in the Gulf of Mexico (GoM), such as Alacranes Reef are considered among the best-preserved in the wider Caribbean, hosting high coral cover and structurally complex species Randazzo-Eisemann and Garza-Pérez (2023).

→ **Figure 2.27.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Mexico (Gulf of Mexico). The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

THREATS

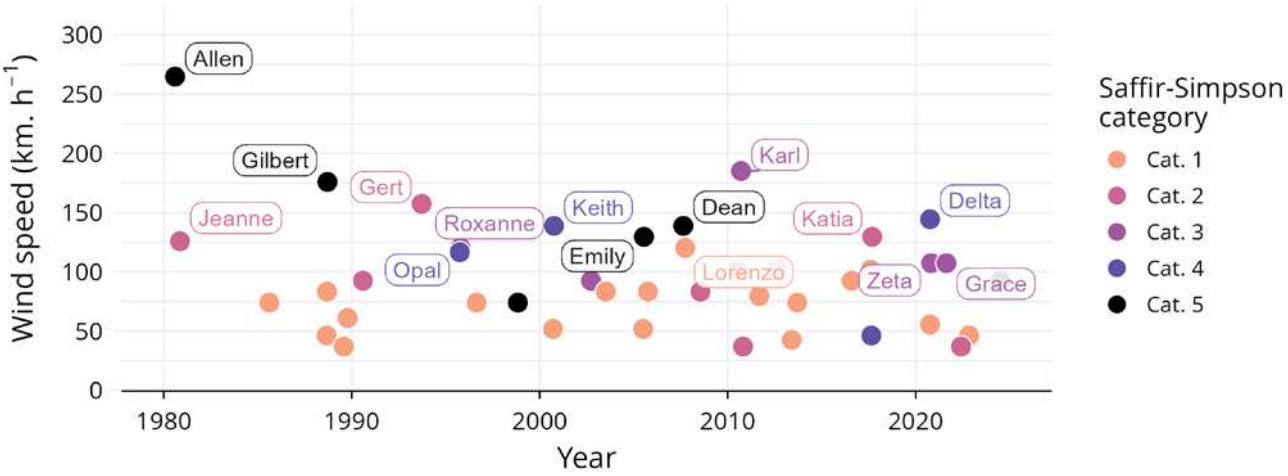
Thermal regime

The long-term mean SST from 1985 to 2024 over coral reefs of the GoM was 26.69 °C. Over this period, SST increased by 1.40 °C, representing a warming rate of 0.036 °C per year (**Figure 2.27.1**). In 2023 and 2024, temperatures in coral reefs located ~200 km offshore recorded temperatures near 31 °C. Although GoM reefs have historically experienced thermal stress, recent decades have seen an increase in the frequency and intensity of marine heatwaves. The 2005 event caused widespread bleaching across the Caribbean, but had only minor effects in the GoM. Upwelling linked to the Campeche cyclonic gyre may partially buffer some reefs by bringing cooler waters to the surface. In 2023 all southern GoM reefs experienced a severe bleaching event that affected most of the remaining *Acropora* spp. colonies.



Hurricanes

Between 1980 and 2024, 87 tropical storms passed within 100 km of coral reefs in GoM, of which 22 had sustained winds >100 km.h^{-1} (**Figure 2.27.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Allen (1980), which passed 13 km from a coral reef with sustained winds of 265 km.h^{-1}. Major hurricanes, with return periods estimated at 37 years for the SW Gulf and 12.3 years for the Campeche Bank, can cause severe mechanical damage. Hurricane Gilbert (1988) caused massive loss of *Acropora palmata* stands that were formerly abundant and healthy. While reefs have historically shown the capacity to recover from such disturbances, this process can be highly variable and delayed by other stressors.



↑ **Figure 2.27.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Mexico (Gulf of Mexico). Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Mexico (Gulf of Mexico) are not represented, although they can have had an impact.

Threats

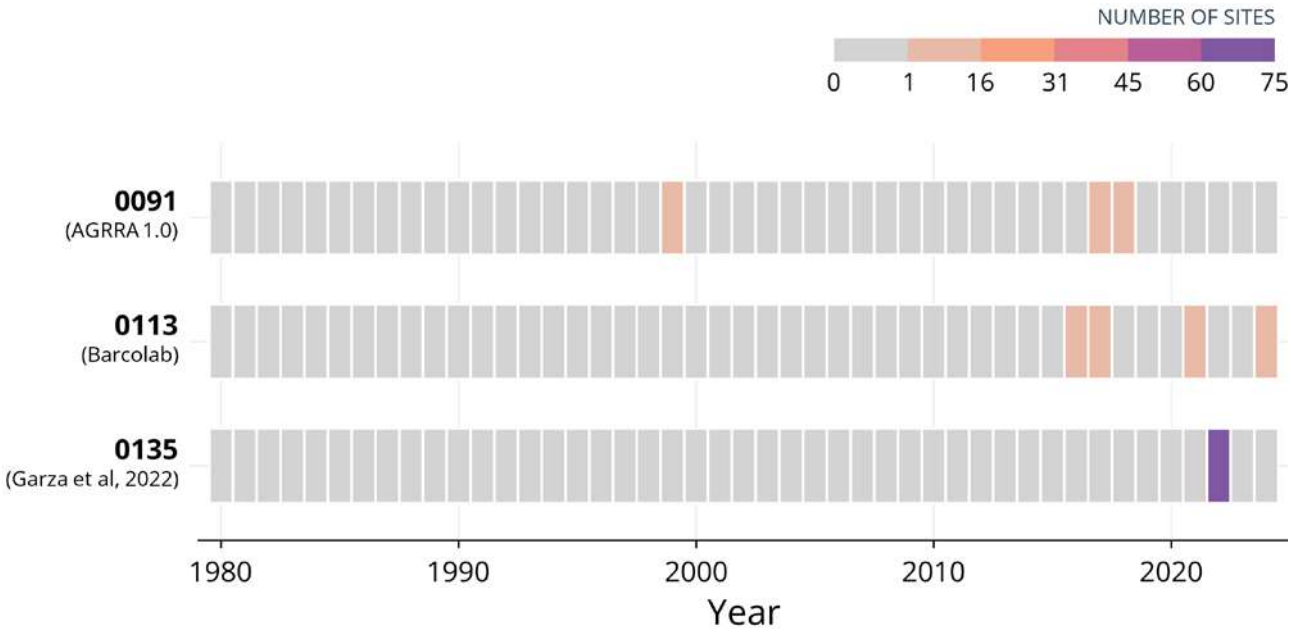
Coral reefs in the GoM face a variety of additional threats. Coral diseases, such as White Band Disease, caused massive mortalities of *Acropora* spp. in the 1970s and 1980s, particularly in the SW Gulf reefs. The lethal stony coral tissue loss disease (SCTLD), which has devastated Caribbean reefs, has not yet been detected in the Mexican Gulf of Mexico. Coastal pollution and development are major stressors, especially for the Veracruz reefs, which are affected by river discharges carrying agricultural and industrial waste, port activities, and urban run-off. (Mateos-Jasson et al., 2012). On the Campeche Bank, oil and gas platforms pose risks of chronic spills and pollution. Overfishing has been a primary use of these reefs for centuries, and while some regulations exist, enforcement is often lacking, leading to depleted stocks of key species like groupers, snappers, and lobster.

Invasive species also threaten reefs through competition, predation, habitat alteration, and disease. In the southwestern/northern gulf, the Indo-Pacific demoiselle *Neopomacentrus cyanomos* is now established, raising concerns about competition with native planktivores and assemblage shifts (Robertson et al., 2016; Bennett et al., 2019). "Sun corals" (*Tubastraea coccinea, T. micranthus*) reach high densities on oil/gas platforms and sometimes spread to natural banks, competing for scarce hard substrate (Sammarco et al., 2015; Hernández-Solís, 2024). One of the best-documented invaders, Indo-Pacific lionfish (*Pterois volitans, P. miles*), are prolific generalist predators that depress native reef-fish recruitment and trigger trophic cascades (Albins and Hixon, 2008; Santander-Monsalvo et al., 2012)

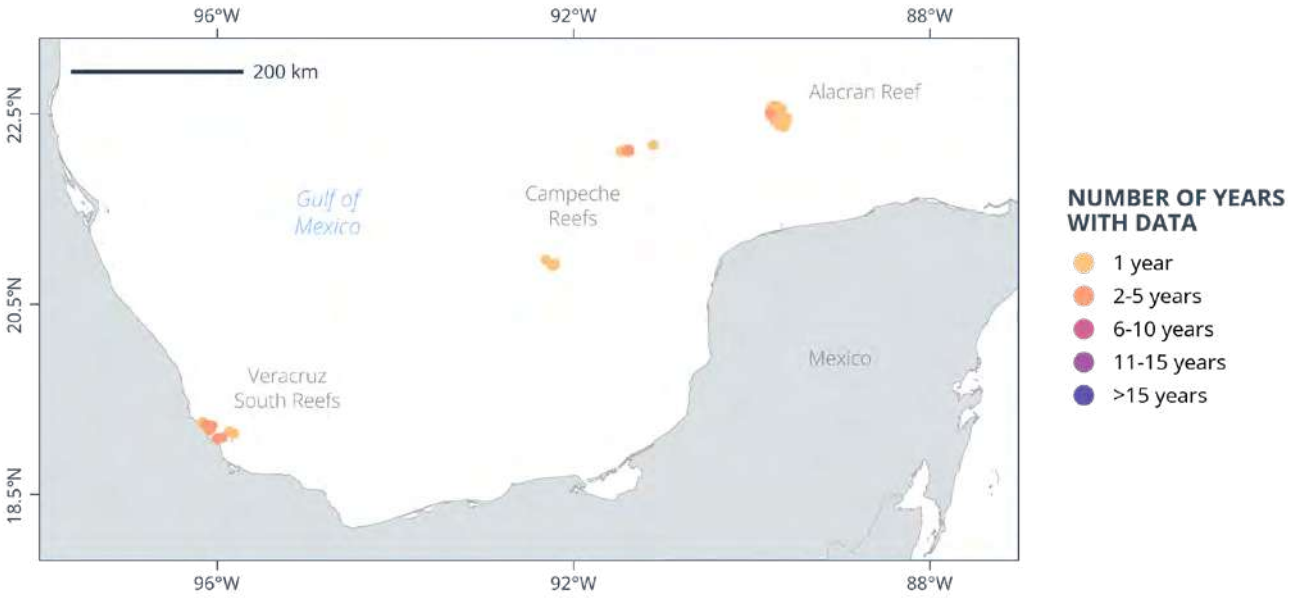
TRENDS IN BENTHIC COVER

Temporal trends in benthic cover from 1980 to 2024 were estimated using data from three datasets (**Figure 2.27.3**) encompassing 123 monitoring sites. These data were collected through 143 surveys conducted between 1999 and 2024.

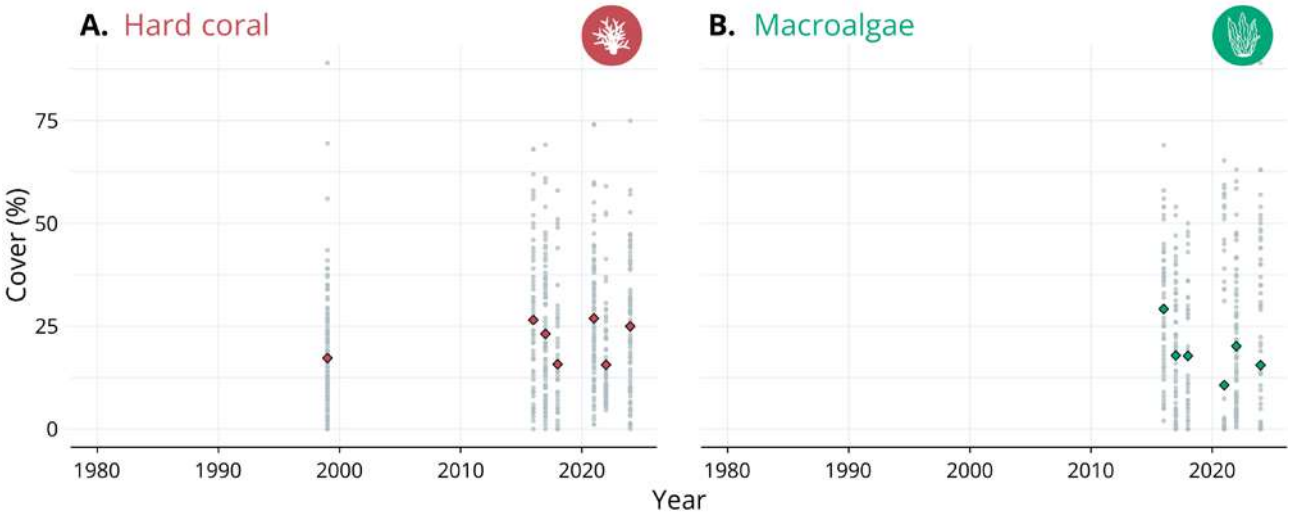
Coral cover in the GoM shows significant regional variability (Pérez-Cervantes, et al 2016). A 2014 report documented a decline in the region from 34.1% in 1965 to 17.2% by 1999. More recent assessments reveal a sharp contrast between different reef systems. A 2022–2023 study led by Correa *et al.* (2025) using the Reef Health Index (RHI) rated the SGM reefs as "good" (average RHI of 3.50) while Mexican Caribbean reefs were rated "poor" (RHI of 2.50). The higher rating in the SGM was driven by significantly greater hard coral cover (average 38.9%) and very low fleshy macroalgae cover (1.41%). Within the SGM, the Lobos-Tuxpan Reef System (APSALT) showed even better health, with an average coral cover of 55.66%, while the Veracruz Reef System (PNSAV) had 22.14%. In contrast, Alacranes Reef on the Campeche Bank shows that 57% of its sites have coral cover above 10%, dominated by structurally complex, stress-tolerant species like Orbicella, making it a potential refuge in the wider Caribbean. In the Cayo Arenas reef system, coral cover decreased ~19% between 2023-2024. Dominant reef-constructing species such as *Orbicella faveolata, Orbicella franksi, Colpophyllia natans* and *Diploria labyrinthiformis* were the most affected.



↑ **Figure 2.27.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.27.4.** Spatio-temporal distribution of benthic cover monitoring sites across Mexico (Gulf of Mexico). Sites monitored for the longest period are displayed above those monitored for shorter periods. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.27.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Mexico (Gulf of Mexico) from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Safeguarding Unique Coral Reef Ecosystems in the Gulf of Mexico

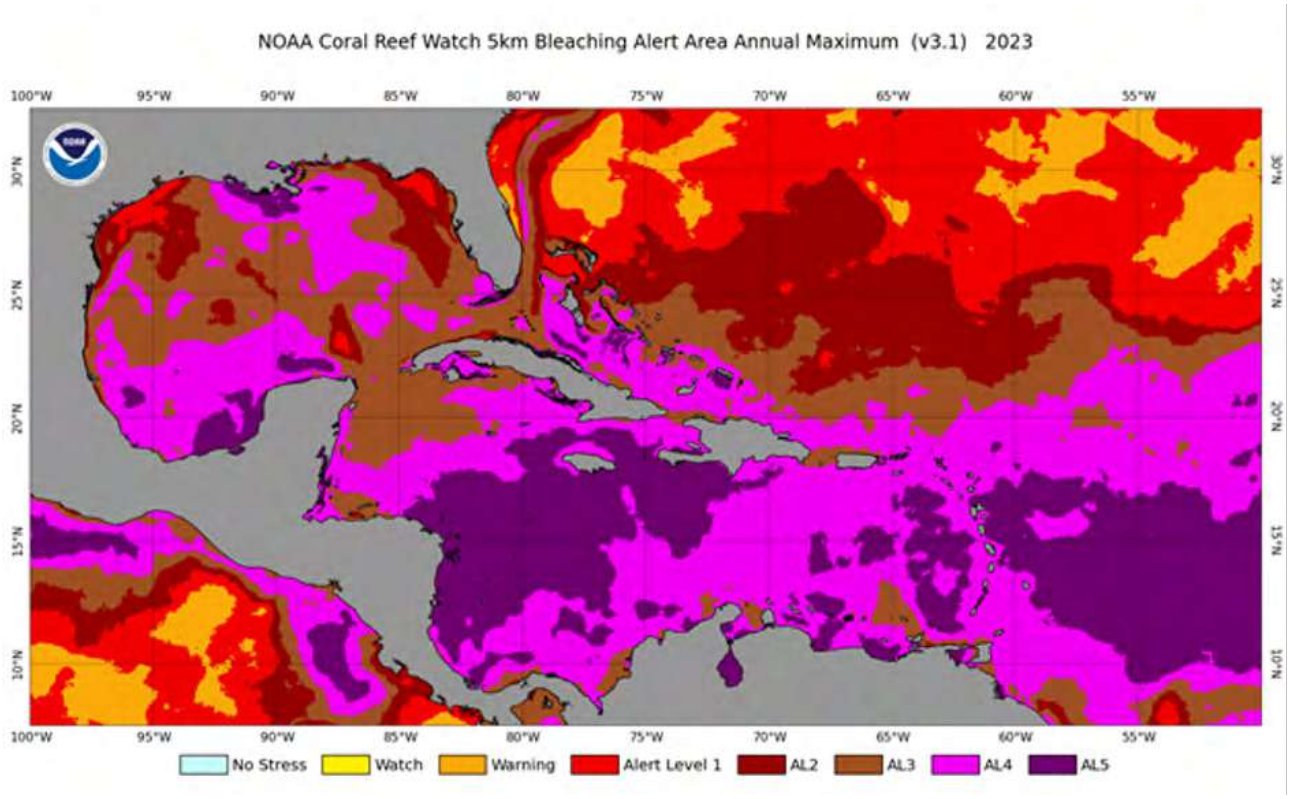
Authors: Ángela Randazzo-Eisemann, Mariana Reyna-Fabian, Antar M. Pérez-Botello

The southern Gulf of Mexico is characterized by remote coral reefs associated with cays, banks, or reef platforms spread over a large geographic area offshore the coast of the Western and Northern Yucatan Peninsula (~140-200 km). This area is connected to the Mexican Caribbean by the Yucatan Channel, a strait between the Yucatan Peninsula in Mexico and Cuba. Despite these oceanographic connections, each region has unique coral reef systems showing different trajectories in the last decades.

Extending 400 km, the Mexican Caribbean, part of the northern section of the Mesoamerican Reef System (MAR), is characterized by a fringing reef system, running parallel to the coast, distributed in a semi-continuous barrier with several geomorphological variations and with some associated islands and cays generally <35 km close to the coast (Rioja-Nieto and Álvarez-Filip, 2019).

The Mexican Caribbean—and particularly the North and Center—has experienced intensive coastal development over the last four decades (Rioja-Nieto and Álvarez-Filip, 2019), chronically impacting the quality of water and the environment where coral reefs developed, making them more susceptible to acute stressors such as bleaching and diseases. Impacted by these multiple stressors, most of the reefs have lost the coral cover and structural complexity through high mortality events induced by the Stony Coral Tissue Loss Disease (SCTLD) outbreak (2018-2019) and the 2023 severe bleaching event (Álvarez-Filip, 2025).

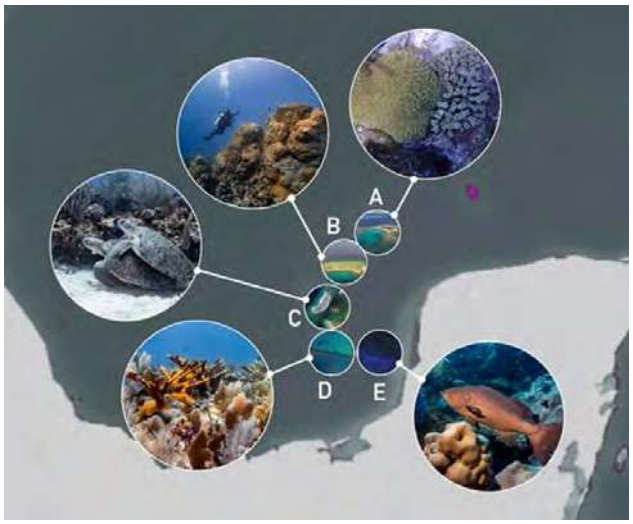
The neighboring and remote reefs of the Southern Gulf of Mexico have also experienced severe heat stress during 2023, with even higher bleaching alerts than the Caribbean (**Figure 1**). To assess the ecological condition of remote reefs, a scientific expedition conducted by Oceana in June 2024, took place in the Southern Gulf of Mexico, allowing to collect reef health and biodiversity data using novel techniques including environmental DNA (eDNA) and reef photogrammetry.



↓ **Figure 1.** Composite Bleaching Alert Coral Watch Product in 2023 (NOAA, 2025). The level of alert (AL) increment from left to right.

This 10-day expedition, navigating through a vast geographical area, enabled us to find signs of hope for the Wider Caribbean (Gulf of Mexico and Caribbean). Despite the thermal stress event in 2023, in these remote reef systems, we found an abundance of high structural and old coral colonies, no sign of SCTLD, and the presence of susceptible corals, presence of critically endangered coral species, high herbivore and predator fish abundance, high coral and fish diversity for the region and presence of macrofauna in feeding, reproductive and nurseries areas (**Figure 2**).

The scientific data generated through the expedition provided a baseline to the Mexican National Commission for Protected Areas (Comisión Nacional de Areas Protegidas, CONANP) to justify the declaration on 27 September 2024 of the biggest Marine Protected Area (MPA) of the Gulf of Mexico: Arrecifes del Golfo-Sur National Park, extending through 41,907 km². Moreover, this system connects with a recently declared protected area Bajos del Norte National Park (13,041.15 km²), and the 3 decades national Park Arrecife Alacranes (3,338 km²). Together, these three MPAs form a contiguous corridor of reef habitats across the Campeche Bank through ~58,300 km².



↑ **Figure 2.** Oceana 10-day expedition through A) Cayo Arenas, B) Cayo Nuevo, C) Cayos Triángulos, D) Bancos Obispos, and E) Banco Nuevo. Examples of signs of hope found (from A to E respectively): no sign of SCTLD and presence of susceptible corals, abundance of high structural and old coral colonies, macrofauna reproductive areas, presence of critically endangered coral species, and predator fish abundance.

The range of depths, habitat types, and thermal conditions support the "stepping-stone" dynamic that allows species—especially corals and reef fish—to move, settle, and thrive across spatial scales, thereby enhancing ecosystem resilience and adaptive capacity. This strategic protection of interconnected reef systems in good ecological condition represents a proactive conservation model. It strengthens biodiversity safeguards, supports fisheries sustainability, and helps buffer marine ecosystems against future impacts. These MPAs not only fill critical gaps in marine conservation in the Gulf of Mexico -they also offer an opportunity to study reef connectivity, resilience, and climate adaptation in one of the most dynamic and productive marine regions in the Caribbean and Atlantic Ocean.

Further research is still needed to fully understand this remote reef system and the reasons for its resistance to thermal stress and coral disease outbreaks. However, these preliminary findings might suggest that when local stressors are low—due in this case to remoteness-, this can give the coral reefs a higher capacity to cope and resist global and regional stressors. Indeed, former research in the Caribbean has shown that reducing the impacts of local stressors and protecting coral reefs through MPAs can give a higher chance of facing thermal stress (Randazzo-Eisemann et al., 2021). In the context of multiple stressors impacting our reef system, these protection actions promote the resilience of these valuable and biodiverse ecosystems against global impacts.

CHAPTER 28

Montserrat

Authors: Laëtitia Mathon.
Data contributors: Alistair Cole.

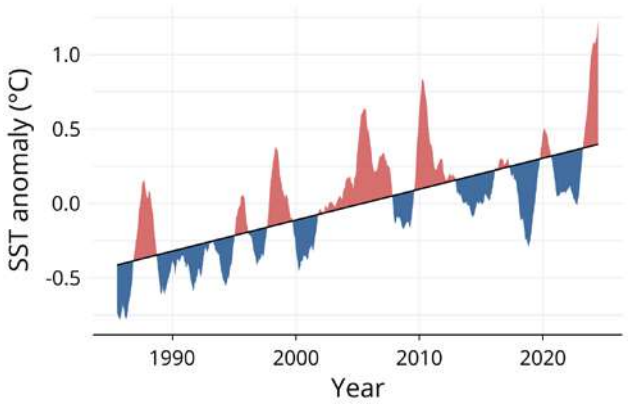
INTRODUCTION

Montserrat, a United Kingdom Overseas Territory in the eastern Caribbean with a land area of 102 km², is home to 8 km² of coral reefs, representing just 0.032% of the Caribbean's total reef extent. These reefs support a wide range of corals, invertebrates, sponges, marine plants, and fish, and provide critical nursery habitats for juvenile species (Myers, 2013). Once home to over 11,000 people in 1980, Montserrat's population declined to around 4,389 by 2024 due to mass emigration following volcanic eruptions in the 1990s. The island also hosts one of the Caribbean's smallest fishing sectors, with only 24 registered artisanal vessels in 2021, targeting reef-associated, demersal, coastal pelagic, and pelagic species (Dosell et al., 2021). Despite its small scale, the sector contributes an estimated annual value of XCD 1.8 million (USD 666,000) to the economy (Eftec, 2019).

THREATS

Thermal regime

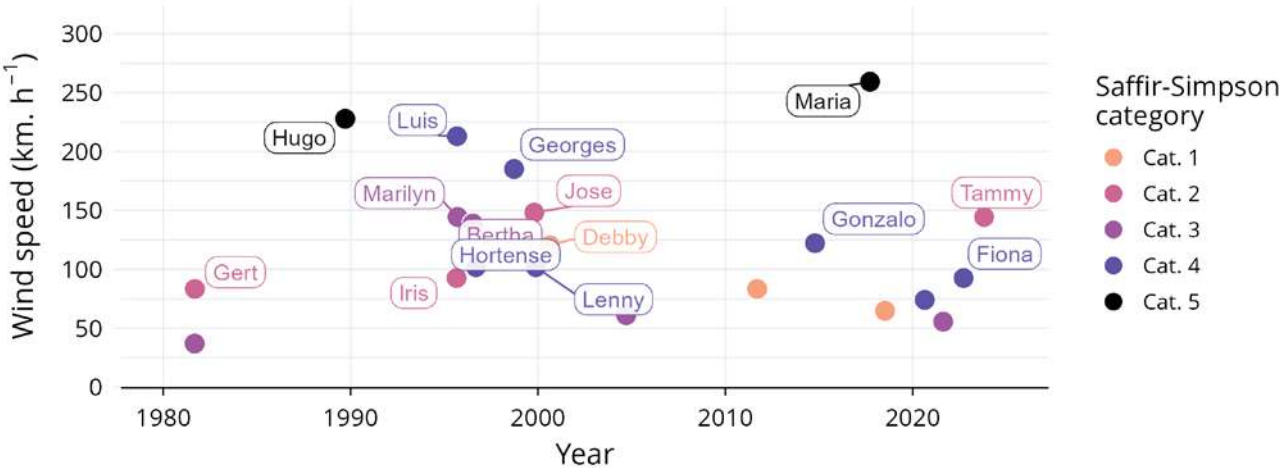
The long-term average in SST from 1985 to 2024 over coral reefs of Montserrat was 27.63 °C. Over this period, SST increased by 0.98 °C, representing a warming rate of 0.025 °C per year (**Figure 2.28.1**).



↑ **Figure 2.28.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Montserrat. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 29 tropical storms passed within 100 km distance from a coral reef of Montserrat, and of these 12 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.28.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Maria, in 2017, which passed 82 km from a coral reef with sustained wind speed of 259 km.h^{-1}.



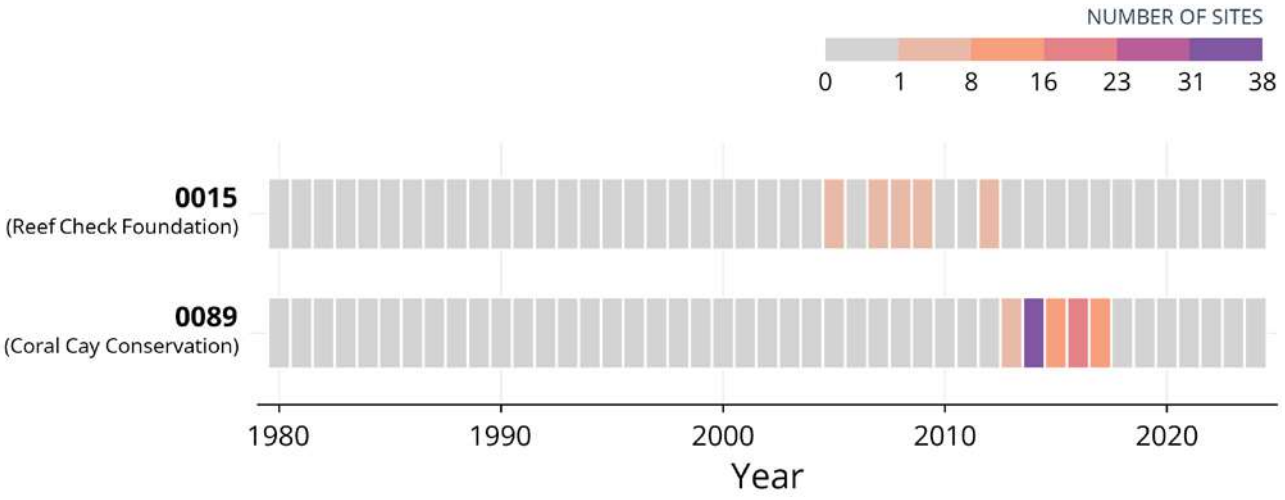
↑ **Figure 2.28.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Montserrat. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Montserrat are not represented, although they can have had an impact.

Other threats

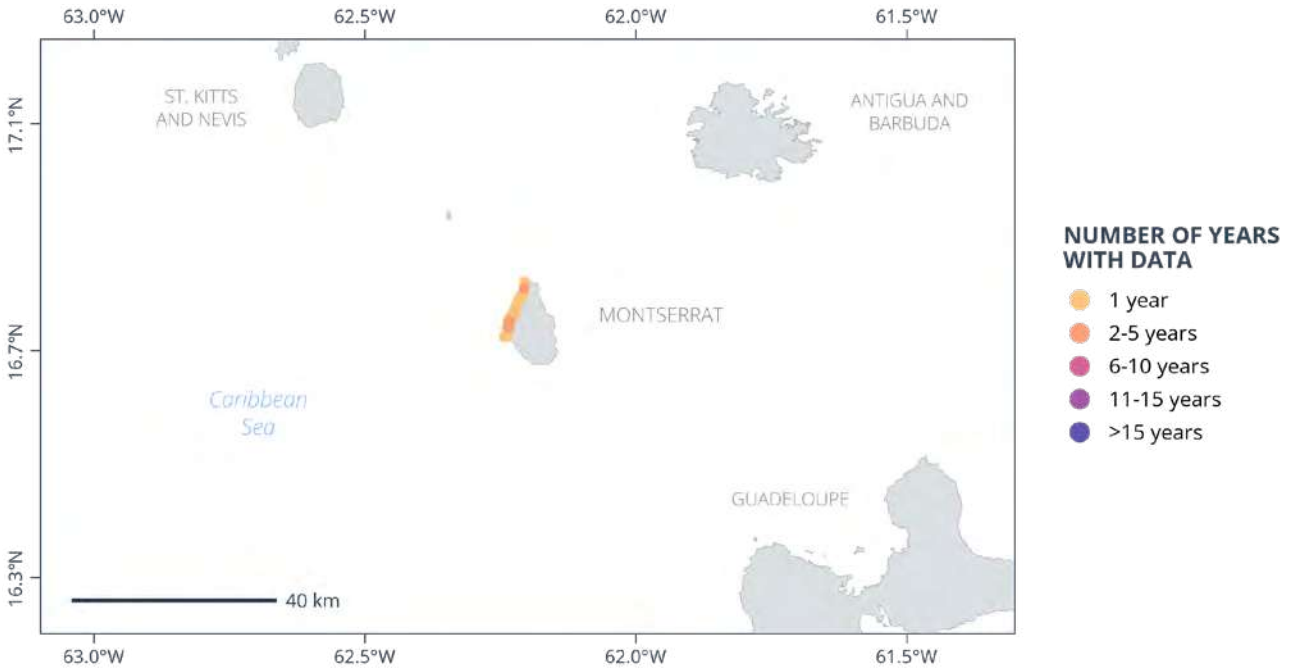
Montserrat's coral reefs are under pressure from both natural disturbances and human activities. Volcanic eruptions have smothered once-productive reef systems with highly acidic material, particularly along the southern coast, while sedimentation and runoff continue to hinder reef recovery (Smith et al., 1997; Estep et al., 2018). These eruptions also created maritime exclusion zones and reduced fishing grounds, concentrating activity in the remaining accessible areas and increasing pressure on the marine environment (Dosell et al., 2021). In addition, expanding shipping linked to the aggregate sector, coupled with the absence of marine spatial planning, has turned the island's west coast into a hotspot of conflict between users. Frequent interactions between fishing gear and vessels damage or displace fish pots, leading to ghost fishing and direct physical impacts on the reefs (Dosell et al., 2021).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.28.3**) encompassing 87 monitoring sites (**Figure 2.28.4**). These data were collected through 109 surveys conducted between 2005 and 2017.

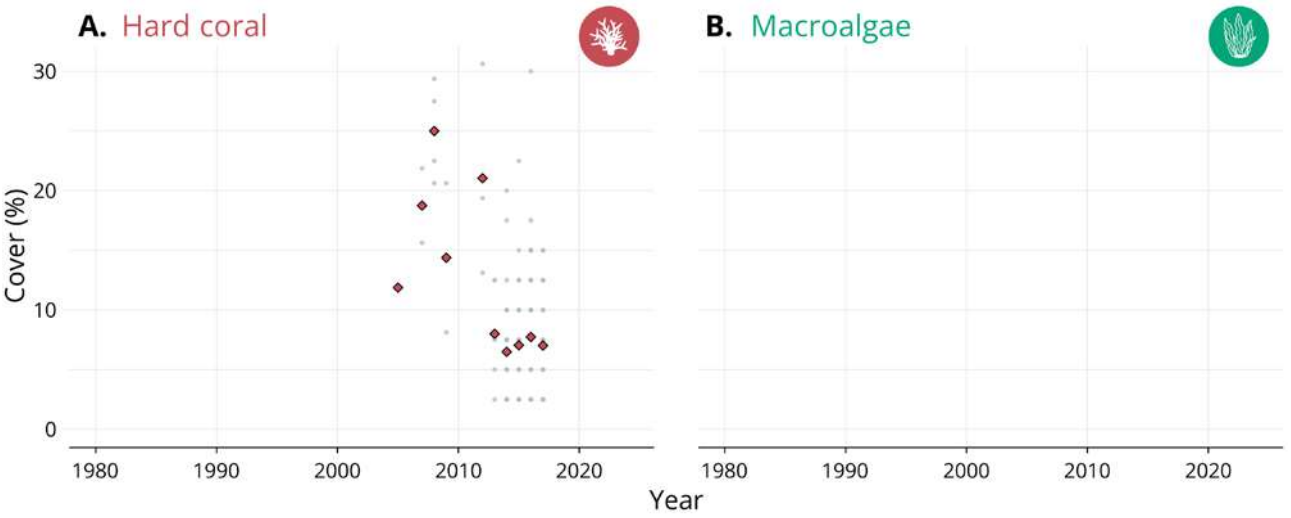


↑ **Figure 2.28.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.28.4.** Spatio-temporal distribution of benthic cover monitoring sites across Montserrat. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Montserrat's coral reefs have experienced long-term decline in health, shaped by repeated natural disasters and ongoing stressors. Hurricane Hugo in 1989 and the prolonged eruption of the Soufrière Hills Volcano between 1995 and 2010 caused widespread damage, with ashfall and pyroclastic flows severely affecting nearshore ecosystems (Berke and Wenger, 1991; Coussens et al., 2017; Myers, 2013). By 2014–2015, reef surveys revealed low coral cover averaging just 10%, with communities dominated by turf algae rather than hard corals (Estep et al., 2018). Overall, coral diversity is also poor, with nearly one-third of sites supporting four or fewer coral species (Ferguson and Daniel, 2015). While Montserrat's reefs show little evidence of widespread coral disease, white band disease has devastated much of the island's Elkhorn coral, leaving only scattered but relatively healthy colonies in the northern and northwestern areas (Myers, 2013).



↑ **Figure 2.28.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Montserrat from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 29

Nicaragua

Authors: Joe Ryan, Yamil Zapata, Alex Hunt, Andrea Rivera-Sosa.
Data contributors: Yamil Zapata* Duncan Hume*, Alex Hunt*, Randolph Velterop*.
*AGRRA data contributor(s).

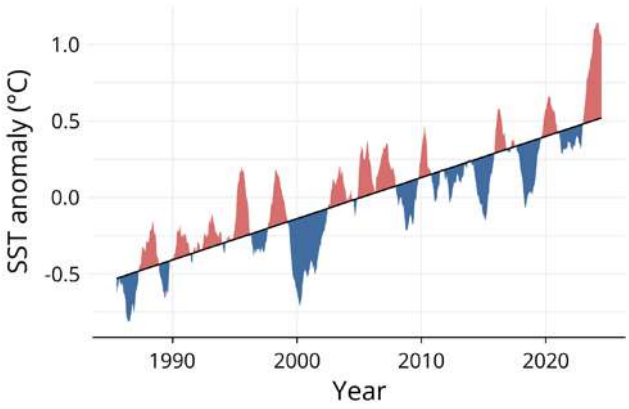
INTRODUCTION

Nicaragua's coasts border the Caribbean Sea and the Pacific ocean, with Honduras and Costa Rica neighboring to the north and south respectively. The Nicaragua shelf (45,000 Km²) - the largest in the Caribbean - hosts over 750 km² of coral reefs, representing 3% of the Caribbean's total coral reef cover. Reef systems are not only concentrated around offshore cays and islands (Corn Islands, the Miskito Cays and Pearl Cays), spanning nearshore, central shelf, and shelf-edge habitats (Ryan and Zapata 2003), but Nicaragua recently doubled its coral reef habitat protection by establishing the 'Nicaraguan Caribbean Marine Biosphere Reserve' (44,000 Km²) after a 2012 ruling by the International Court that divided Colombia's claim to the Seaflower Biosphere Reserve (Garcia 2022). This under-studied region poses significant challenges for reliably tracking ecological changes over time. Reef biota include over 39 species of stony corals in the Miskito Cays (Williamson et al. 2019; Fonseca, 2008), while the 25 stony corals were identified around the Corn Islands (Ryan and Zapata 2003; Ryan et al., 1998).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Nicaragua was 28.09 °C. Over this period, SST increased by 1.16 °C, representing a warming rate of 0.030 °C per year (**Figure 2.29.1**). In a study, which assessed heat stress over 30 years, the Miskito Cays and Nicaraguan islands exhibited high heat stress exposure and in particular during the year 2011 (Muñiz-Castillo et al. 2019). However, no specific studies have been conducted to date to understand the impacts of bleaching events in 1998, 2005, and 2023-2024.

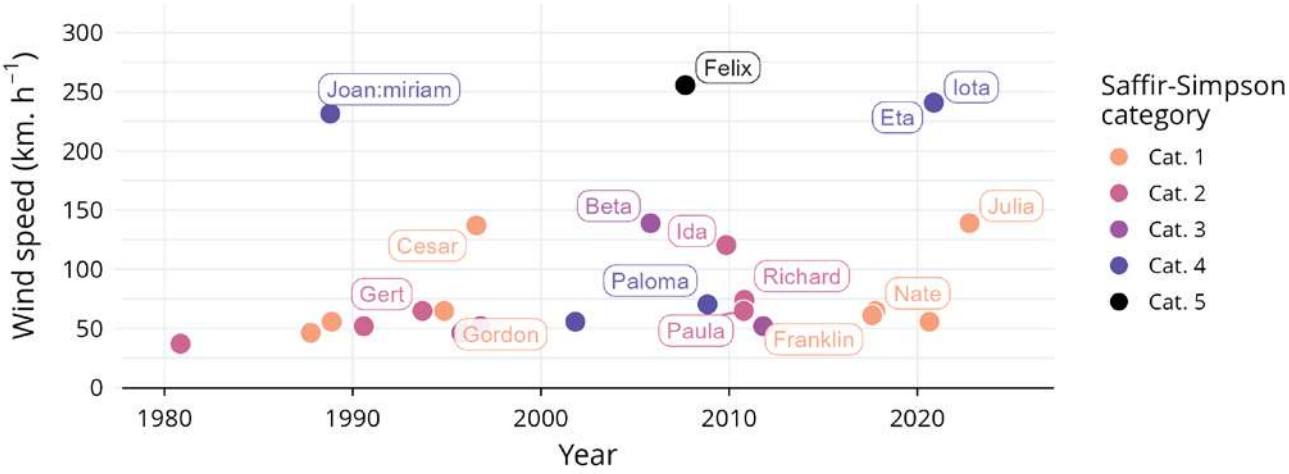


↑ **Figure 2.29.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Nicaragua. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 41 tropical storms passed within 100 km distance from a coral reef of Nicaragua, and of these 8 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.29.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Felix, in 2007, which passed 0 km from a coral reef with sustained wind speed of 256 km.h^{-1}.

↓ **Figure 2.29.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Nicaragua. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Nicaragua are not represented, although they can have had an impact.

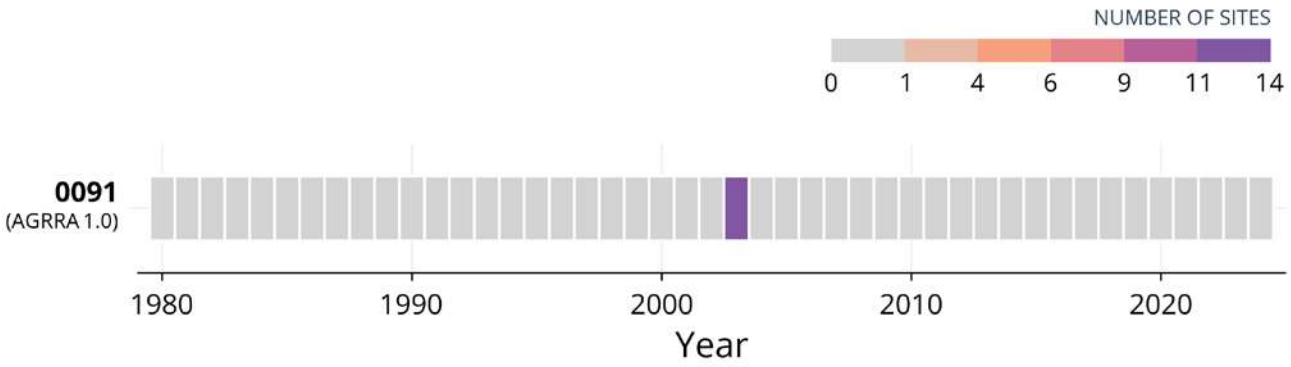


Other threats

The principal direct threats to nearest to mainland Miskito Cays and the Pearl Cays are deforestation-driven terrestrial sedimentation transported seaward by two of the five largest rivers in Central America that expand the volume of the turbid, southward-flowing turbid Coastal Boundary Layer (Ryan, 1992). Overfishing and environmentally impacting coastal development also threaten the nearshore and offshore reefs (Fonseca, 2008; Ryan, 1992), while Corn Islands corals are directly affected by unsustainable fishing impacts (lobster divers using chlorine, physical damage from lobster traps dropped on reefs), as well as pollution, including nutrient enrichment from human sewage reaching the nearshore reefs through fissures in their basaltic foundation. Finally, those threats will continue unless policies and management instruments are developed strategies to address those and other unsustainable activities such as alterations of reef ecological and trophic dynamics are driven by intentional or indiscriminate taking of reef herbivore and juvenile species by Jamaican fish traps, nearshore shrimp trawlers taking larval and juvenile reef fishes, as well as deepwater fish traps set along the shelf edge that take reproductively active carnivores that appear to be important for replenishing natural and fishing mortalities on the reefs (Ryan, 2012; Ryan and Zapata 2003; Ryan 1994).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 1 dataset (**Figure 2.29.3**) encompassing 14 monitoring sites. These data were collected through 14 surveys conducted in 2003.

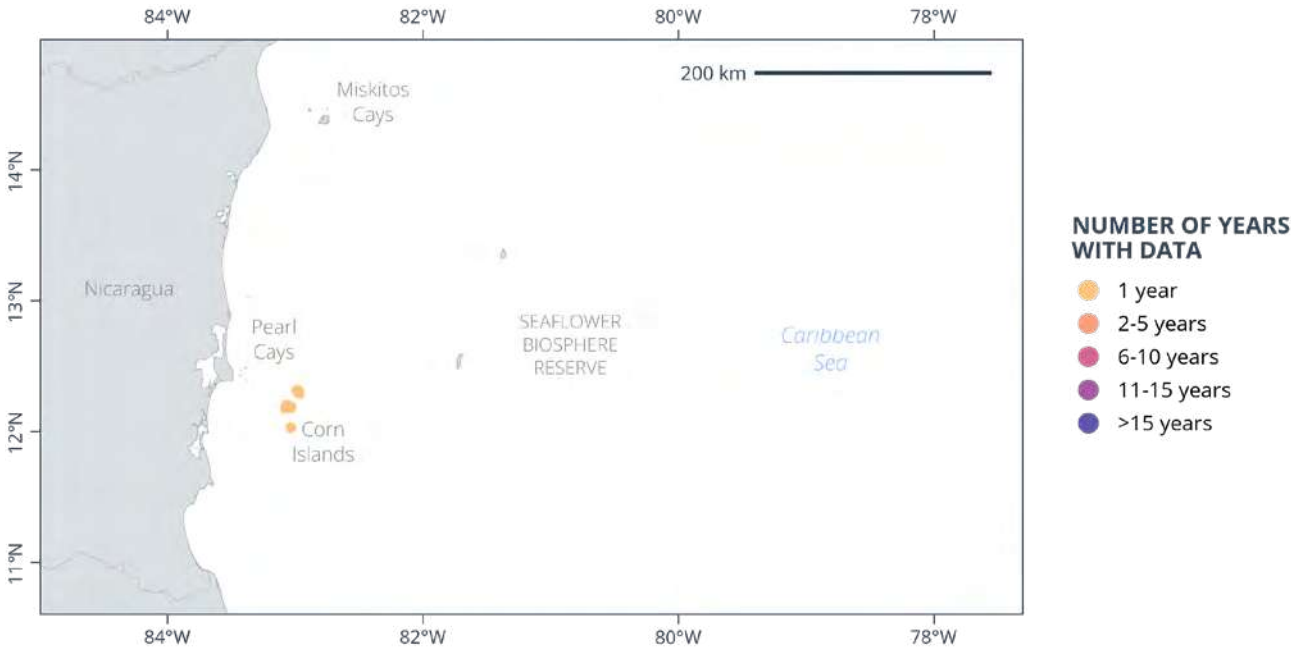


↑ **Figure 2.29.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

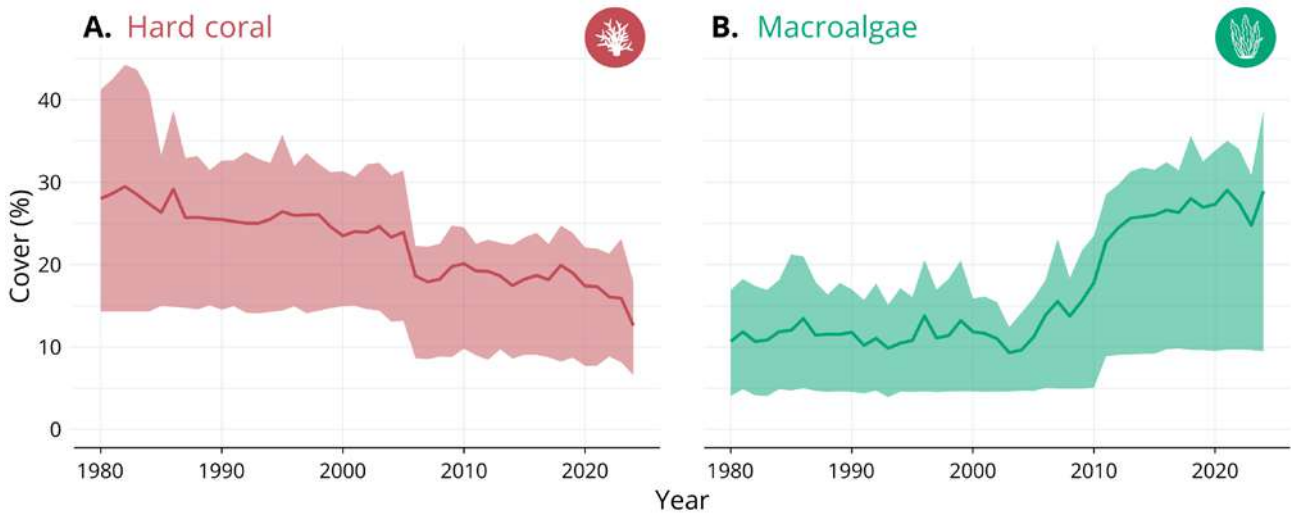
Live coral cover at 18 sites in the Miskito Cays averaged 43.4% (± 14.8; Fonseca 2008) with 54.2 (± 14.1) algal cover. A subsequent study (Williams et al 2019) found four new species with similar live cover and 3-4% bleaching in their transects, as well as a coral health index of 3.29, which exceeded the 2.8 average value for the Mesoamerican reef system. ReefMap (Velterop 2003) recorded 26 species of hard coral and 8 gorgonians at transects at four sites in the Pearl Cays using the species abundance analysis. Despite the heavy sediment loads reaching the Pearl Cays, *Solenastrea bournoni* abundance showed a strong positive correlation with the acid insoluble (inorganic) non-calcareous component), as did *Agaricia tenuifolia* and *Agaricia agaracites*. Positive correlations of sedimentation intolerant species were found between *Porites porites* and turbidity.

The ReefMap study (Hunt 2003) also surveyed 14 AGRRA sites and found only occasional bleaching at Big Corn Island (mean=0.9 % coral colonies affected) and the highest being 2.2 % of sampled colonies, mainly *A. agaricites*. Identifiable sources of coral mortality algal overgrowth was the most prevalent with 3.4 % of all colonies affected and the highest value of 12.6%. Just under 3% of the fifty-eight colonies sampled showed signs of disease, with 72.4 % of all diseases reported to be white plague on *Montastraea annularis* and, occasionally, *M. faveolata*.

Earlier studies of live coral cover at 18 sites on Big Corn Island ranged between 5-55% (BCI 13), while one of five sites on Little Corn Island (LCI 2) reached 90% (Ryan 1994). However, a dive 2 decades later found live cover below 5% (Ryan unpublished data).



↑ **Figure 2.29.4.** Spatio-temporal distribution of benthic cover monitoring sites across Nicaragua. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.29.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Nicaragua from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

CHAPTER 30

Caribbean Panama

Authors: Juan Mate, Judith Lang, Andrea Rivera-Sosa.

Data contributors: Marilyn Brandt*, Arcadio Castillo*, Graham J. Edgar, Robert Ginsburg†*, Heather Halter*, Greta Klungness*, Judith Lang*, Yadibaler López*, Eilikipler Martínez*, Juan Maté*, Ryan Moyer*, Lizzi Oh, Felipe Ossandon, Paula Sills, Rick D. Stuart-Smith.

*AGRRA data contributor(s).

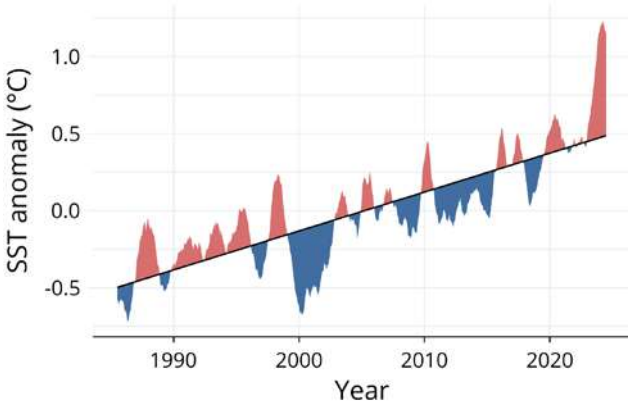
INTRODUCTION

Panama hosts 1,097 km² of coral reefs, which represent 4.527% of the Caribbean's total coral reef extent. Panama's coral reefs are concentrated on the Caribbean coast, where conditions support extensive fringing, barrier, and patch reef formations. The most developed systems occur in the Guna Yala Archipelago (610 km²) to the East, followed by Bocas del Toro Archipelago on the West, and the Central Caribbean Panamá, which spans the Colón Province, from the eastern entrance to the Panama Canal to Escribanos Bay, West of Guna Yala and is closely linked with mangroves and seagrass beds. These ecosystems host a wide range of Caribbean coral, fish, and invertebrate species, while providing coastal protection, fisheries resources, and a foundation for marine tourism. Reefs in Bocas del Toro and Guna Yala are also integral to the cultural identity and livelihoods of coastal and Indigenous communities.

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over the Caribbean coral reefs of Panama was 28.16 °C. Over this period, SST increased by 1.10 °C, representing a warming rate of 0.028 °C per year (**Figure 2.30.1**). Coral reefs along Panama's Caribbean coast have been affected by mass bleaching events. The 1998 El Niño caused moderate bleaching linked to regional thermal stress (Wilkinson and Souter 2008). The 2005 event was severe, with widespread bleaching and coral disease outbreaks in Bocas del Toro (Eakin et al., 2010; Wilkinson and Souter 2008). Most recently, the 2023–2024 global bleaching event caused extensive bleaching and mortality across Guna Yala and Bocas del Toro, driven by record heat stress and deoxygenation (Johnson et al. 2021). These recurrent events underscore the increasing vulnerability of reefs to warming seas.

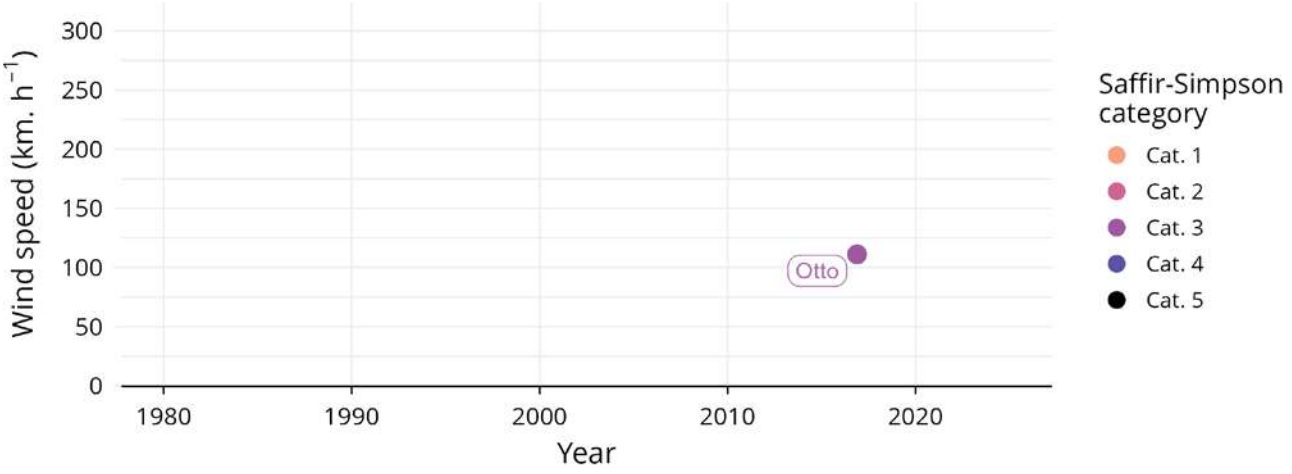


↑ **Figure 2.30.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over Panama's Caribbean coral reefs. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 2 tropical storms passed within 100 km distance of a Panamanian coral reef, and one of these e was characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.30.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Otto, in 2016, which passed 86 km from a coral reef with sustained wind speed of 111 km.h^{-1}.

↓ **Figure 2.30.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in the Panamanian Caribbean. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Panama are not represented, although they could have had an impact.



Other threats

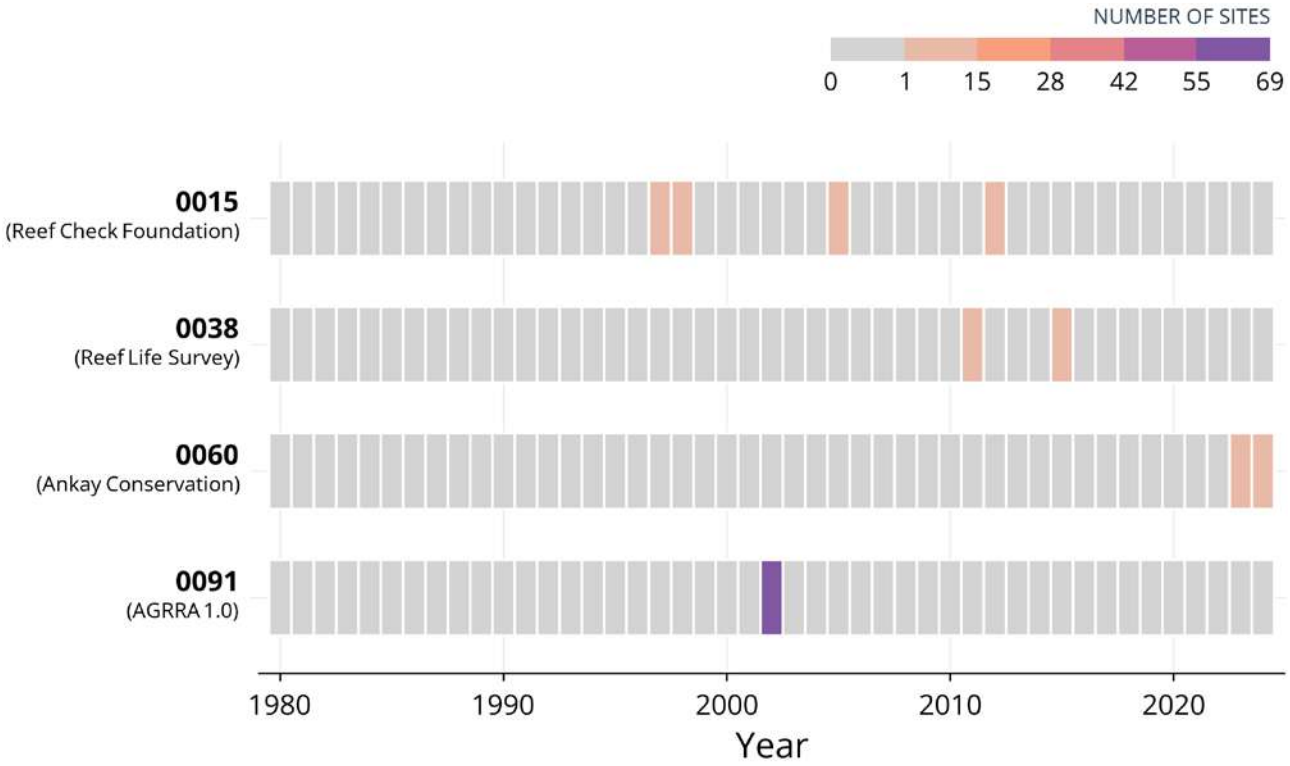
Coral reefs in the Panamanian Caribbean are under growing pressure from local and regional human activities that threaten their ecological health and recovery capacity. Coastal development, deforestation, and land-based runoff are major drivers of sedimentation and eutrophication, reducing light penetration and smothering reef organisms, particularly in areas such as Bocas del Toro (D'Croz et al., 2005; Guzmán and Jiménez, 1992). Pollution from sewage, agriculture, and heavy metals has degraded water quality and altered benthic communities, while nutrient enrichment favors algal proliferation that competes with corals for space and resources (Guzmán et al., 2003). Overfishing and the removal of key herbivores have disrupted trophic dynamics, reducing the natural control of algae and weakening reef resilience. Additionally, unsustainable tourism, including anchor damage and physical trampling, further exacerbates reef degradation in popular coastal and island destinations. Emerging stressors such as ocean acidification and deoxygenation pose long-term threats by impairing coral growth and reef framework stability (Johnson et al., 2021). Low dissolved oxygen levels (hypoxia) has been reported for Almirante Bay in Bocas del Toro resulting in widespread bleaching and mortality of other organisms such as sponges, crustaceans, gastropods and echinoderms (Altieri et al. 2017). To ensure the persistence of Panama's reefs, it is essential to strengthen marine protected areas, enforce sustainable fisheries, improve watershed management, and foster community-based stewardship that integrates local livelihoods with conservation.

TRENDS IN BENTHIC COVER

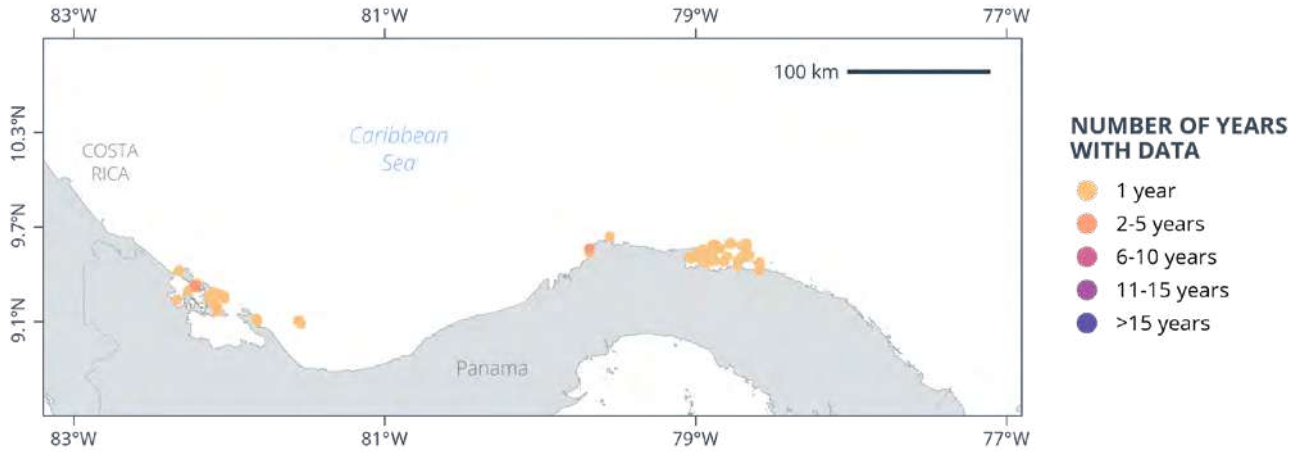
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 4 datasets (**Figure 2.30.3**) encompassing 92 monitoring sites. These data were collected through 184 surveys conducted between 1997 and 2024. Long-term monitoring and historical reconstructions reveal pronounced declines in hard coral cover and increases in macroalgal dominance across Panama's Caribbean reefs over the past four decades which differ from the low macroalgae cover earlier reported in **Figure 2.30.5**. (Shulman and Robertson, 1996; Guzmán, Guevara and Castillo, 2003).

Early surveys in the 1980s documented high coral cover in the San Blas Archipelago (currently known as Guna Yala), averaging 31–45%, which dropped to 12–26% by 1990, while macroalgae rose from ~2% to nearly 30%, largely due to the *Diadema antillarum* die-off, sedimentation, and nutrient enrichment (Shulman and Robertson, 1996).

Similar declines were recorded along the Guna Yala coast, where coral mining, land-based runoff, and local disturbances contributed to a ~79% loss of coral cover between 1970 and 2000, accompanied by a marked increase in algal and rubble substrates (Guzmán, Guevara and Castillo, 2003).



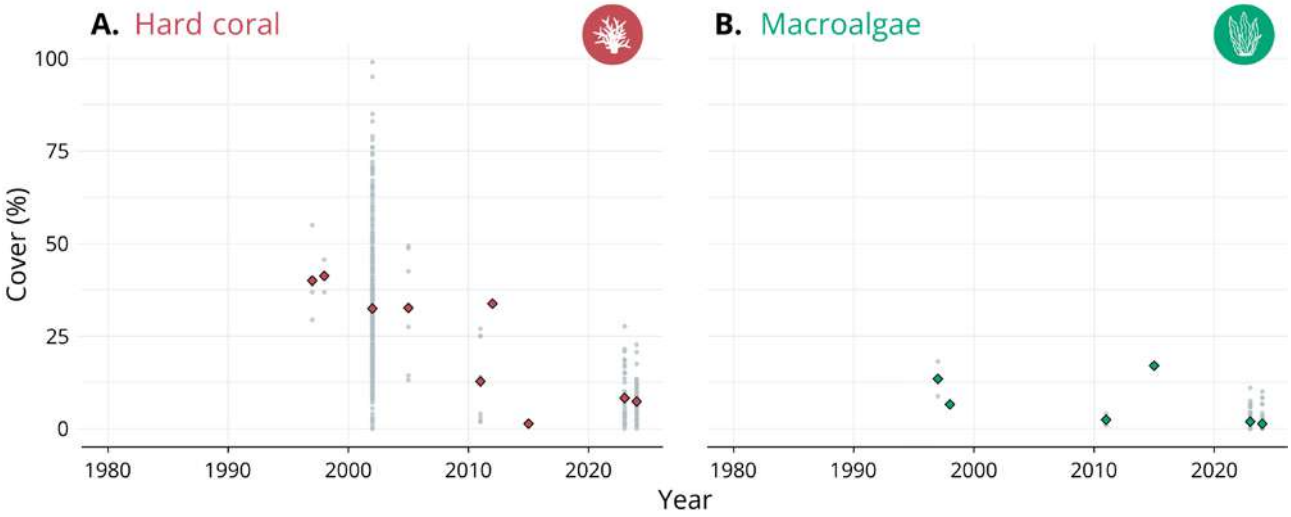
↑ **Figure 2.30.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories in Panama's Caribbean coral reefs. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.30.4.** Spatio-temporal distribution of benthic cover monitoring sites across the Caribbean coast of Panama. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Regionally, these trends mirror Caribbean-wide declines of 50–80% in live coral cover since the 1970s, attributed to cumulative chronic stressors—overfishing, pollution, and disease outbreaks—that reduced reef-building coral abundance and favored opportunistic macroalgae (Cramer et al., 2021). The 2005 mass bleaching and disease event further exacerbated coral loss in Panama and would have slowed recovery trajectories (Eakin et al., 2010).

In the past decade, reefs in Bocas del Toro and the Guna Yala continue to experience low coral cover (often <15%) and persistent macroalgal dominance, sustained by degraded water quality, eutrophication, and limited herbivore populations. Episodes of deoxygenation and sedimentation have caused localized coral mortality and further shifts toward algal-dominated states (Johnson et al., 2021). Collectively, these temporal trends demonstrate a long-term transition from coral-dominated to macroalgal-dominated reef communities in Panama, driven by the interaction of chronic local pressures and acute disturbances. Strengthened management of coastal runoff, fisheries, and marine protected areas is crucial to reverse these declines and enhance reef resilience.



↑ **Figure 2.30.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Panama from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Case study n°6

Coral reef fish abundance and reef rugosity in Bocas del Toro, Panama

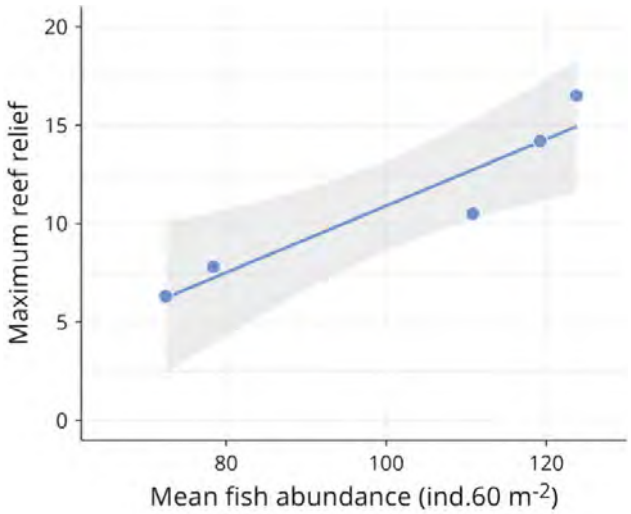
Author: Paula Sills[1]

[1] Ankay Conservation

Coral reef ecosystems are highly productive habitats that support diverse fish communities and play a crucial role in marine biodiversity conservation. The structural complexity of a reef, often measured using Maximum Reef Relief (MRR), plays a crucial role in shaping fish assemblages by providing shelter and feeding opportunities. The structural complexity of coral reefs has been declining, primarily due to the loss of framework-building corals, which poses a significant threat to reef-associated biodiversity. (Alvarez-Filip et al., 2009). This study examines the relationship between fish abundance and reef rugosity in Bocas del Toro, Panama, using a belt transect methodology.

Fish surveys were conducted using the belt transect method, covering a 60 m² area per transect (2m width × 30m length). Reef structural complexity was assessed using Maximum Reef Relief (MRR), measured every 5 meters along each transect. The data was collected from five dive sites: Pandora, Grandma's Garden, Manuel's Wall, Casablanca, and Buoy Line.

Analysis revealed a strong positive correlation (r = 0.95) between reef structural complexity (MRR) and fish abundance. Sites with higher MRR supported a greater number of fish per transect. According to Gratwicke and Speight (2005), "There is a strong, positive relationship between structural complexity and both species richness and abundance, emphasizing the importance of physical habitat complexity for maintaining fish communities."



↑ **Figure 1.** Relationship between Maximum Reef Relief (MRR) and Fish Abundance in Bocas del Toro, Panama.

The findings confirm the well-documented relationship between reef structural complexity and fish abundance. Higher MRR values provide more shelter and resources, leading to increased fish diversity and biomass. This highlights the importance of preserving structurally complex reef habitats for maintaining fish populations. (Hixon and Beets, 1993; Alvarez-Filip et al., 2009). Predation and the availability of refuge space jointly influence fish community structure, with complex habitats offering more opportunities for prey to escape predators. Higher MRR values reflect more shelter and resources, leading to increased fish diversity and biomass. This highlights the importance of preserving structurally complex reef habitats for maintaining fish populations. Future research should assess long-term trends and potential anthropogenic impacts on reef structure complexity in the region.

↑ **Figure 2.** Spongey Reef, Coral reef with predominance of sponges, April 2024. Location: Bocas del Toro, Panamá. Credit: Felipe Ossandón

↑ **Figure 3.** Bocas del Toro Reef, Coral reef with predominance of Agaricia, September 2024. Location: Bocas del Toro, Panamá. Credit: Paula Sills

CHAPTER 31

Puerto Rico

Authors: Eric Appeldoorn Sanders, Shay Viehman, Erica K. Towle, Jérémy Wicquart.

Data contributors: Eric Appeldoorn Sanders, Miguel Figuerola Hernández, Amit Hazra*, Alex Hunt*, Erica K. Towle, Maria Vega Rodriguez, Andrew W. Bruckner*.

*AGRRA data contributor(s).

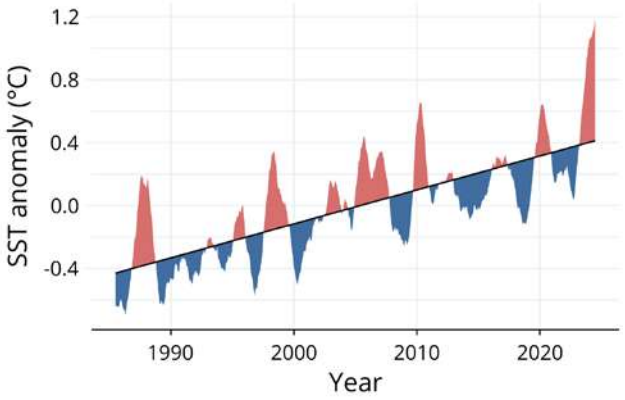
INTRODUCTION

Puerto Rico, an unincorporated territory of the United States located in Greater Antilles, lies east of the Dominican Republic and west of the U.S. Virgin Islands. The main island of Puerto Rico is surrounded by several smaller islands, including Vieques, Culebra, Mona, Monito, Desecheo, and Caja de Muertos (**Figure 2.31.4**). Coral reef extent is about 471 km², which represents nearly 2% of the Caribbean's total coral reef extent.

THREATS

Thermal regime

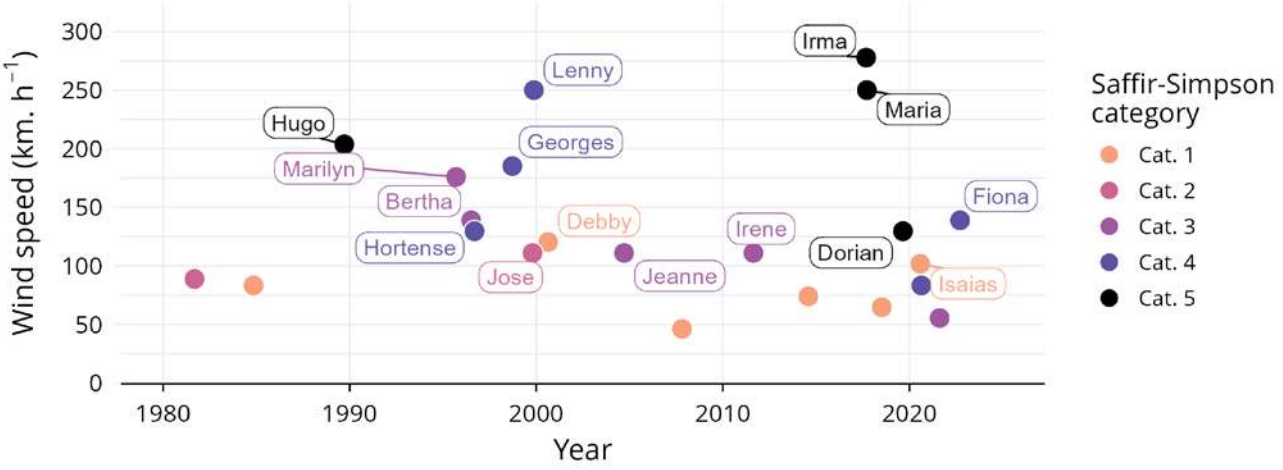
The long-term average in SST from 1985 to 2024 over coral reefs of Puerto Rico was 27.72 °C. Over this period, SST increased by 1.00 °C, representing a warming rate of 0.026 °C per year (**Figure 2.31.1**). Mass coral bleaching events occurred in 2005, 2019, and 2023-2024 (Garcias-Sais et al. 2023).



↑ **Figure 2.31.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Puerto Rico. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 33 tropical storms passed within 100 km distance from a coral reef of Puerto Rico, and of these 15 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.31.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed 51 km from a coral reef with sustained wind speed of 278 km.h^{-1}. Hurricanes Irma and Maria caused extensive coral breakage across shallow reefs (< 7m) around the Puerto Rico archipelago (Viehman et al. 2020).



↑ **Figure 2.31.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Puerto Rico. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Puerto Rico are not represented, although they can have had an impact.

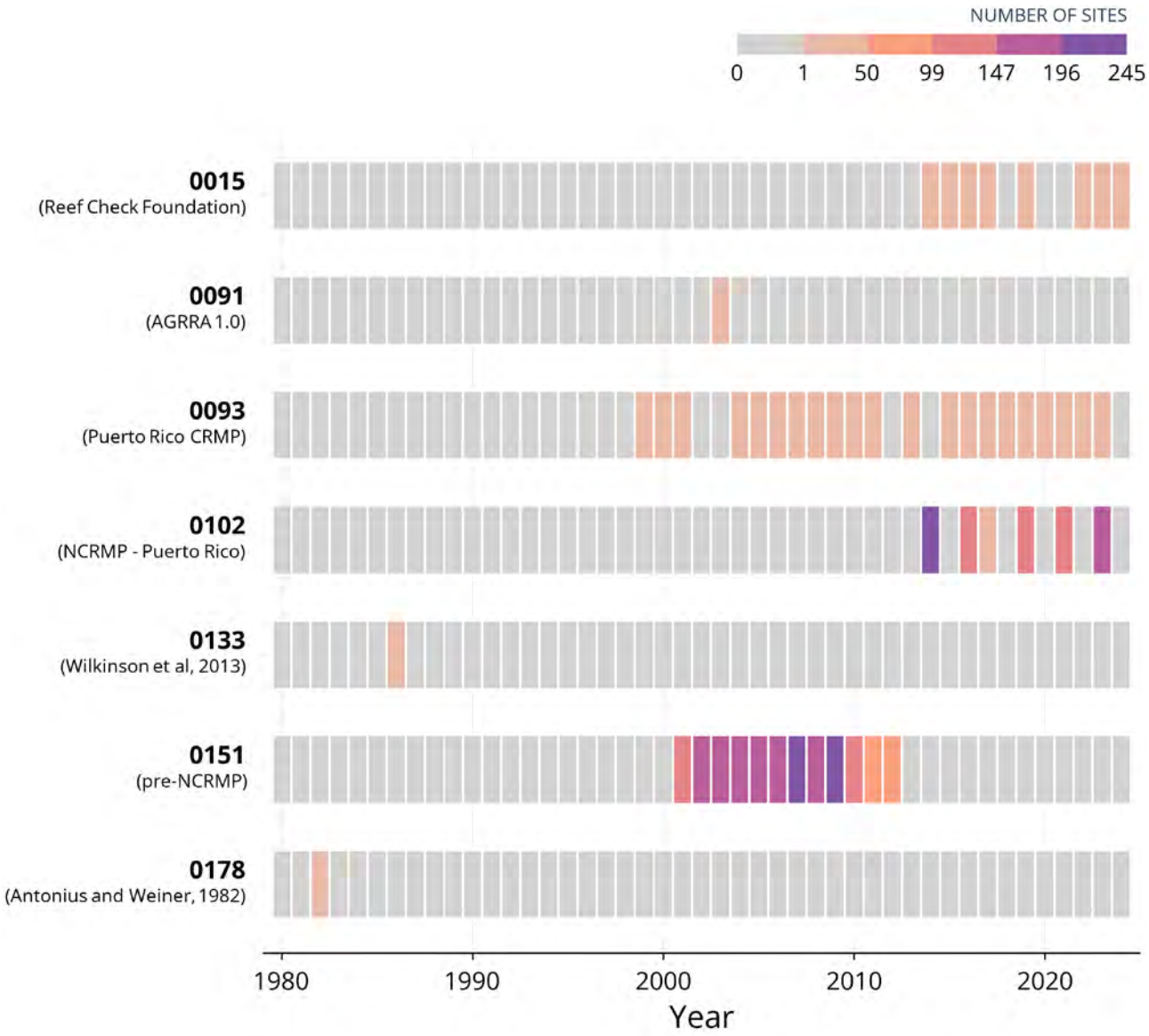
Coral diseases

Stony coral tissue loss disease (SCTLD) is the latest epizootic in a series that started with mass mortality of *Diadema antillarum* (1983-1984), white band disease affecting Acroporid corals (Gladfelter 1982) and other white diseases that affected massive species. SCTLD impacted many coral species, greatly reducing colony density, cover, and shifting population distributions of highly susceptible species to smaller sizes (Garcias-Sais et al. 2023, Viehman et al. 2025).

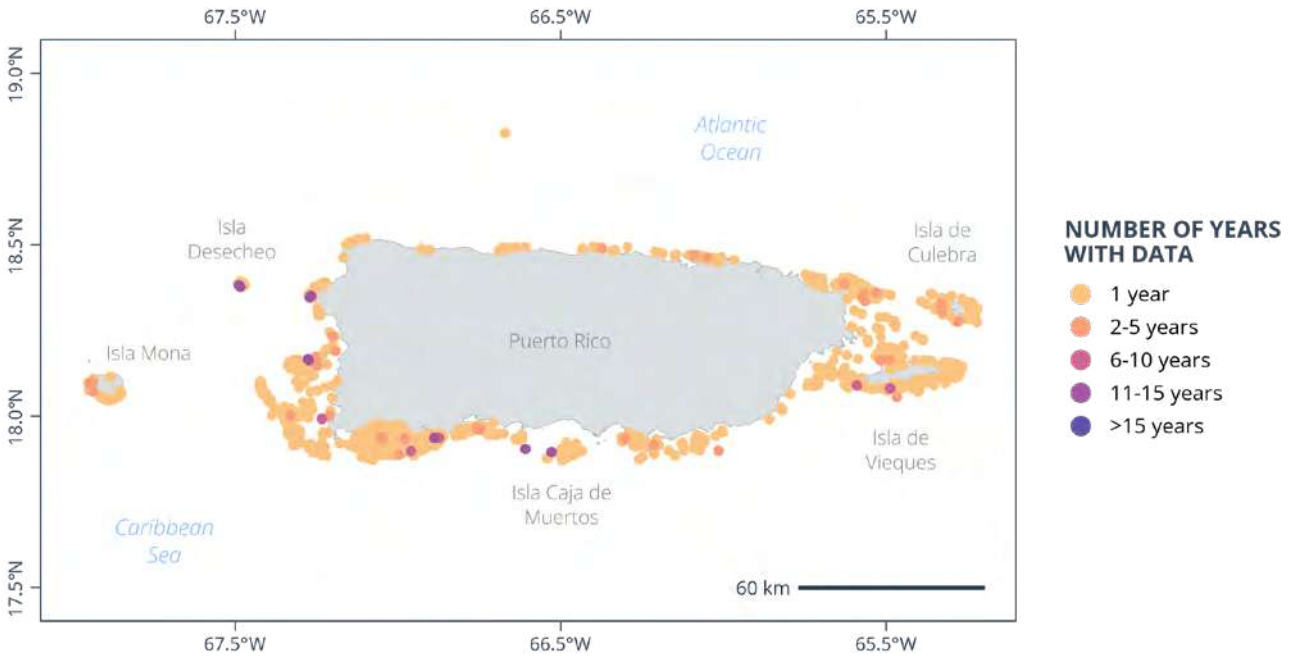
TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 7 datasets (**Figure 2.31.3**) encompassing 2,825 monitoring sites. These data were collected through 3,130 surveys conducted between 1982 and 2024, with a considerable increase in survey effort and spatial distribution after 2014.

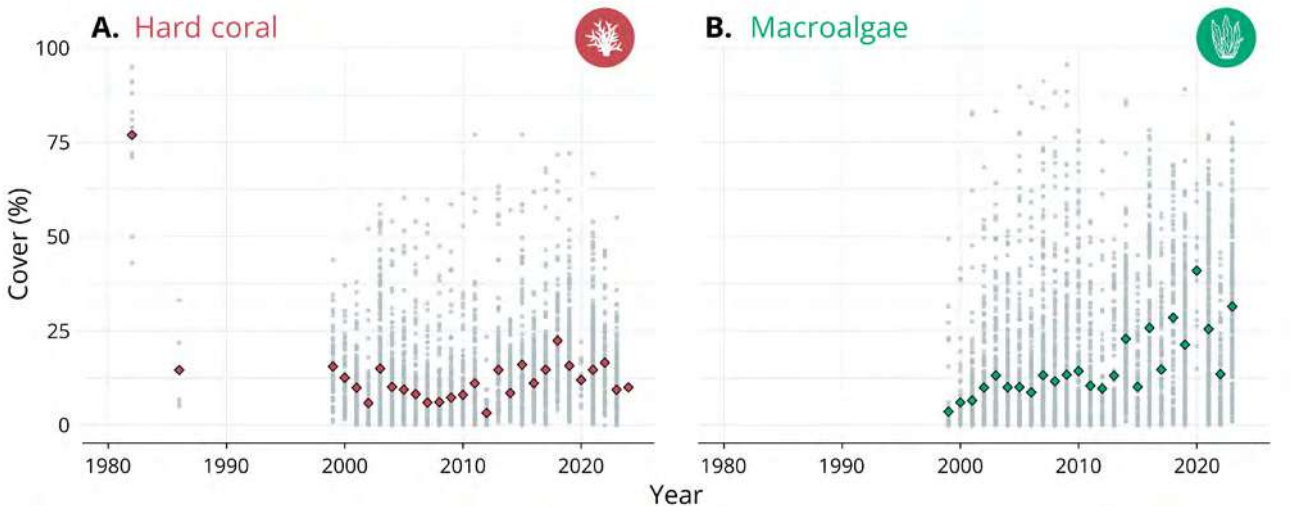
The temporal patterns of benthic cover of hard corals in Puerto Rico reflect a trajectory consistent with the well-documented decline of Caribbean coral reefs. Hard coral cover showed a sharp collapse during the 1980s, a period marked by the mass mortality of the herbivorous urchin Diadema antillarum (Lessios, 1988) and compounded by successive disturbances including hurricanes, disease outbreaks, and bleaching events (Gardner et al., 2003; Weil et al., 2009). Localized studies have documented similar declines linked to anthropogenic stressors such as sedimentation, pollution, and coastal development (Rogers, 1990; Hernández-Delgado, 2000; Ballantine et al., 2008). By the mid-1990s, coral cover had fallen below 15% and has remained low since, underscoring the poor recovery ability of Puerto Rican reefs and mirroring broader Caribbean trends where cumulative pressures and recurrent impacts have hindered coral resurgence (Hernández-Delgado et al., 2014). The sharp decline observed after 2005 is likely attributable to the region-wide mass bleaching event that year, which caused unprecedented coral mortality in Puerto Rico and across the Caribbean (Eakin et al., 2010; Hernández-Delgado et al., 2014). In contrast, macroalgae cover exhibited an inverse trajectory, steadily increasing over the same period and exceeding 20% over since 2015. The high percentage cover of macroalgae emerged through weakened grazing pressure and reduced competition from corals, reflecting a classic phase shift commonly reported in Caribbean reefs (Hughes, 1994; Mumby et al., 2007). Studies have indicated that algal expansion has been reinforced by low Diadema recovery and continued fishing pressure on herbivorous fishes, further limiting coral resilience (Ballantine et al., 2008; Hernández-Delgado et al., 2014).



↑ **Figure 2.31.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.31.4.** Spatio-temporal distribution of benthic cover monitoring sites across Puerto Rico. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.31.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Puerto Rico from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

Socioeconomic Trends in the U.S. Caribbean

Authors: Mary Allen[1]

[1] Lynker contract in support of NOAA Coral Reef Conservation Program; Silver Spring, Maryland, USA

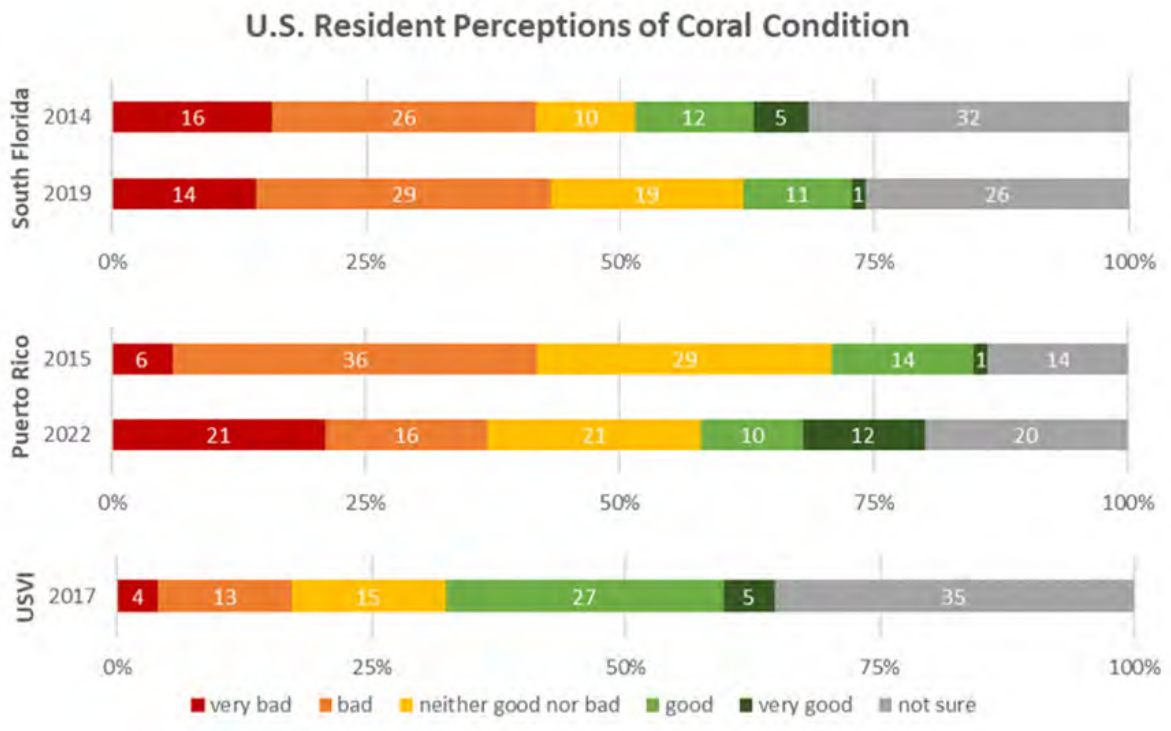
Socioeconomic monitoring is critical to understanding the connections between humans and coral reef ecosystems. This is important because coral reefs provide a variety of benefits to people such as recreation, tourism, seafood, cultural heritage, and much more. Likewise, people play a central role in protecting coral reef ecosystems to sustain these benefits, while fostering their relationships with nature. Recognizing that people are an integral part of the ecosystem, NOAA's National Coral Reef Monitoring Program (NCRMP) is one of the few programs in the world that integrates human dimensions into coral reef ecosystem monitoring. As a whole, NCRMP provides a holistic understanding of the status of coral reefs and the coastal communities who depend on these ecosystems.

The Socioeconomic Component of NCRMP collects and monitors socioeconomic data in seven U.S. coral reef states and territories, including three areas in the Caribbean: Florida (Allen et al., 2019), Puerto Rico (Allen et al., 2024), and the U.S. Virgin Islands (USVI) (Gorstein et al., 2017).

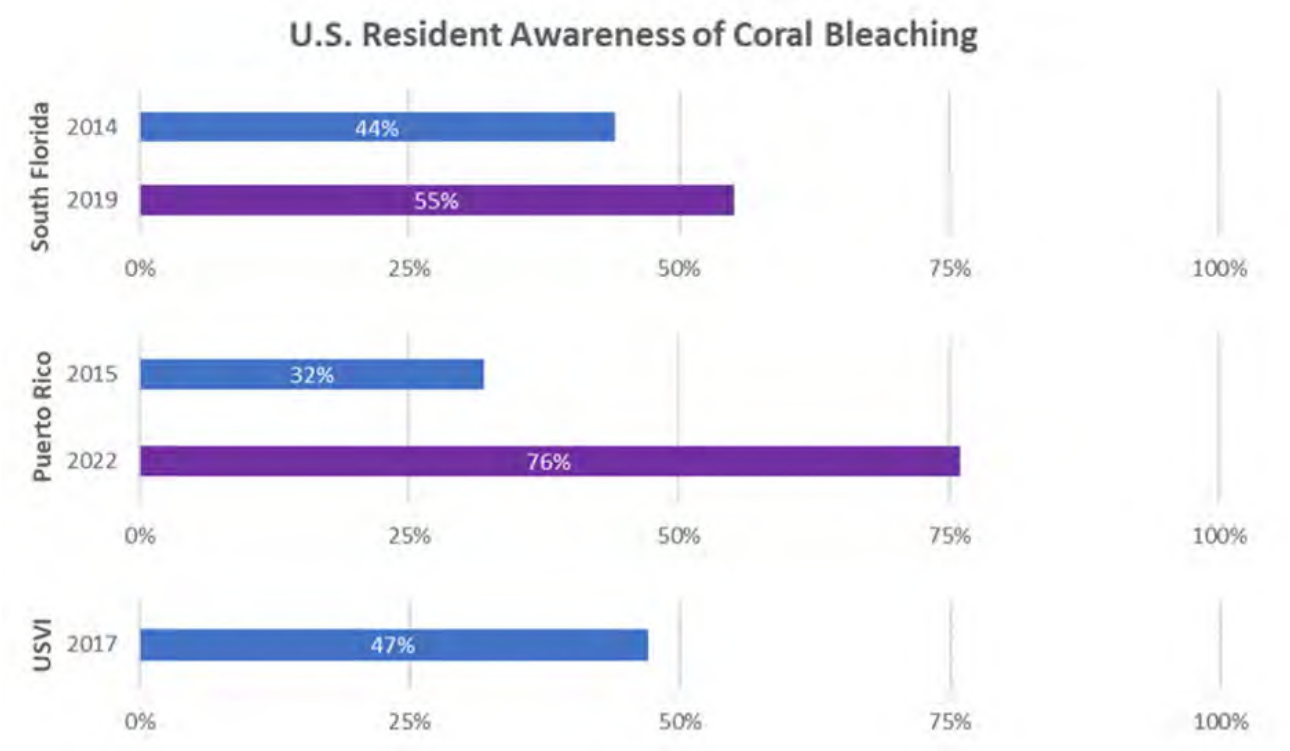
To assess the status of socioeconomic conditions related to coral reefs, surveys are conducted with household residents every 5-7 years. The surveys collect data on measures such as human use of coral reef resources, knowledge, attitudes, and perceptions of coral reefs and management. These data inform the human connections to coral reefs and how their interactions with coral reefs may be changing over time.

In the U.S. Caribbean, two socioeconomic monitoring cycles have been completed. In South Florida, surveys were completed in 2014 and 2019; in Puerto Rico, surveys were completed in 2015 and 2022; and in USVI, surveys were completed in 2017 and 2025 (data not available at time of this report). The data are representative of each U.S. Caribbean state and territory's resident population, and offer interesting insight from residents' perspectives on coral reef conditions throughout the Caribbean.

In general, U.S. Caribbean residents have rated the amount of live coral as being in "bad" condition (**Figure 1**).



↑ **Figure 1.** U.S. resident perceptions of coral condition in South Florida, Puerto Rico, and the U.S. Virgin Islands from surveys conducted in 2014-2022.



↑ **Figure 2.** U.S. resident awareness of coral bleaching in the Caribbean; Blue=first monitoring cycle, Purple=second monitoring cycle.

On a 5-point scale (1=very bad, 5=very good), residents of South Florida perceived corals as being in the worst condition (mean=2.5 in 2014, mean=2.4 in 2019) relative to ratings from other U.S. Caribbean areas. Residents of Puerto Rico also perceived corals as being in "bad" condition, with a mean rating of 2.7 in both 2015 and 2022. Residents of USVI had relatively more positive perceptions of coral condition, with a mean rating of 3.3 in 2017; however, in 2017, 35% of USVI residents were "not sure" about the condition of coral. The results from the USVI 2025 survey are still pending, so any change in perceptions can not yet be determined. However, changing patterns are being detected elsewhere. In South Florida, residents became more certain about coral condition in 2019 (decrease in "not sure" ratings); and in Puerto Rico, residents were less certain about coral condition in 2022 (increase in "not sure" ratings). While these changes in perceptions reflect different time periods, the results may be considered with corresponding biophysical conditions such as coral bleaching or extreme heat events that have been recorded.

In 2014-2017, less than 50% of residents across the U.S. Caribbean were familiar with coral bleaching as a threat to the ecosystem (**Figure 2**). Results from the second surveys in 2019 and 2022 indicated an increase in awareness of coral bleaching among residents in South Florida and Puerto Rico. This increasing trend in awareness may be influenced by exposure to the increasing frequency of coral bleaching events in the region, outreach and communication about this issue, or other factors.

The findings on perceptions of resource conditions and threats to coral reefs, in addition to the other NCRMP socioeconomic results, can be useful for a) assessing public perceptions regarding the relative degree of success of current management efforts and as a means for indicating how such efforts might be adjusted to accommodate changing conditions; b) designing new management approaches that are readily understood and therefore more likely to be accepted and followed by resource users; and c) adjusting outreach and educational efforts per changing local observations about threats to the local marine environment.

CHAPTER 32

Saba

Authors: Alwin Hylkema, Mareike de Breuyn, Bert W. Hoeksema, Laëtitia Mathon.

Data contributors: Kristi D. Klomp*, Mareike de Breuyn, Alice Deschamps*, Julie Hawkins, Alwin Hylkema, Walter Hynds, Ayumi Izioka, Oliver Klokman, Marnik Lehwald, Chris Moses*, Callum Roberts, Sascha Steiner*, Marijn Van Der Laan, Alex Van Der Last.

*AGRRA data contributor(s).

INTRODUCTION

Saba, a small 13 km² Caribbean island within the Kingdom of the Netherlands, is home to approximately 2,100 people (2024) and is surrounded by diverse marine ecosystems. Along with Saba Bank, Saba's waters host 203 km² of coral reefs, representing 0.84% of the Caribbean's total reef area.

Before 2010, Saba was part of the Netherlands Antilles, a country within the kingdom of the Netherlands, also including Bonaire, St. Eustatius, Curaçao, and St. Maarten. After the dissolution of the Netherlands Antilles, Curaçao and St. Maarten continued as separate countries within the kingdom of the Netherlands, together with Aruba, while the smaller islands of Bonaire, Saba, and St. Eustatius continued as special municipalities of the Netherlands and are referred to as Caribbean Netherlands.

Saba is a very steep volcanic island, harboring terrestrial ecosystems ranging from deciduous dry forest to cloud forest at the top of Mount Scenery. Saba has no permanent beaches and, as a result, very little coastal development. The four villages are all more than 200 meters and up to 400 meters above sea level.

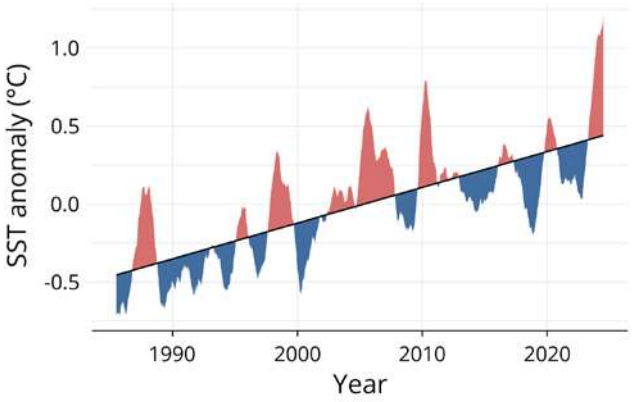
Coral on Saba grows on lava outcrops and fingers or, in only a few places, on limestone. It is likely that frequent hurricanes and possibly volcanic activity constrained the formation of extensive calcium carbonate reefs.

The Saba National Marine Park (8 km²) was established in 1987 to protect habitats around the island down to 60 m depth. To the southwest lies Saba Bank, a vast 2,200 km² submerged atoll with a high diversity of fish, crustaceans and free-living corals (Hoeksema et al., 2017). and a major lobster and red snapper fishery.

THREATS

Thermal regime

The long-term average SST over Saba's coral reefs from 1985 to 2024 was 27.73 °C. Over this period, SST increased by 1.05 °C, representing a warming rate of 0.027 °C per year (**Figure 2.32.1**).

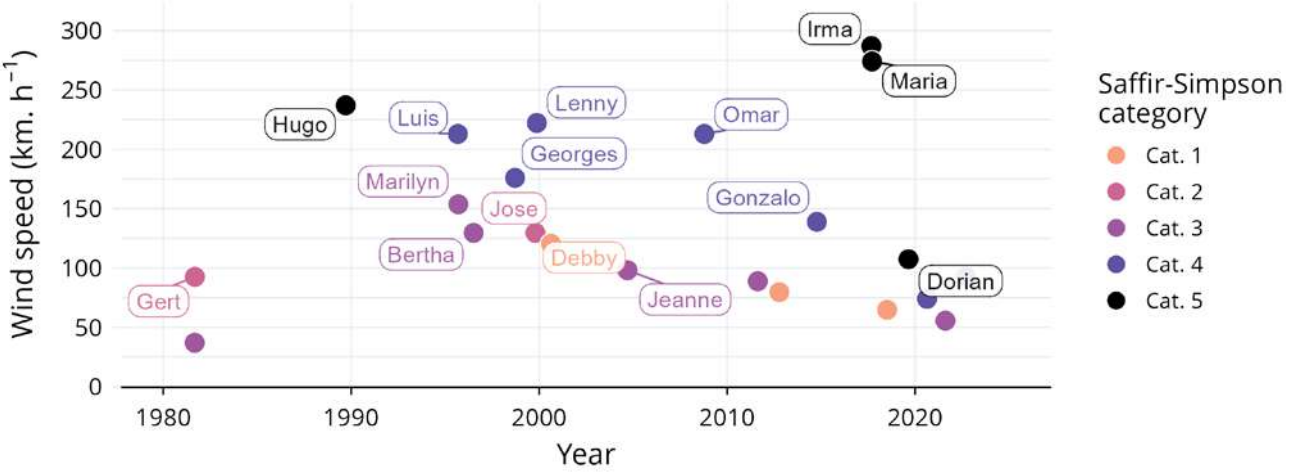


↑ **Figure 2.32.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Saba. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 33 tropical storms passed within 100 km distance from a coral reef of Saba, and of these 13 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.32.2**). The hurricane with the highest sustained wind speed over the studied period was Hurricane Irma, in 2017, which passed 58 km from Saba's coral reefs with sustained wind speed of 287 km.h^{-1}.

↓ **Figure 2.32.2.** Maximum sustained wind speed of tropical storms passing within 100 km of Saba between 1980 and 2024. Colors correspond to the cyclone's maximum Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Saba are not represented, although they may still have had an impact.



Algae overgrowth

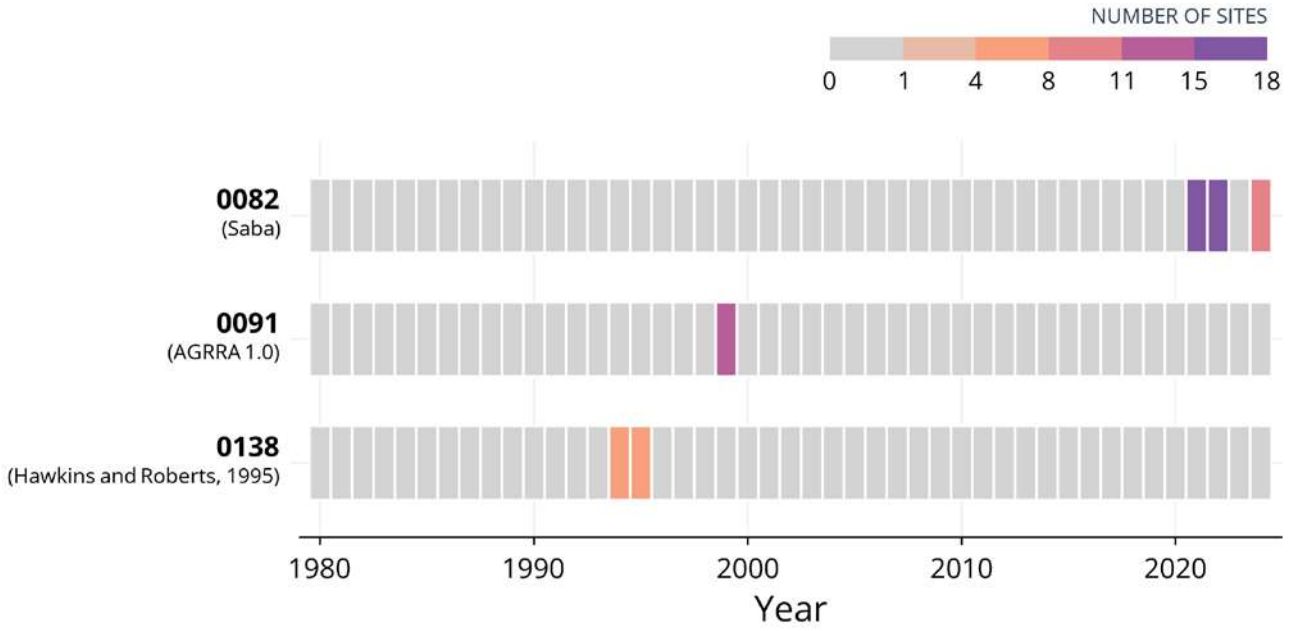
Coral reefs around Saba face multiple threats from both marine and land-based activities. Apart from climate change-induced bleaching events and hurricanes, coral disease has been the most significant threat to Saba's reefs. The most recent outbreak is Stony Coral Tissue Loss Disease (SCTLD), first recorded near the island in 2021. Benthic algal cover around Saba is very high (>60%), severely limiting coral recruitment. This high algal abundance likely results from reduced grazing pressure, eutrophication, and increased substrate availability due to decreases in hard coral cover.

Nutrient enrichment further stimulates algal growth. Apart from regional basin wide eutrophication, Saba's steep slopes, combined with construction activities, result in substantial terrestrial run-off. Long-term overgrazing by thousands of feral goats exacerbated this problem, although the goats were recently removed through a government-led project. Wastewater is managed at the household level using cesspits. Given the island's small population and the distance of most cesspits from the shoreline, this is considered a minor nutrient source. Passing cruise ships, which transit close to Saba at night without stopping, are likely a more significant contributor to eutrophication (Meesters et al. 2025).

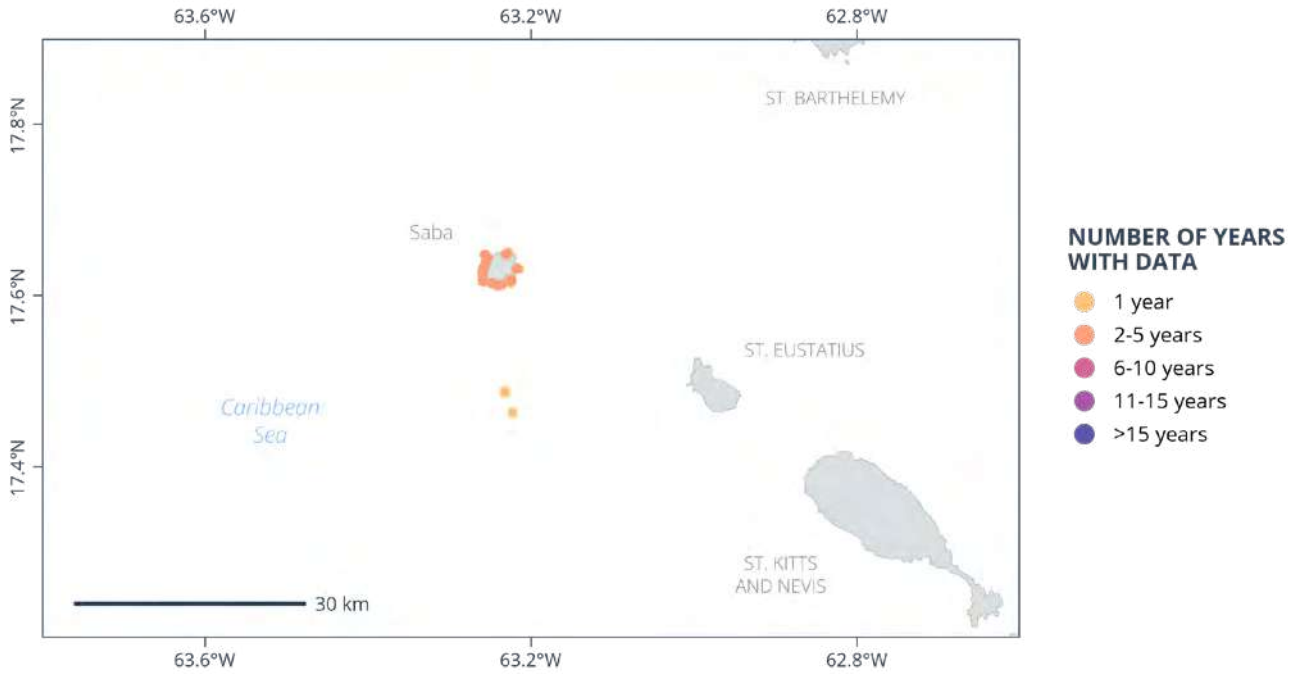
In addition, poor land management on Saba, including dumping of removed soil and construction waste, contributes to erosion and sedimentation, which smothers coral reefs and degrades water quality, further stressing these sensitive ecosystems (Bouchon et al., 2008).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.32.3**) encompassing 30 monitoring sites (**Figure 2.32.4**). These data were collected through 68 surveys conducted between 1994 and 2024.



↑ **Figure 2.32.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.32.4.** Spatio-temporal distribution of benthic cover monitoring sites across Saba. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Saba's coral reefs have undergone a stepwise degradation over the past decades. The datasets used for this report show a clear decline in coral cover from around 20% in the early 1990s till 2.15% in 2024.

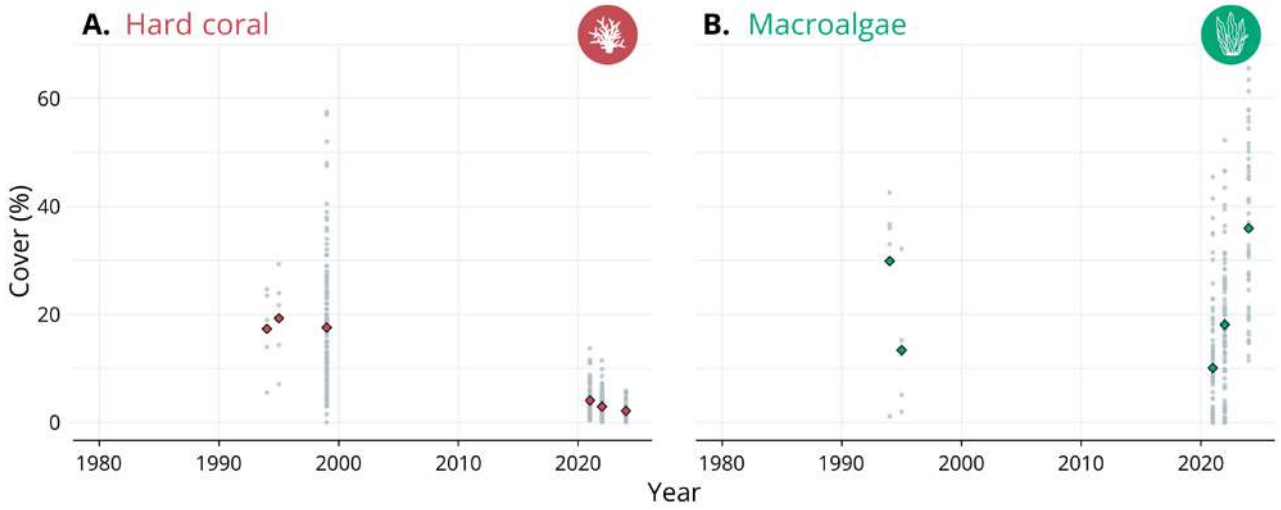
Roberts (1995) estimated coral cover around Saba to be 10-18% at 5 m depth and 25-35% at 10 m depth in 1991 and 1993, which is in line with the data used for this report. Climate change induced heatwaves, hurricanes and coral diseases decimated coral populations in the years that followed. Noble et al. (2013) re-estimated the coral

cover in 2008 on four locations around the island and concluded that coral cover declined by 38% compared to the early 90s, with percentage cover around 15% at 5m depth and 7% at 10m depth. Meesters et al. (2025) showed that coral continued to decline in the 2010s.

The decline of corals opened up more space for algae on the seabed. In the early 1990s, total algae cover was already above 50% and increased further till above 60% in the early 2020s. Grazing by herbivorous fishes was clearly not enough to prevent algal dominance. Another important grazer, the long-spined sea urchin *Diadema antillarum*, was almost entirely eliminated by disease in 1983–1984 (Lessios et al. 1984) and again in 2022 (Hylkema et al. 2023). Although settlement of this keystone herbivore is relatively high and comparable to pre-die-off densities (Klokman and Hylkema 2024), high post-settlement predation constrains population recovery (Hylkema et al. 2022). Herbivorous fishes are not actively targeted by fishers, but have been caught as bycatch in lobster fisheries, which operated around Saba until 2013. Since then, commercial fishing has been restricted to the Saba Bank, with no lobster traps deployed on the island's reefs. Recreational fishing continues, mainly hook-and-line from shore and spearfishing, neither of which target herbivores directly. However, removal of large predators such as groupers and snappers may exert indirect trophic effects through mesopredator release.

In addition, recruitment of herbivorous fish might be reduced by the limited availability of fish nursery habitats: mangroves and shallow seagrass beds are absent, and shallow coral reefs are scarce.

It is important to note that local threats to Saba's reefs are relatively minor compared to nearby islands such as St. Maarten, which experiences higher human population density, more intensive fishing, greater run-off, and extensive coastal development. Based on these differences, Saba's reefs would be expected to remain comparatively healthy. However, this has not been observed. Instead, regional and global stressors, including marine heatwaves and coral diseases, appear to exert a stronger influence on reef condition than local pressures in this part of the Caribbean. Research is being conducted on how algae cover on Saba's reefs can be reduced by culturing and enhancing invertebrate herbivore densities.



↑ **Figure 2.32.5.** Annual means of observed benthic percentage cover data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Saba from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 33

Saint Kitts and Nevis

Authors: Michelle M. Dennis, Laëtitia Mathon.

Data contributors: Jeanne Brown*, Andrew W. Bruckner*, Brooke Gintert*, Danny Green*, Clare Morrall*, Amanda Williams*.

*AGRRA data contributor(s).

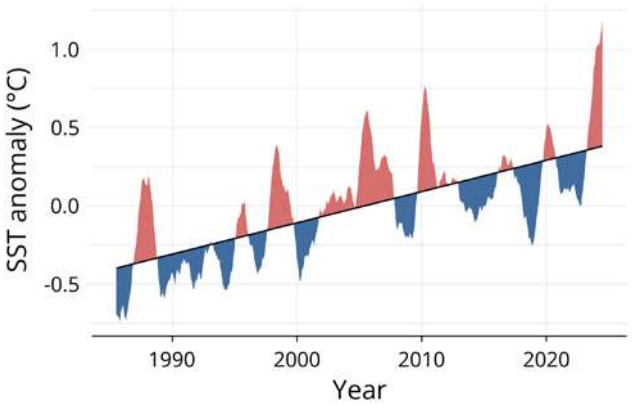
INTRODUCTION

Saint Kitts and Nevis, the smallest sovereign state in the Western Hemisphere with a land area of just 261 km² and a 2024 population of 46,840, is surrounded by an Exclusive Economic Zone (EEZ) spanning 20,400 km². Within these waters, the country hosts approximately 82 km² of coral reefs—representing 0.338% of the Caribbean's total reef extent—that support both biodiversity and livelihoods. The reefs are home to 29 coral species, with most sites showing 6–13% live coral cover, and they sustain 88 species of ecologically and commercially important fishes (Bruckner and Williams, 2012). These ecosystems underpin a coastal economy where tourism is the primary driver, attracting private yachts, cruise ships, and a range of water-based activities, while commercial and artisanal fisheries also remain vital, employing diverse vessels, gear types, and strategies across much of the coastal waters. A zoning plan was built in concertation with stakeholders for a sustainable use of the sea (Agostini et al., 2015).

THREATS

Thermal regime

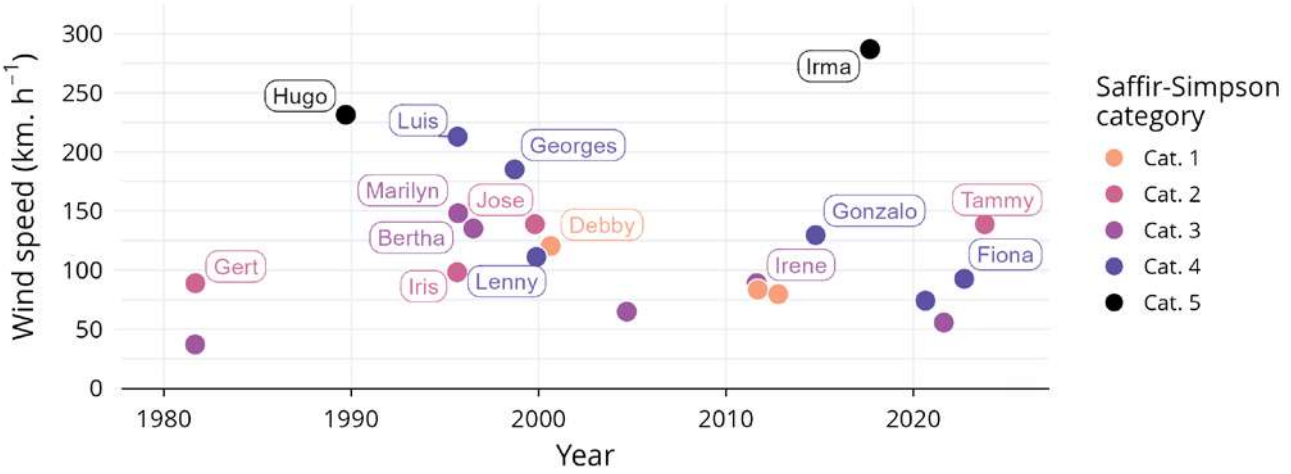
The long-term average in SST from 1985 to 2024 over coral reefs of Saint Kitts and Nevis was 27.61 °C. Over this period, SST increased by 0.94 °C, representing a warming rate of 0.024 °C per year (**Figure 2.33.1**).



↑ **Figure 2.33.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Saint Kitts and Nevis. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 30 tropical storms passed within 100 km distance from a coral reef of Saint Kitts and Nevis, and of these 11 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.33.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed 53 km from a coral reef with sustained wind speed of 287 km.h^{-1}. Hurricane Maria, only 10 days after Irma, passed near Saint Kitts and Nevis, with worse swells on the reefs along the Southern coastline.



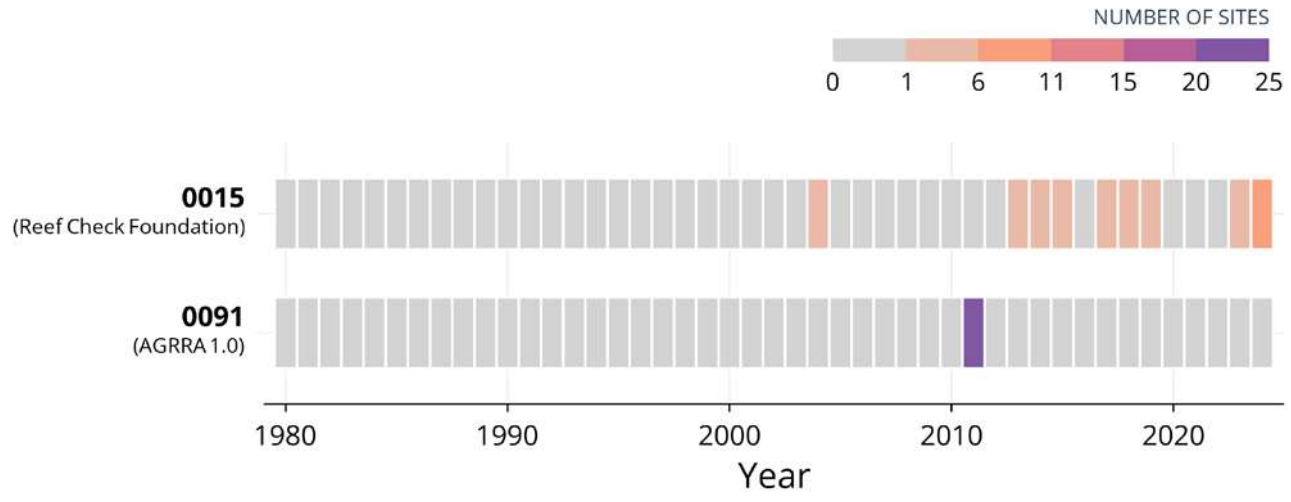
↑ **Figure 2.33.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Saint Kitts and Nevis. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Saint Kitts and Nevis are not represented, although they can have had an impact.

Other threats

The coral reefs of Saint Kitts and Nevis face mounting pressures from both human and environmental stressors. Nearshore fisheries have declined sharply in recent years, with fishers reporting reduced catches of conch, lobster, and large pelagic and demersal fishes, reflecting broader concerns about unsustainable fishing practices. Coastal development, land-based sources of pollution, and physical damage from vessel anchoring have further degraded reef habitats, while rising ocean temperatures and increasingly intense hurricanes amplify the impacts of storm damage (Bruckner, 2012). Disease and coral health decline add another layer of concern: surveys conducted in 2017 documented eight predominant lesion patterns affecting 59% of corals, specifically from the *Orbicella* complex, ranging from pigmentation anomalies and bleaching to tissue loss and skeletal erosion (Dorrestein et al., 2020).

TRENDS IN BENTHIC COVER

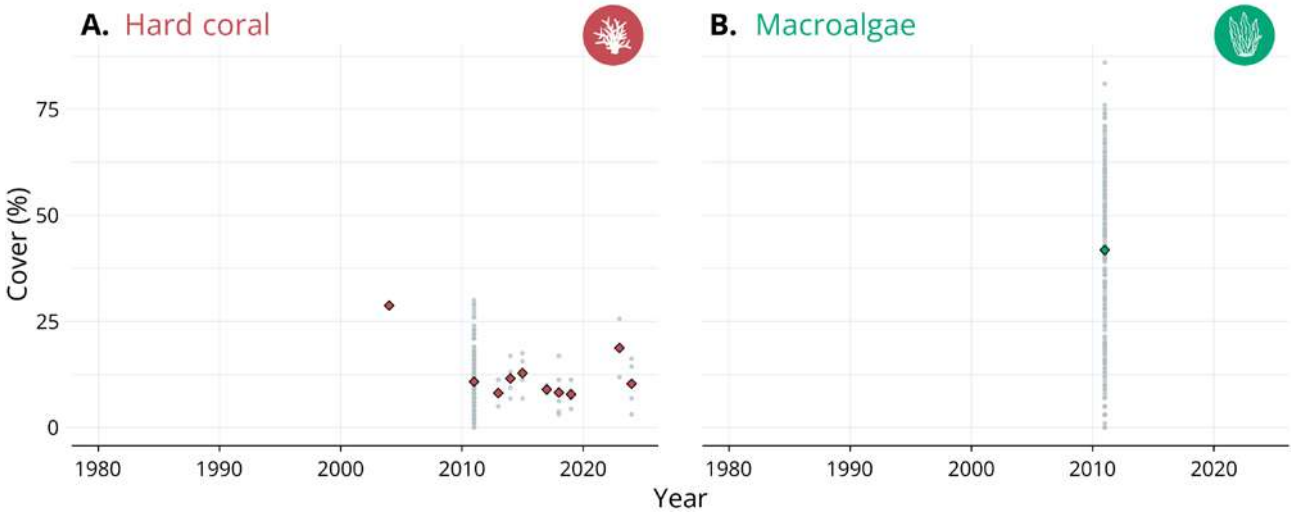
Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.33.3**) encompassing 38 monitoring sites. These data were collected through 55 surveys conducted between 2004 and 2024.



↑ **Figure 2.33.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

↑ **Figure 2.33.4.** Spatio-temporal distribution of benthic cover monitoring sites across Saint Kitts and Nevis. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Coral reef health in Saint Kitts and Nevis has declined over the past few decades due to repeated bleaching events and shifts in species composition. The 1997–1998 bleaching caused about 50% coral mortality (Alcolado, 2003), and another severe event in 2005 led to widespread loss of *Orbicella annularis* complex corals (Bruckner, 2012). While the sea urchin *Diadema antillarum* was still abundant in shallow reefs in 2003, limiting algal growth, deeper reefs showed substantial algal proliferation due to the scarcity of grazing fishes such as parrotfish and doctorfish (Alcolado, 2003). By 2010, disease prevalence was relatively low, likely because many susceptible colonies had already died from the bleaching event (Bruckner, 2012). Surveys in 2012 reported live coral cover below 15% in most areas, with *Orbicella* spp. accounting for about 18% of corals (Bruckner and Williams, 2012). Reefs were strongly impacted by the emergence of Stony Coral Tissue Loss Disease and the second *Diadema* urchin mass mortality in 2022.



↑ **Figure 2.33.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Saint Kitts and Nevis from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

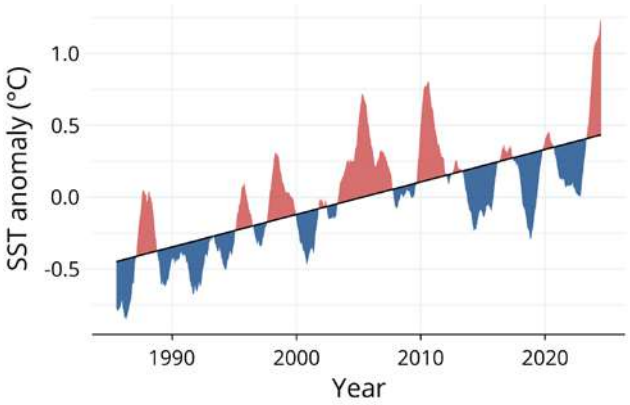
CHAPTER 34

Saint Lucia

Authors: Laëtitia Mathon.

INTRODUCTION

Saint Lucia, a volcanic island in the eastern Caribbean with a land area of 616 km² and a population of approximately 180,000 in 2024, supports an estimated 61 km² of coral reefs, representing 0.25% of the total reef extent in the Caribbean. To enhance reef conservation and sustainable use, the Soufrière Marine Management Area (SMMA) was established on the west coast in 1995, followed by the Canaries-Anse-La-Raye Marine Management Area (CAMMA) in 2001. These ecosystems play a critical ecological role while also underpinning the island's economy, as tourism and cruise visitation generated over 55% of national GDP in 2019, with more than 600,000 cruise arrivals recorded that year (Ellis, 2021).



↑ **Figure 2.34.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Saint Lucia. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

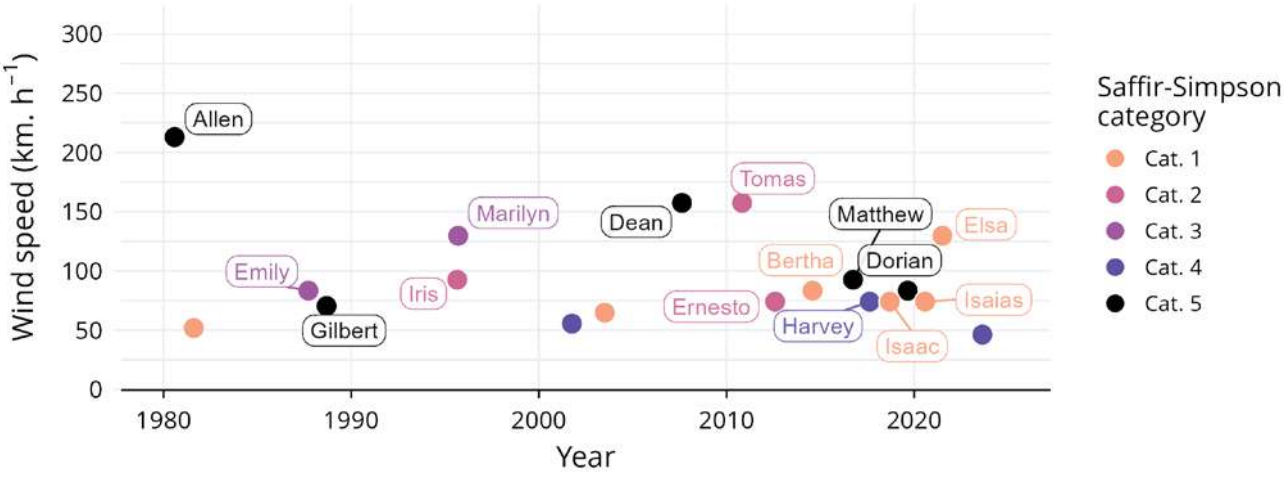
THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Saint Lucia was 27.75 °C. Over this period, SST increased by 1.04 °C, representing a warming rate of 0.027 °C per year (**Figure 2.34.1**).

Hurricanes

Between 1980 and 2024, a total of 31 tropical storms passed within 100 km distance from a coral reef of Saint Lucia, and of these 8 were characterized by sustained wind speed greater than 100 $km.h^{-1}$ (**Figure 2.34.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Allen, in 1980, which passed 13 km from a coral reef with sustained wind speed of 213 $km.h^{-1}$.



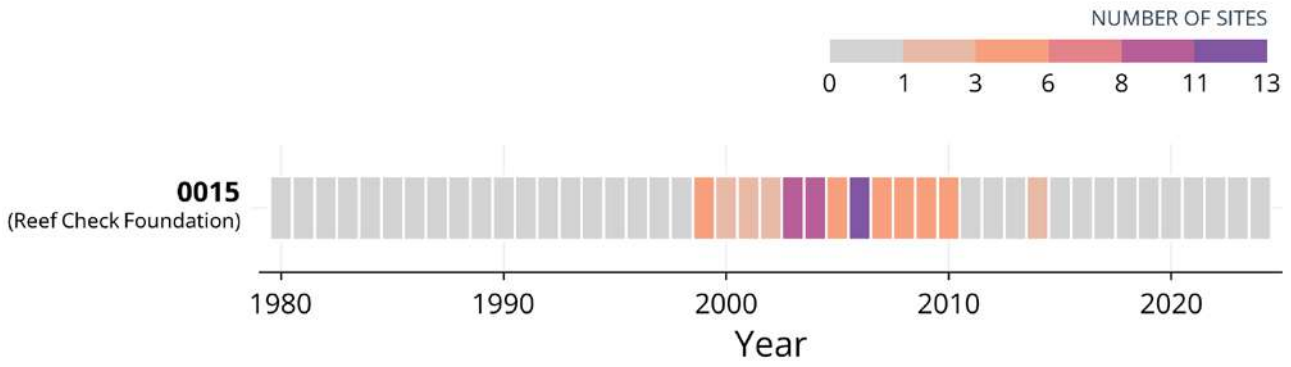
↑ **Figure 2.34.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Saint Lucia. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Saint Lucia are not represented, although they can have had an impact.

Other threats

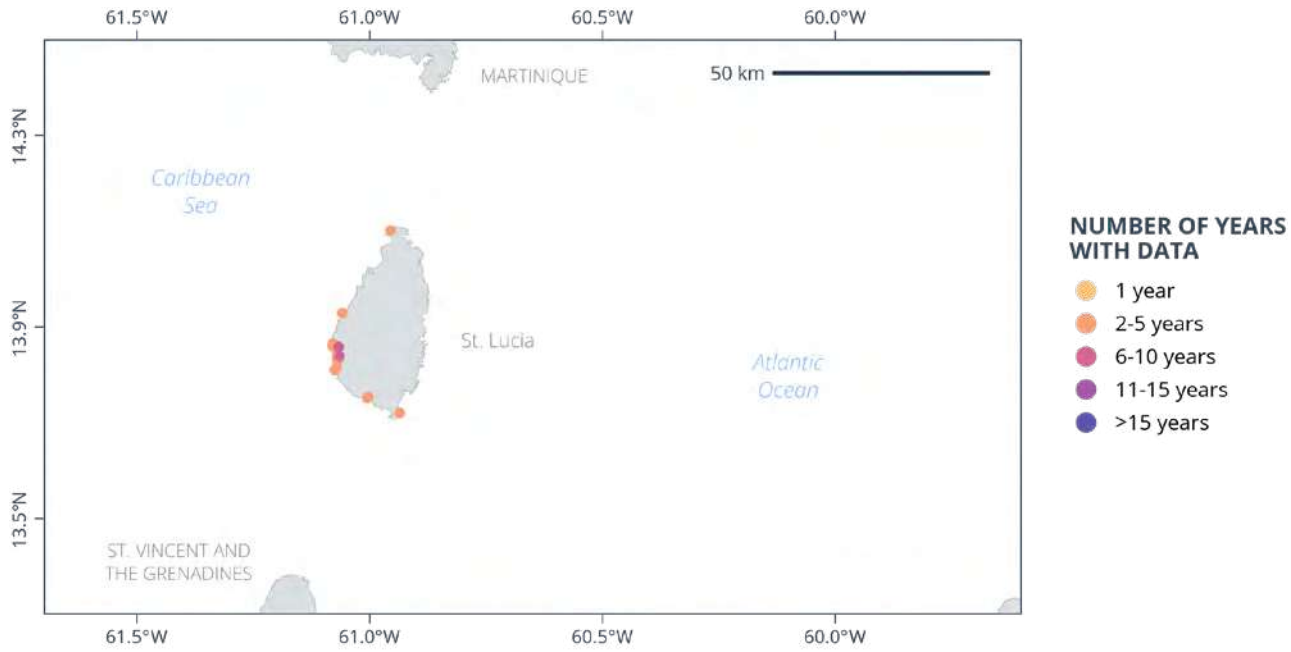
Coral reefs in Saint Lucia face multiple anthropogenic threats, primarily linked to land-use changes and coastal development. Sediment core analyses indicate a two- to three-fold increase in both terrigenous and calcareous sediment accumulation over the past three to four decades, with model estimates attributing 83–95% of sediment yield in affected watersheds to unpaved and degraded roads (Bégin et al., 2014). Elevated sedimentation reduces water clarity, smothers coral colonies, and impairs reef recovery. In addition, rapid growth in the cruise tourism sector has intensified pressures on nearshore ecosystems. Port expansion to accommodate up to six large vessels simultaneously required dredging and berth extension, activities that significantly increased sediment loads and contributed to the degradation and loss of marine flora and fauna (Ellis, 2021).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 1 dataset (**Figure 2.34.3**) encompassing 21 monitoring sites (**Figure 2.34.4**). These data were collected through 61 surveys conducted between 1999 and 2014.

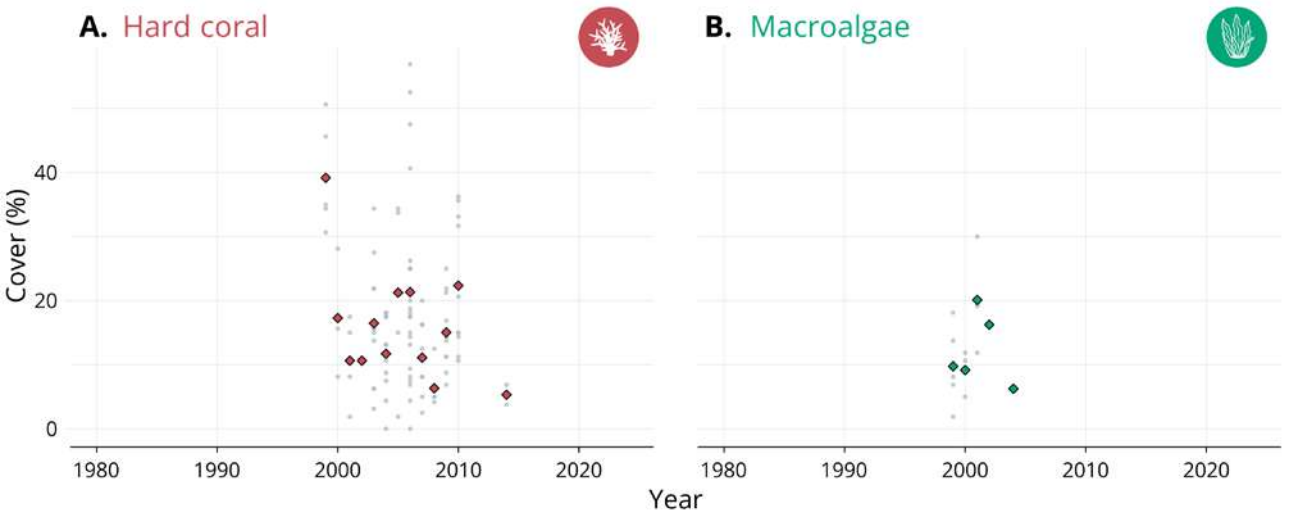


↑ **Figure 2.34.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.34.4.** Spatio-temporal distribution of benthic cover monitoring sites across Saint Lucia. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

The trajectory of coral reef condition in Saint Lucia reflects a pattern of initial fishery benefits from marine reserves, followed by persistent coral decline driven by multiple stressors. The Soufrière Marine Management Area (SMMA), established in 1995, quickly demonstrated ecological gains for fish populations, with total biomass quadrupling inside reserves and tripling outside within six years, largely due to increases in herbivores (Burke et al., 2008; Roberts et al., 2001; Hawkins et al., 2006). However, these benefits did not extend to reef-building corals, which declined by 35–46% between 1995 and 2001. This decline was linked to a combination of disease outbreaks (Nugues, 2002), mass bleaching in 1998, hurricane damage in 1999, and chronic sedimentation stress from river discharge (Nowlis et al., 1997; Nugues and Roberts, 2003). Monitoring in 2011, 10–16 years after the creation of MPAs, revealed further losses: mean coral cover had decreased by an additional 7–12% since 2001, while turf algae declined by 10–26% and macroalgal cover increased by 5–9% (Bégin et al., 2016). Notably, protection status did not predict coral trends; rather, sites with higher terrigenous sediment deposition showed lower coral cover, greater coral declines, and increased macroalgal dominance, underscoring the overriding influence of land-based stressors on reef health.



↑ **Figure 2.34.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Saint Lucia from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 35

Saint Vincent and The Grenadines

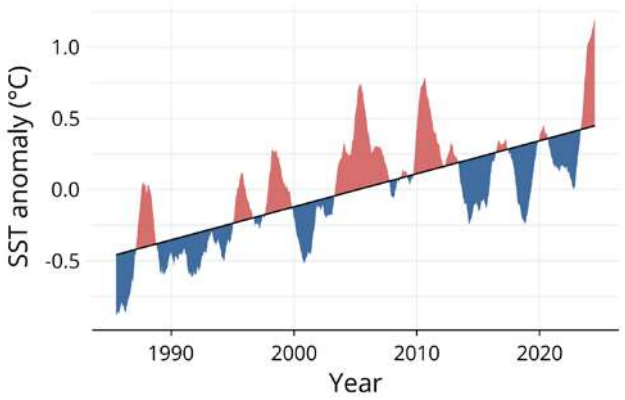
Authors: John Everett Parkinson, Chantale Bégin, Laëtitia Mathon.

Data contributors: Chantale Bégin, Jeanne Brown*, Alice Deschamps*, André Desrochers*, Jerry Enoe*, Jamie Kilgo*, Kristi D. Klomp*, Lindy C. Knowles*, Philip Kramer*, Judith Lang*, Sandra Little, Clare Morrall*, Werner Nagel*, John Everett Parkinson, Sophia Punnett*, Jeremy Searles*, Thomas Sweeney*, Sydney Tritschler, Sandy Voegeli*.

*AGRRA data contributor(s).

INTRODUCTION

Saint Vincent and the Grenadines is an archipelago of over 30 islands and cays on the Grenada Bank, which covers roughly 2,000 km². The country hosts a stable population (approximately 100,600 inhabitants in 2024) within a land area of 389 km² and an Exclusive Economic Zone (EEZ) of 27,733 km². The archipelago is home to 93 km² of coral reefs, accounting for 0.385% of the Caribbean's total reef area. Coral reefs in the region are critical for food security, livelihoods, and cultural activities, with fisheries providing income through shallow-shelf reef fishes, deepwater demersal species, lobsters, conchs, and pelagic resources (Edwards et al., 2018). Multiple marine protected areas, including Tobago Cays Marine Park, Bequia Marine Conservation Area, and others established since 1987, aim to conserve reef biodiversity and fisheries.



↑ **Figure 2.35.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Saint Vincent and the Grenadines. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

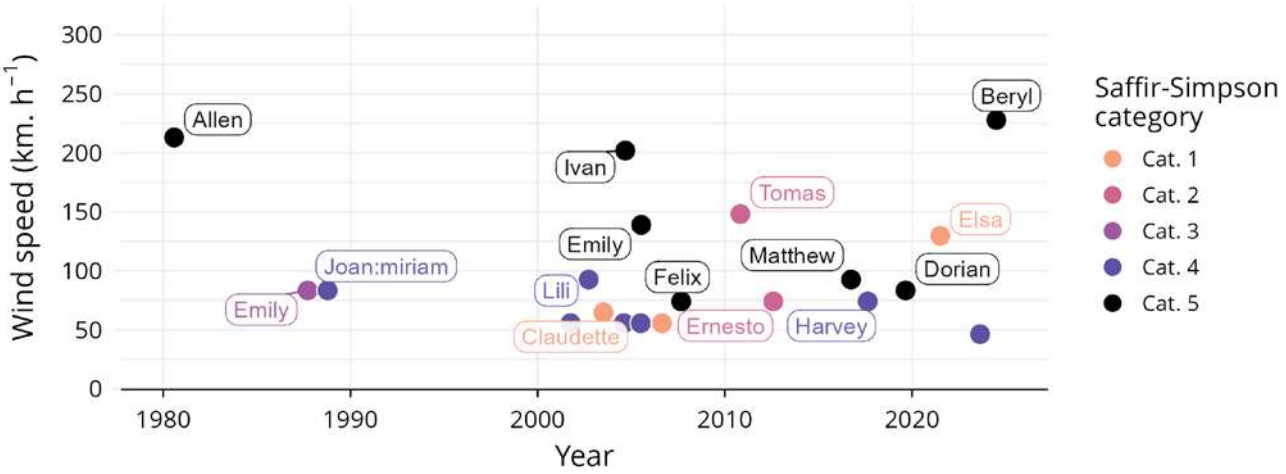
THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Saint Vincent and the Grenadines was 27.79 °C. Over this period, SST increased by 1.07 °C, representing a warming rate of 0.027 °C per year (**Figure 2.35.1**). Bleaching reports from the region are difficult to find, but a previous summary of ReefCheck data from the 2005 event indicated that up to 75% of colonies may have bleached at some locations (McField et al., 2008), while most others exhibited a more modest 0-20% of bleaching (reefcheck.org).

Hurricanes

Between 1980 and 2024, a total of 36 tropical storms passed within 100 km distance from a coral reef of Saint Vincent and the Grenadines, and of these 8 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.35.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Beryl, in 2024, which passed 13 km from a coral reef with sustained wind speed of 228 km.h^{-1}.



↑ **Figure 2.35.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Saint Vincent and the Grenadines. Colors correspond to the hurricane's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that hurricanes passing more than 100 km away from coral reefs of Saint Vincent and the Grenadines are not represented, although they may have had an impact.

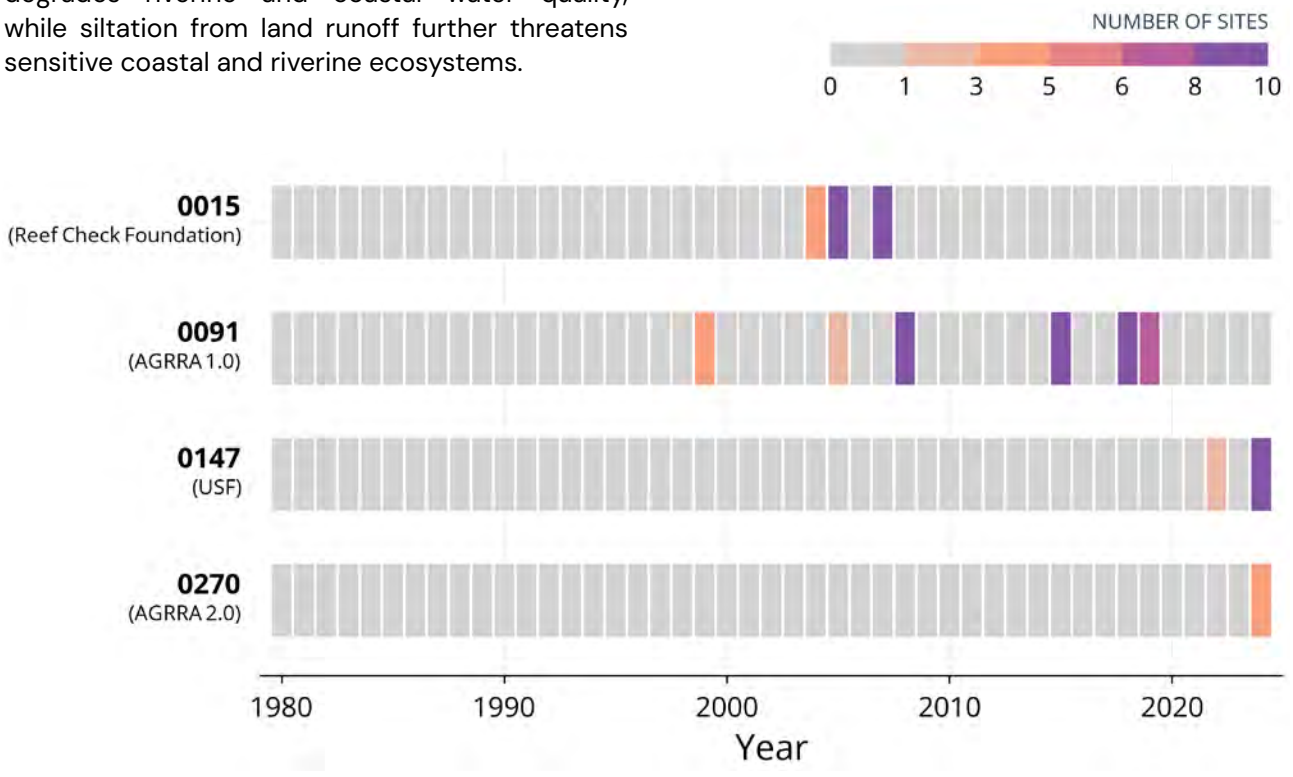
Other threats

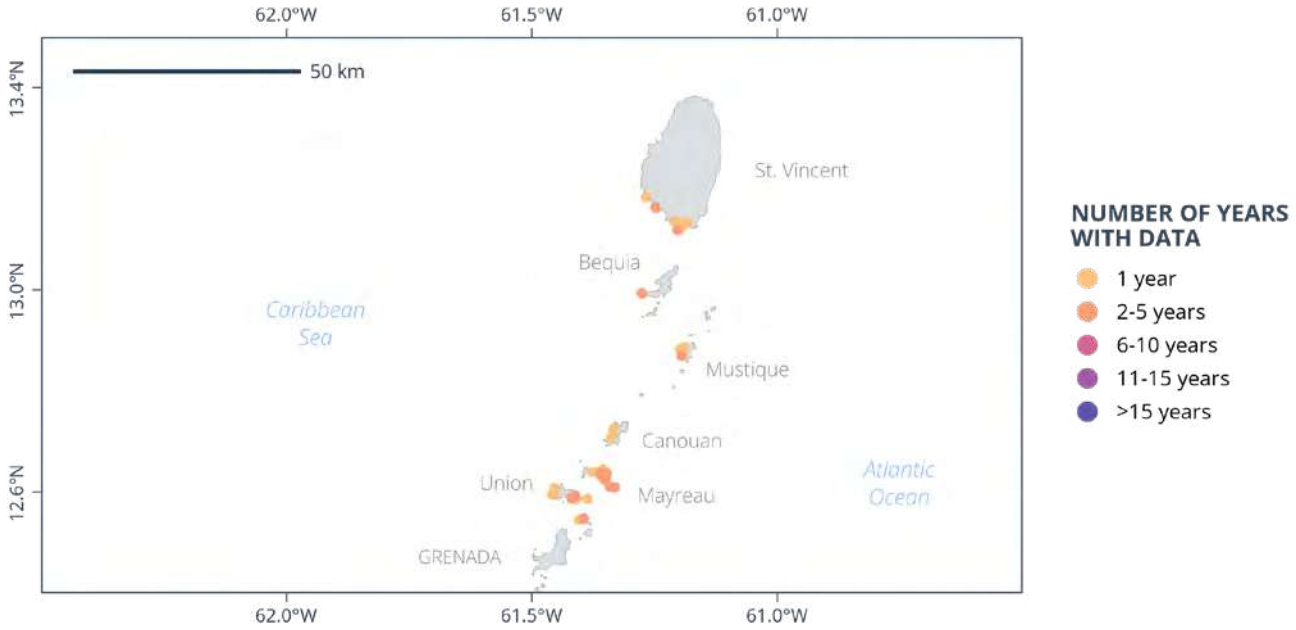
Coral reefs in Saint Vincent and the Grenadines face multiple threats, including uncontrolled tourism development, which can degrade habitats through infrastructure construction, waste accumulation, recreational damage, and the introduction of invasive species (Christie et al., 2015). Overfishing also poses a major risk by reducing fish populations, disrupting reef ecology, and causing long-term ecological impacts (Creary, 2010). Additionally, pollution from liquid and solid waste, including agrochemicals and sewage, degrades riverine and coastal water quality, while siltation from land runoff further threatens sensitive coastal and riverine ecosystems.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.35.3**) encompassing 42 monitoring sites. These data were collected through 60 surveys conducted between 2004 and 2024.

↓ **Figure 2.35.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

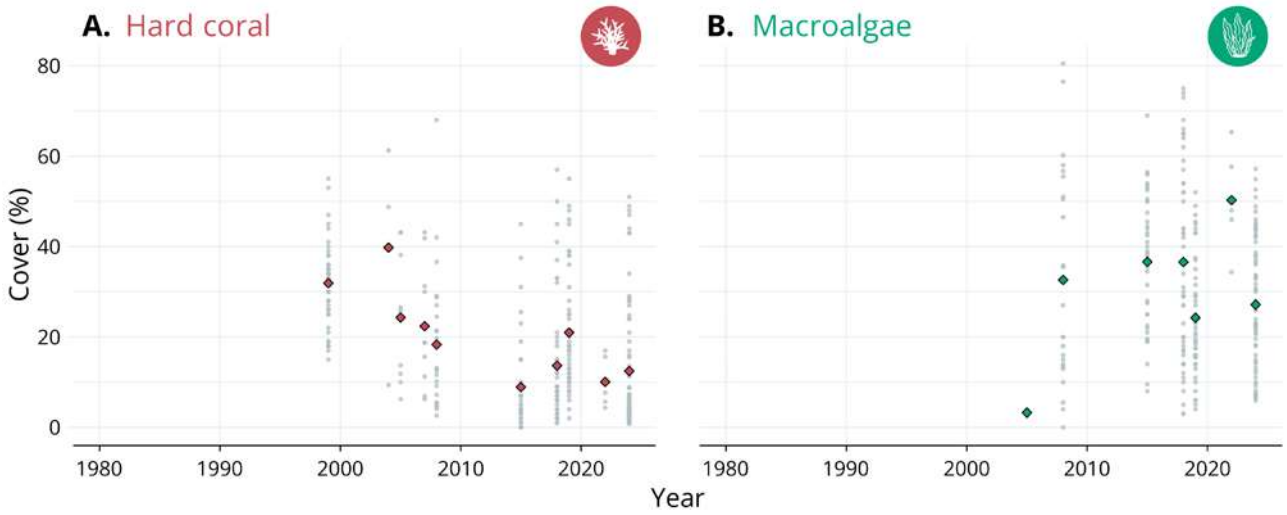




↑ **Figure 2.35.4.** Spatio-temporal distribution of benthic cover monitoring sites across Saint Vincent and the Grenadines. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Coral reef health in Saint Vincent and the Grenadines has changed over the past two decades, but overall conditions remain relatively better than in many other parts of the Caribbean. In 1999, fore reefs at Horseshoe Reef showed substantial live coral cover, averaging 30% at shallow depths and nearly 40% at deeper zones, dominated by *Orbicella annularis* and *Porites* species, although *Acropora palmata* had largely disappeared (Deschamps et al., 2003). By 2005, surveys across the Grenadine Bank reported lower coral cover, ranging from 2.5% at St. Vincent's Gardens site to 17.3% at Plantain, Mustique, with larger commercial and herbivorous fish species, such as groupers and parrotfish, becoming scarce (Baldwin et al., 2007).

In 2014, coral cover in Tobago Cays Marine Park had declined to 14.7%, about half the 1999 value, largely due to bleaching events in 2005 and 2010, as well as storm impacts (Philips et al., 2015). Despite this, herbivorous fish biomass remained very high in Tobago Cays, and signs of coral recruitment suggested potential for recovery. By contrast, in Mustique Marine Conservation Area and the South Coast Management Area, coral cover remained moderate to good in 2014 (24–27%) along with positive coral-to-algae ratios, but both herbivorous and commercially important fish biomass were critically low, highlighting ongoing ecological pressures. By 2024, coral cover declined to ~10% both within and without the Tobago Cays Marine Park. Overall, the general trajectory of coral decline in Saint Vincent and the Grenadines mirrors that of other Caribbean islands, where similar challenges such as bleaching, sedimentation, and overfishing persist, but the observed stabilization after a bleaching event may indicate more recovery potential than other locations.



↑ **Figure 2.35.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Saint Vincent and the Grenadines from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 36

Saint-Barthélemy

Authors: Claude Bouchon, Yolande Bouchon-Navaro, Sébastien Gréaux, Laëtitia Mathon, Jérémy Wicquart.

Data contributors: Claude Bouchon, Yolande Bouchon-Navaro, Sébastien Gréaux, Karl Questel, Franciane Lequellec, Pedro Portillo.

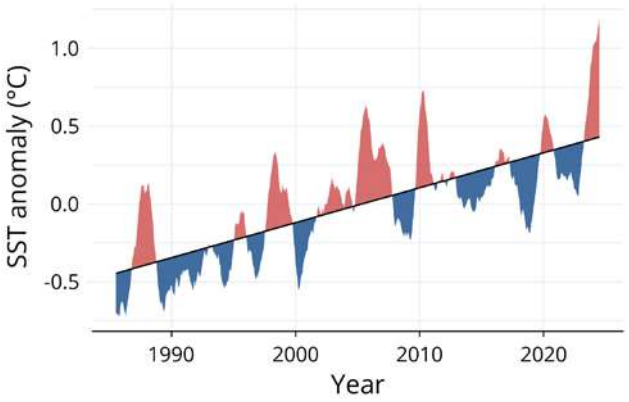
INTRODUCTION

Saint-Barthélemy, a French overseas collectivity in the Lesser Antilles, lies just southeast of Sint Maarten (**Figure 2.36.4**). Coral reef extent is about 27 km², which represents only 0.1% of the Caribbean's total coral reef extent. Three main reef morphotypes characterize the coastal environment of Saint-Barthélemy (Bouchon et al., 2008a). First, veneer reefs, which develop on most rocky substrates along the island's coastline and surrounding islets. They consist of a thin layer of coral growth that does not obscure the underlying geomorphology. These coral assemblages often display greater species richness and biodiversity than those observed in well-developed reef structures. Second, embryonic reefs, formed by the coalescence of coral patches, typically located at the heads of certain bays such as Petite Anse. Over time, these formations naturally evolve into fringing reefs. Finally, fringing reefs, found in several bays (*e.g.*, Saint-Jean Bay, Petit and Grand Cul-de-Sac bays) and sometimes connecting islets to the mainland (*e.g.*, Tortue, Caye Verte).

In 2022, Saint-Barthélemy had about 10,500 residents, with a population density of 440 inhabitants per km² (INSEE, 2022). The local economy is almost entirely driven by luxury tourism (Bouchon et al., 2008). The value of coral reefs and associated ecosystems in Saint Barthélemy is estimated to €32 million annually, mostly through tourism, coastal protection and reef fisheries (IFRECOR, 2021). A marine reserve covering approximately 1,200 hectares was established in 1996.

THREATS

Thermal regime



↑ **Figure 2.36.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Saint-Barthélemy. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

The long-term average in SST from 1985 to 2024 over coral reefs of Saint-Barthélemy was 27.51 °C. Over this period, SST increased by 1.04 °C, representing a warming rate of 0.027 °C per year (**Figure 2.36.1**).

In 1998, a bleaching event affected coral communities throughout the Lesser Antilles. The impact of this event on the coral reefs of Saint-Barthélemy was not documented. However, reefs in neighboring islands such as Guadeloupe and Martinique experienced substantial post-bleaching mortality, with coral cover declining by 20–30%, depending on the site.

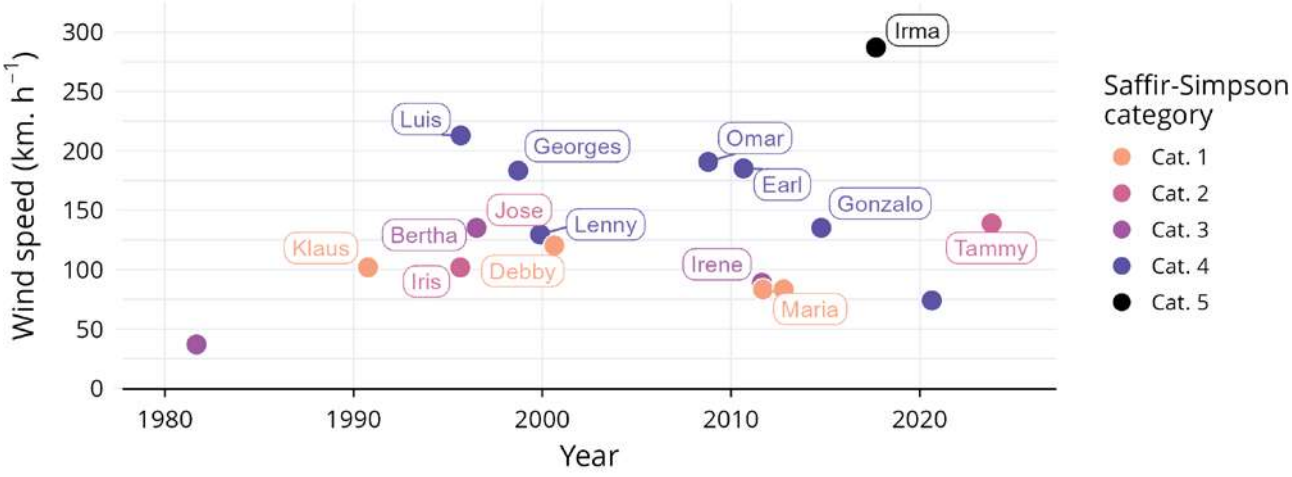
A major bleaching event occurred again in 2005, severely impacting both the Lesser and Greater Antilles, and to a lesser extent, the wider Caribbean region (Eakin, 2010). In Saint-Barthélemy, delayed coral mortality observed in 2006 led to a reduction in coral cover ranging from 20% to 30%, depending on the location (Bouchon et al., 2006; 2008b,c).

More recently, in 2023 and 2024, two consecutive bleaching events affected reefs across the Antilles. Approximately 90% of coral species and 85% of individual colonies exhibited bleaching, with marked interspecific differences—*Acropora* spp. being among the most severely affected. Coral mortality was estimated at around 12% of colonies (Bouchon and Bouchon-Navaro, 2024).

Hurricanes

Between 1980 and 2024, a total of 26 tropical storms passed within 100 km distance from a coral reef of Saint-Barthélemy, and of these 14 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.36.2**). The hurricane with the highest sustained wind speed over the studied period was Hurricane Irma, in 2017, which passed 1 km from a coral reef with sustained wind speed of 287 km.h^{-1}.

Following Hurricane Irma, reef benthic organisms, including corals, sponges, and gorgonians, sustained severe damage at depths between 0 and 5 m, while deeper zones were only moderately affected. Contrary to expectations, macroalgal overgrowth was not reduced by wave action. Mobile benthic invertebrates, particularly holothurians and sea urchins, experienced marked population declines. Post-cyclone ichthyological surveys revealed no significant effects on fish biodiversity and no major changes in species assemblages, indicating a high level of ecological resilience (Bouchon and Bouchon-Navaro, 2019).



↑ **Figure 2.36.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Saint-Barthélemy. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Saint-Barthélemy are not represented, although they can have had an impact.

Other threats

Anthropogenic pressures on the environment of Saint-Barthélemy began historically with extensive deforestation, exacerbated by the presence of a large semi-feral goat population. These processes, now compounded by poorly regulated landscaping and urban development, have resulted in significant soil erosion and excessive sedimentation in coastal marine habitats.

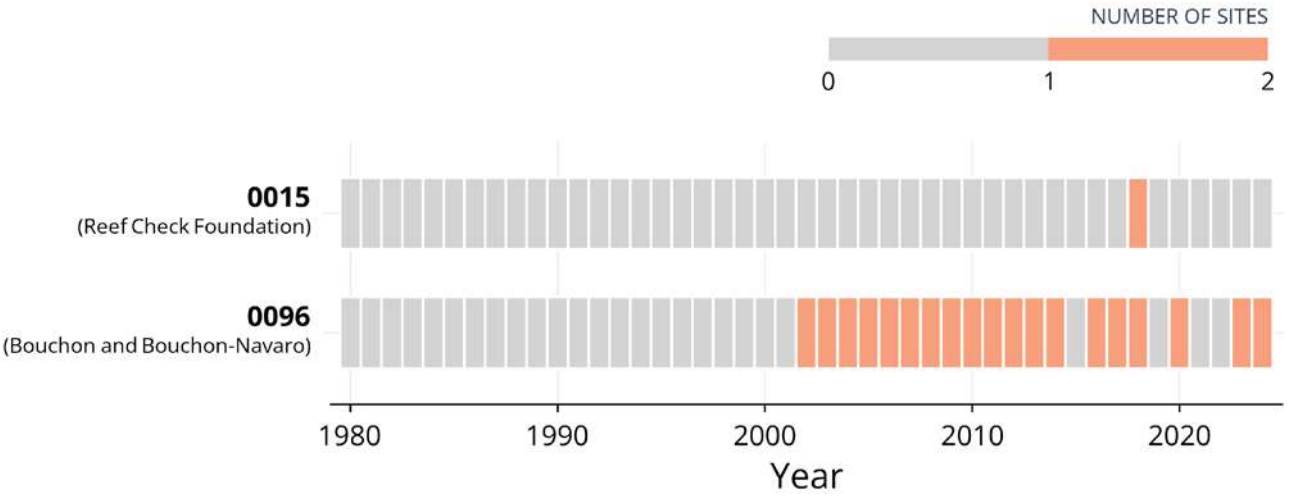
The island's centralized wastewater management system is generally effective, relying on a tertiary-level treatment facility. However, decentralized sanitation linked to scattered residential areas continues to contribute to nutrient enrichment and eutrophication in nearshore waters.

Among natural stressors, epizootic mass mortality events affecting the long-spined sea urchin *Diadema antillarum*, a key benthic herbivore, were recorded in 2019 and 2023. The decline of this species has promoted macroalgal proliferation, which competes with and overgrows coral colonies.

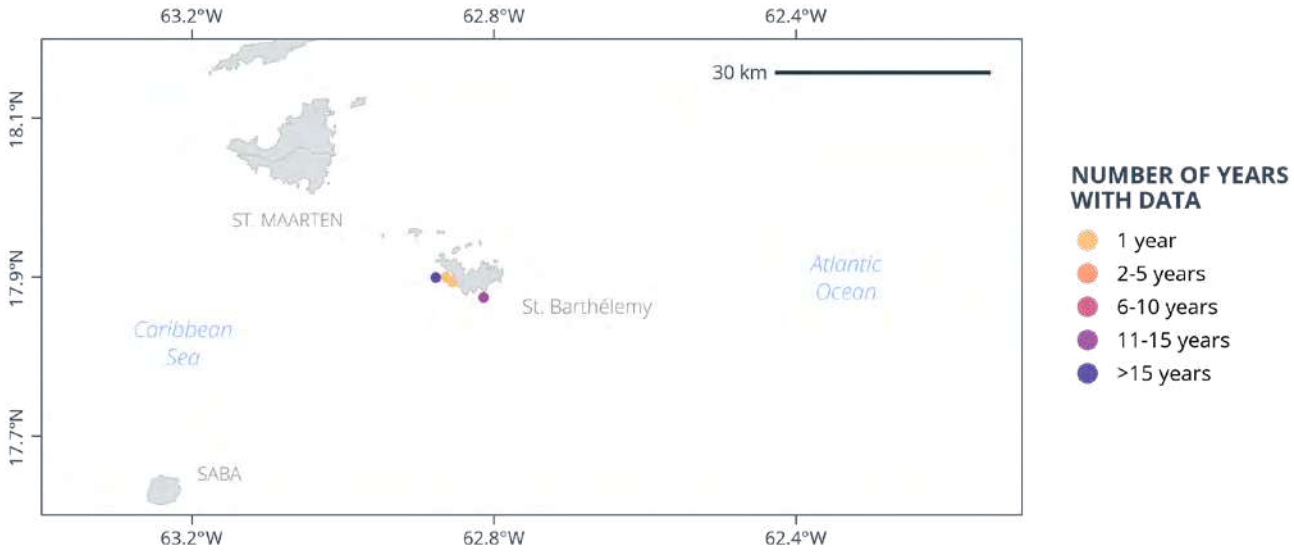
In 2022, coral communities around the island were impacted by Stony Coral Tissue Loss Disease (SCTLD). Of the 40 scleractinian coral species recorded in Saint-Barthélemy, 30 exhibited varying degrees of susceptibility, whereas all five hydrocoral species appeared unaffected. The outbreak resulted in the mortality of approximately 30% of coral colonies (Bouchon and Bouchon-Navaro, 2023).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.36.3**) encompassing 4 monitoring sites. These data were collected through 43 surveys conducted between 2002 and 2024. Quantitative monitoring of the coral reefs in Saint-Barthélemy began in 2002 with the establishment of the GCRMN network at two reef sites: one located within the marine reserve and the other in a non-protected area (**Figure 2.36.3**, Bouchon et al., 2002).



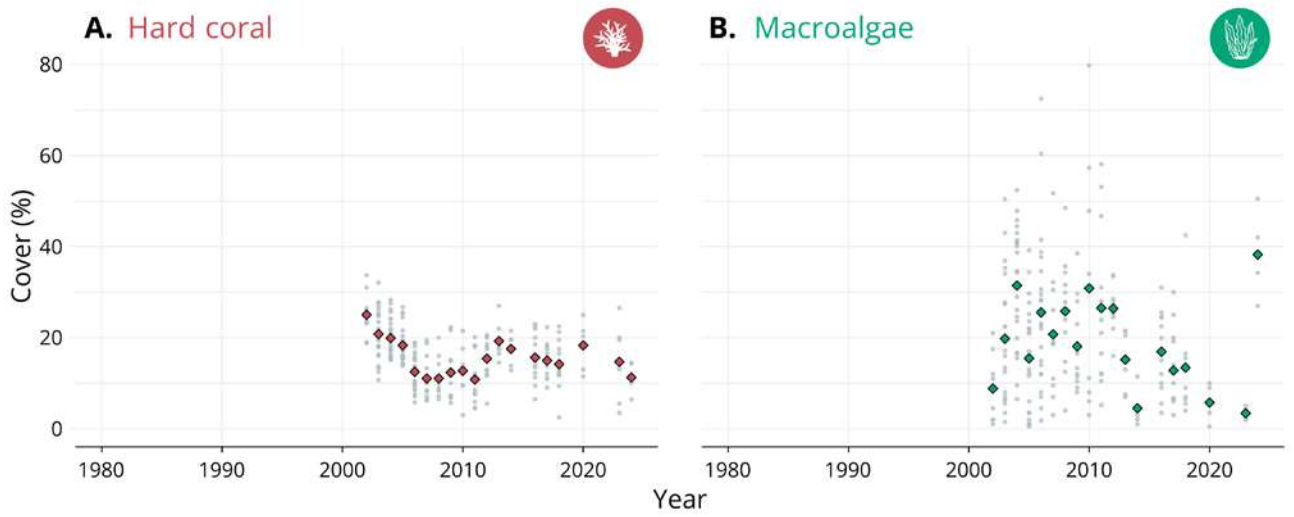
↑ **Figure 2.36.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.36.4.** Spatio-temporal distribution of benthic cover monitoring sites across Saint-Barthélemy. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

From 2002 to 2024, hard coral cover decreased from 20% to 10%. During the same period, the density of coral colonies dropped by nearly 60%. This pattern was irregular: a steady decline of hard coral assemblages was observed between 2002 and 2024, with the last year marked by a major coral bleaching event that caused significant coral mortality. Over the following years, hard coral cover and the number of colonies remained stable and even slightly increased until 2022, when SCTLD spread. In 2023 and 2024, two consecutive coral bleaching events impacted the coral assemblages of Saint-Barthélemy. These years were characterized by a decline in coral coverage to 10%, linked to a new drop in colony abundances.

Regarding macroalgae cover, strong seasonal fluctuations in the monitoring data from 2002 to 2024 obscure the identification of a clear long-term trend.



↑ **Figure 2.36.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Saint-Barthélemy from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 37

Saint-Martin

Authors: Laëtitia Mathon, Julien Chalifour, Hadrien Bidenbach.
Data contributors: Aude Berger, Clément Bonnardel, Amandine Bordin, Julien Chalifour, Vincent Oliva.

INTRODUCTION

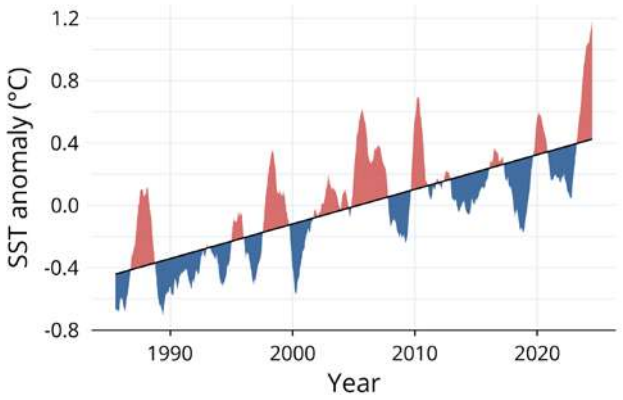
Saint-Martin, a French Caribbean island of 54 km² with a combined Exclusive Economic Zone of 2,665 km² shared with Saint-Barthélemy (see chapter 36), hosts 13 km² of coral reefs—representing 0.053% of the Caribbean's total extent. Its population has grown rapidly, from 8,000 in 1980 to 35,000 in 2024. To protect its fragile ecosystems, a marine and terrestrial nature reserve was established in September 1998, covering 3,060 ha across the island's north and east coast, including Pinel and Tintamarre islands and their fringing reef. Coral reefs and associated ecosystems play a crucial economic role: their annual value is estimated at €27 million, ranging from €19–35 million, supporting around 2,600 jobs, 400 businesses, and 1,000 households. The largest contributions come from marine tourism (€20M/year), coastal flood protection (€3M/year), and carbon sequestration by mangroves and seagrass beds (€3M/year), underscoring the ecological and socio-economic importance of Saint-Martin's reef systems (Pascal et al., 2016).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Saint-Martin was 27.49 °C. Over this period, SST increased by 1.03 °C, representing a warming rate of 0.026 °C per year (**Figure 2.37.1**).

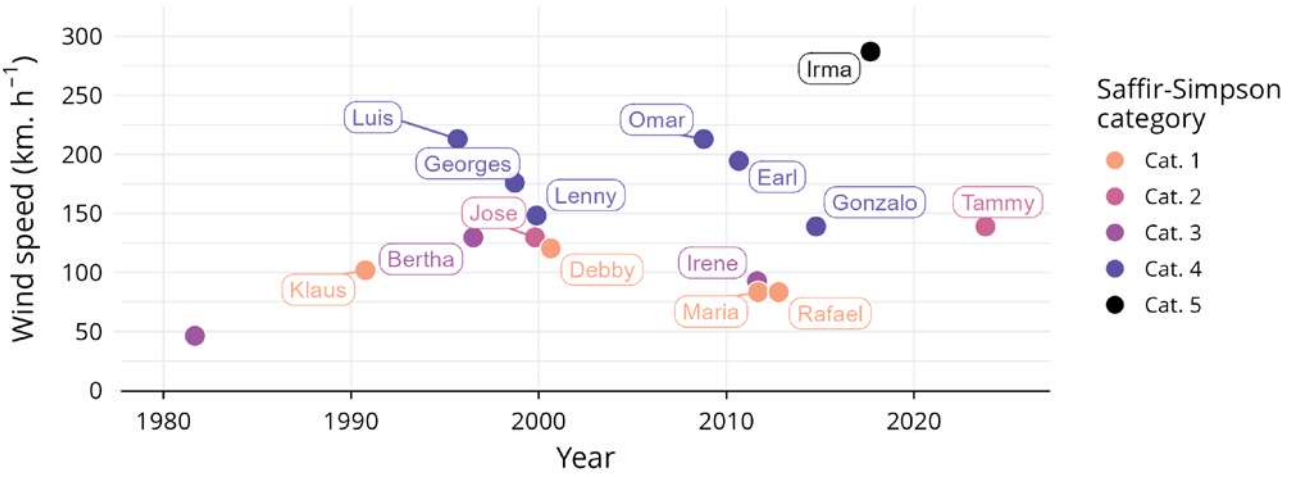
In the Guadeloupe archipelago, including Saint-Martin, the 1998 bleaching event affected 56% of corals, with 80% of their surface showing signs of bleaching. During the following year, coral mortality remained relatively low compared to other parts of the Caribbean (20–30% of affected corals), except for *Diploria labyrinthiformis*, which experienced mortality rates close to 80% (Bouchon et al., 2002).



↑ **Figure 2.37.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Saint-Martin. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 24 tropical storms passed within 100 km distance from a coral reef of Saint-Martin, and of these 13 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.37.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed 0 km from a coral reef with sustained wind speed of 287 km.h^{-1}.



↑ **Figure 2.37.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Saint-Martin. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Saint-Martin are not represented, although they can have had an impact.

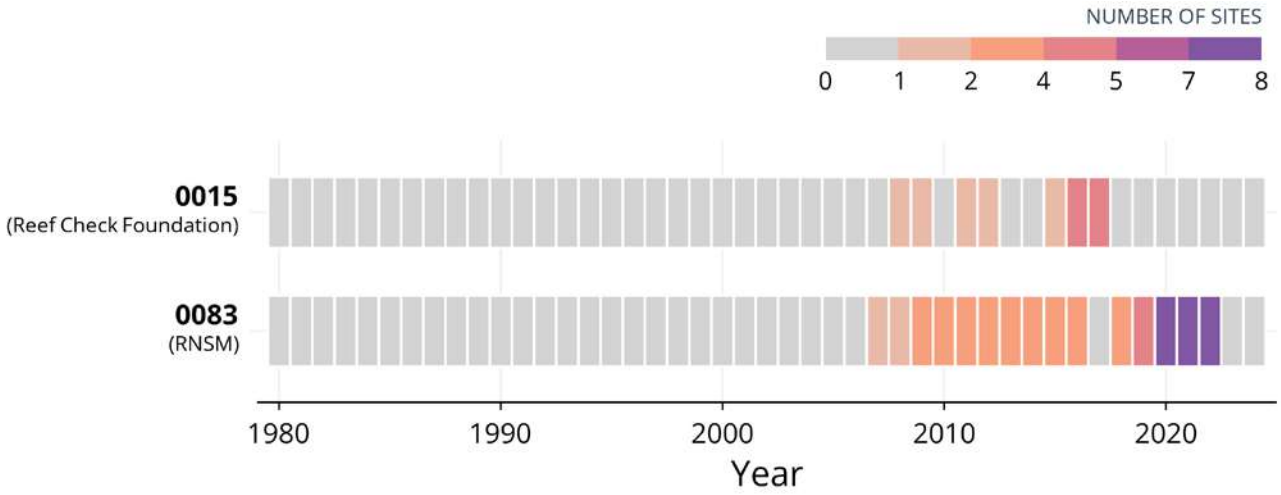
Hurricanes Luis and Marilyn (1995), and Lenny (1999) caused severe damage to coral reefs, with waves up to 13 m destroying corals, sponges, and gorgonians down to 25 m depth, while heavy rains triggered erosion and sediment overload. In Saint-Martin and Saint-Barthélemy, Hurricane Luis not only damaged coral communities directly but also disturbed shallow shelf sediments, producing fine silt that smothered benthic organisms (Bouchon et al., 2002). More recently, monitoring after Hurricane Irma showed further declines in benthic cover, with average decreases of 6.25% in hard corals, 9.75% in sponges, and 15% in macroalgae between February and December 2017 (Chalifour, 2018).

Other threats

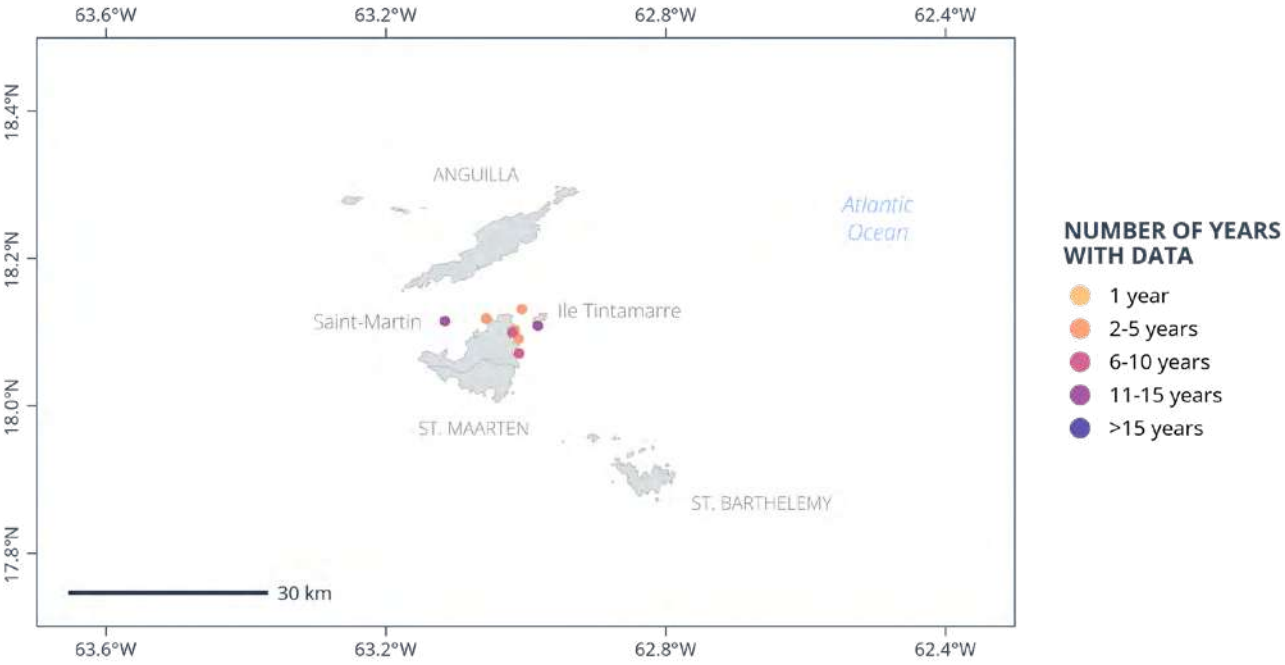
Beyond storm damage, coral reefs in the French Antilles face multiple human-induced pressures. Temporary algal blooms often occur after cyclones, fueled by coastal eutrophication from massive land runoff and the resuspension of nutrients trapped in sediments. Overfishing further compounds ecosystem stress, with food species heavily exploited and queen conch (*Aliger gigas*) populations continuing to decline despite regulations on capture size and reproduction periods (Bouchon et al., 2002). Broader anthropogenic impacts include coastal urbanization, wastewater discharge, inadequate sanitation, and the effects of overuse from tourism, all of which degrade reef health (IFRECOR, 2021). Nautical traffic and tourist activity had decreased after Irma and COVID-19, but are now back to prior levels and are still growing. The French marine protected area (RNN de St Martin) alone is hosting 69 registered commercial companies operating more than 198 vessels, which carried more than 50,000 customers to the 3,000 ha of MPAs in 2024. *Sargassum* strandings and the release of organic matter from their decomposition in coastal waters is responsible for increased eutrophication of coastal accumulation areas, occasional episodes of anoxia, increasingly recurrent phenomena of high turbidity in coastal waters, and hypersedimentation of algal debris. Secondarily, there are also increasingly frequent and massive seasonal blooms of filamentous green and brown macroalgae, which can lead to the formation of benthic mats covering the seabed (coral communities and seagrass beds).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 2 datasets (**Figure 2.37.3**) encompassing 12 monitoring sites. These data were collected through 83 surveys conducted between 2007 and 2022.



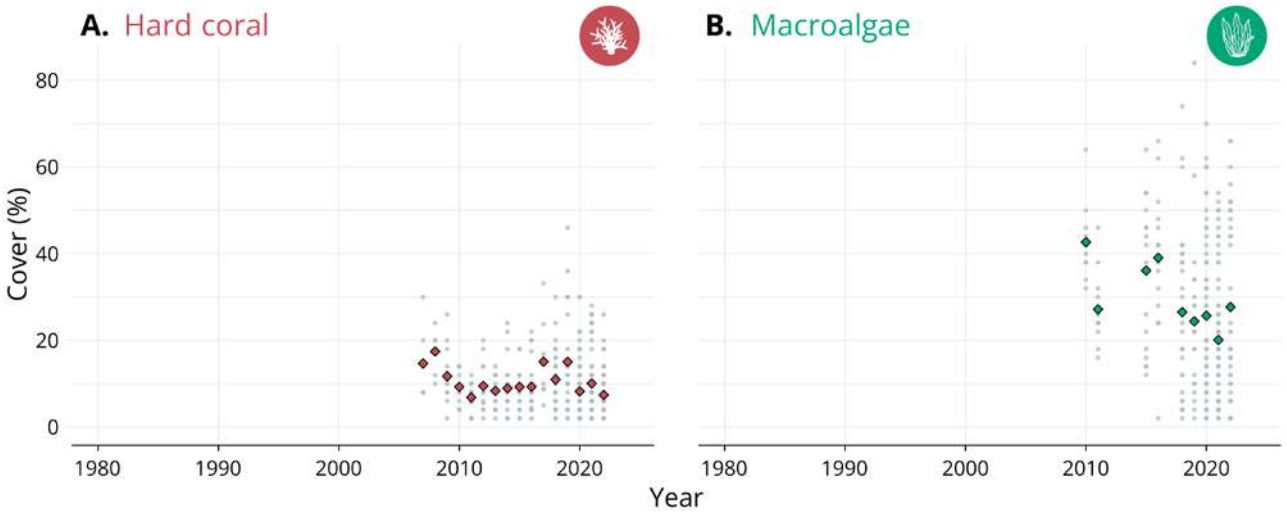
↑ **Figure 2.37.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



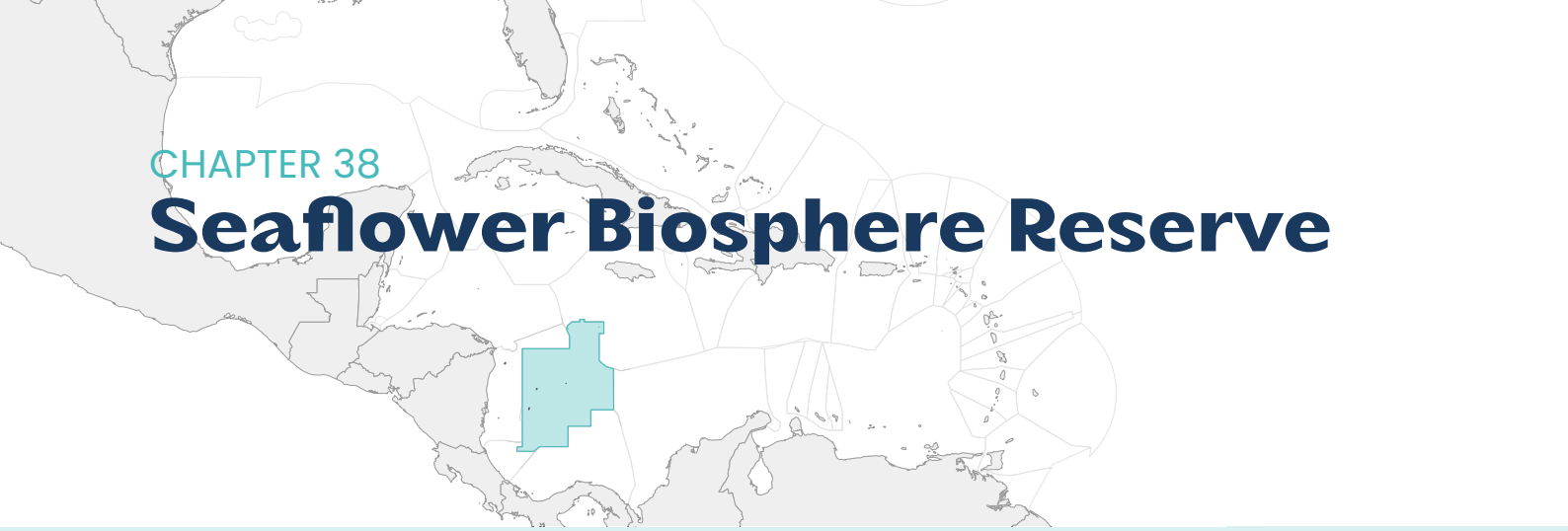
↑ **Figure 2.37.4.** Spatio-temporal distribution of benthic cover monitoring sites across Saint-Martin. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Coral reef health in Saint-Martin has followed a general trend of decline since the early 2000s, with coral communities increasingly replaced by algal assemblages (Bouchon et al., 2002). In 2004, live coral cover ranged between 20–26%, but competition from macroalgae such as *Dictyota* and *Lobophora* was already evident (Bouchon et al., 2008). Between 2007/08 and 2011/12, monitoring recorded a statistically significant drop in coral cover within the marine reserve, with averages of 13% inside and 9% outside, while algal cover dominated (67% and 54%). Recruitment rates remained very low, and herbivorous urchins (*Diadema antillarum*), crucial for controlling algae, were absent (Pareto, 2012). By 2017, site-specific differences emerged: Rocher Créole displayed relatively good reef health, with 33% coral cover and only 8% macroalgae, while Galion was characterized by high macroalgal cover (51%) and much lower coral presence (Chalifour, 2017). The Galion station is located in close proximity to the discharge point into the natural environment of the island's largest wastewater treatment plant, and the site is particularly vulnerable to illegal fishing,

which may also harm the natural potential for regulating algal growth. The passage of Hurricane Irma in 2017 caused additional stress, contributing to declines observed in 2018. Yet, by 2019, coral cover averaged 14.9% across monitoring networks, reflecting a slow recovery trend since 2015, though still below the 2008 maximum of 18.5%. Meanwhile, macroalgal cover has generally decreased since 2015, despite strong variability between sites, with some stations still showing high levels (IFRECOR, 2021). The wide variability in macroalgae cover may be related to the sites' exposure to prevailing swells and storms.



↑ **Figure 2.37.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Saint-Martin from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.



CHAPTER 38

Seaflower Biosphere Reserve

Authors: Valeria Pizarro, Juliana Vanegas Gonzalez, Julian Prato-Valderrama, Juliana Torres Acero, Andrea Rivera-Sosa.

Data contributors: Andrés Felipe Acosta-Chaparro, David Alejandro Alonso-Carvajal, Francisco Arias Isaza, Brian Beck*, Carlos Daza-Guerra, Alexandra Dempsey*, Graham J. Edgar, Diana Isabel Gómez López, Catalina Gómez-Cubillos, Mike Haley*, Judith Lang*, Raúl Navas-Camacho, Lizzi Oh, Valeria Pizarro, Rick D. Stuart-Smith, Laura Sánchez-Valencia, Andrew W. Bruckner*, Sven Zea.

*AGRRA data contributor(s).

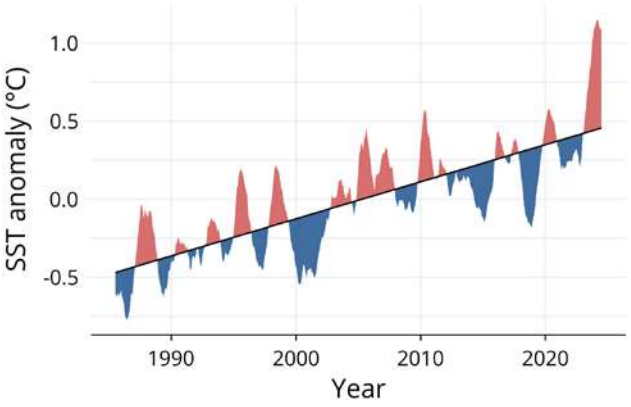
INTRODUCTION

The Seaflower Biosphere Reserve (SBR), declared in 2000 by UNESCO, is located at the extreme jurisdiction of Colombia in the Central-Western Caribbean Sea within the Archipelago of San Andrés, Providencia and Santa Catalina. It has an area of approximately 180,000 km², of which less than 1% is land, and the vast majority is marine waters (CORALINA-INVEMAR, 2012). Additionally, in 2005, the Ministry of Environment and Sustainable Development declared the Seaflower Marine Protected Area (MPA), covering 65,000 km² within the SBR. In 2014, the area was assigned as an Integrated Management District, constituting the largest and most populated of the country's marine island reserves (CCO, 2024). The SBR comprises nine islands, three small, inhabited islands (San Andrés, Providencia and Santa Catalina), six reef islands with cays and banks (Alburquerque, Este Sudeste, Quitasueño, Serrana, Roncador, Serranilla) and two shoals (Bajo Alicia and Bajo Nuevo) (CORALINA-INVEMAR, 2012 ; decreto 1946 de 2013, Presidencia Colombia, Prato et al., 2024). The SBR has diverse ecosystems such as coral reefs, seagrass meadows, mangroves and wetlands. Within its jurisdiction it hosts 880 km² of coral reefs, which represent 3.632% of the Caribbean's total coral reef extent and around 77% of Colombia´s reefs. The marine environment supports 89 hard coral species, 40 octocoral species, 164 sponge species, and around 731 fish species (CORALINA-INVEMAR, 2012; Vides et al., 2016).

THREATS

Thermal regime

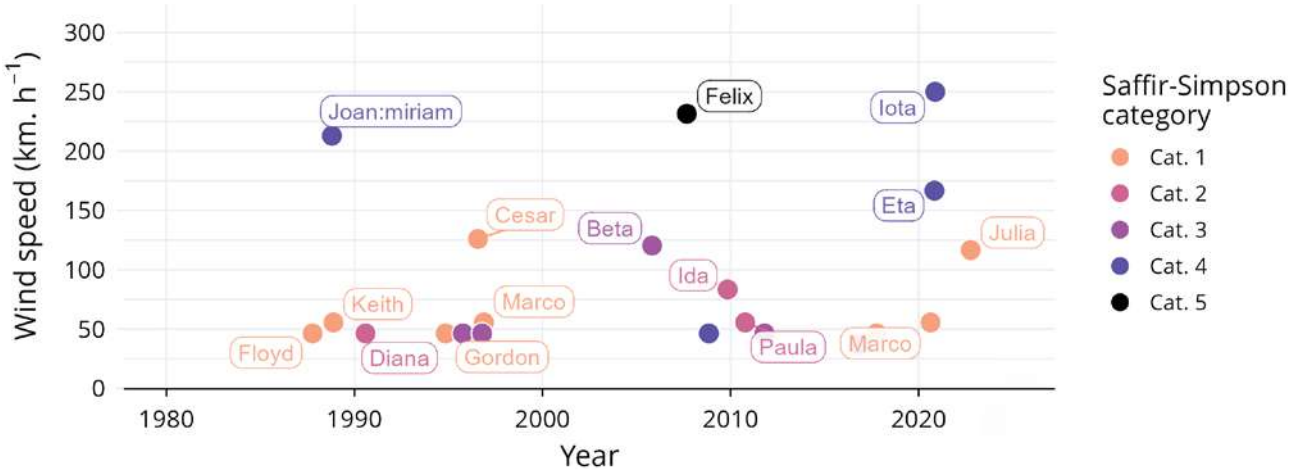
The long-term average in SST from 1985 to 2024 over coral reefs of the Seaflower Biosphere Reserve was 27.99 °C. Over this period, SST increased by 1.05 °C, representing a warming rate of 0.027 °C per year (**Figure 2.38.1**). Between 2023 and 2024, the fourth global coral bleaching event impacted the SBR. From July 2023 to May 2024, the western coast of San Andres Island indicated a decline of over 90% in live coral cover for *Acropora cervicornis, Agaricia agaricites, A. fragilis, A. humilis,* and *A. lamarcki*, and a reduction of approximately 80% for *Porites porites* and *P. divaricata* (Pizarro et al., in prep). In 2023 the bleaching rates observed varied between 39% and 54% in San Andrés, 4.8% to 6.5% in Providencia and Santa Catalina, 9.7% to 11% in Este Sudeste, and 4.2% to 4.8% in Cayo Alburquerque (CORALINA, 2024). Mass coral bleaching was also observed at the reef´s crests of the San Andres barrier which affected around 90% of corals, with the exception of a few *P. astreoides* colonies (Prato et al.,2024b).



↑ **Figure 2.38.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Seaflower Biosphere Reserve. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 35 tropical storms passed within 100 km distance from a coral reef of Seaflower Biosphere Reserve, and of these 7 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.38.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Iota, in 2020, which passed less than 10 km north of the islands of Providencia and Santa Catalina with sustained wind speed of 250 km.h^{-1}. For the islands of San Andres, Providencia and Santa Catalina, hurricane Iota severely affected the reefs located on the west side of the islands at depths of up to 15 meters, while no further impact was observed in deeper areas, shallow hard-bottom platforms (<5 m) on the west side of San Andrés were severely impacted, with near-total loss of benthic cover (CORALINA, 2020; Prato, 2020, pers. obs,). According to Gómez et al. (2022) *Orbicella annularis*, *O. faveolata* and *Pseudodiploria strigosa* were the most affected species with several colonies overturned and fragmented.



↑ **Figure 2.38.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Seaflower Biosphere Reserve. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Seaflower Biosphere Reserve are not represented, although they can have had an impact.

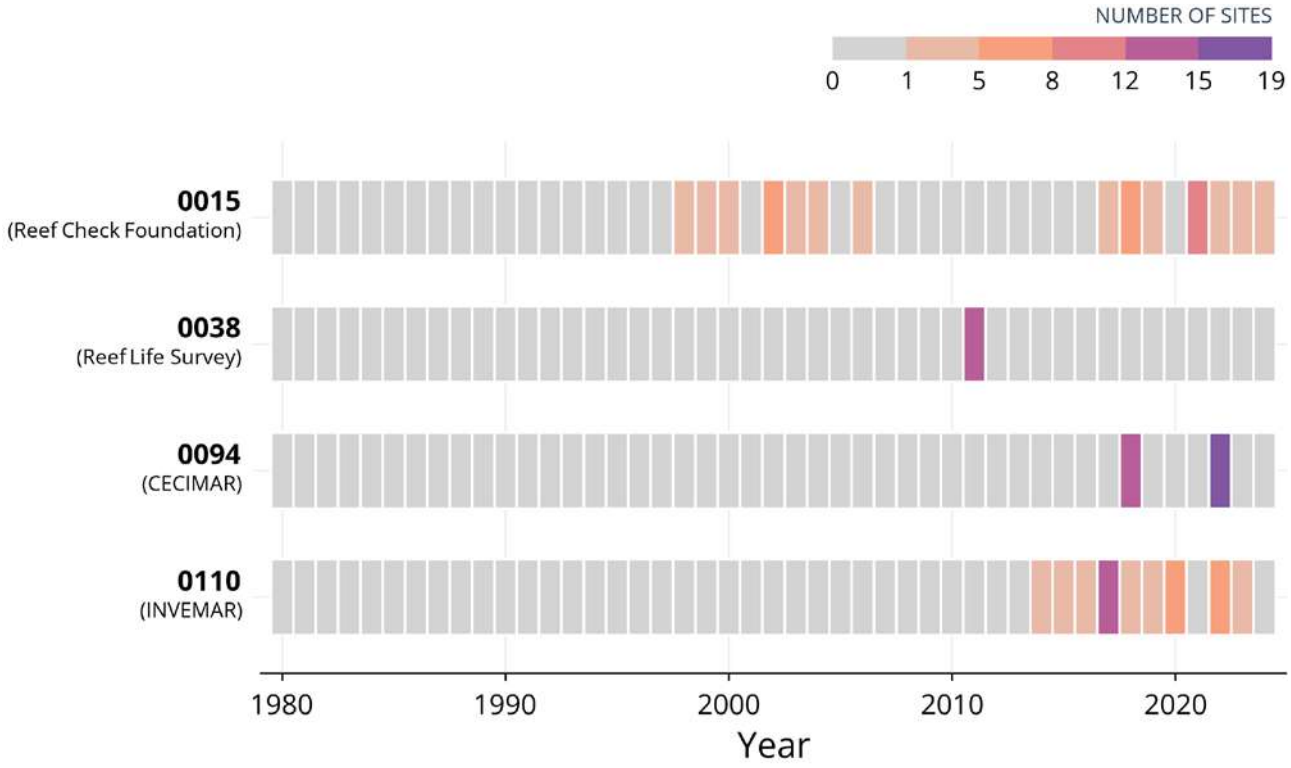
Coral diseases

Stony Coral Tissue Loss Disease was first observed in the Seaflower Biosphere Reserve at Bajo Nuevo Cays in November 2021 during a Seaflower Expedition (CCO, 2024). It was subsequently confirmed at both Serranilla and Bajo Nuevo Cays during the 2022 Pristine Seas Expedition, and later at San Andrés Island. It has since spread throughout the Archipelago. The disease has rapidly proliferated throughout the region, causing widespread mortality across nearly three dozen coral species and significantly disrupting coral communities, population dynamics and ecosystem functionality. By December 2023, between 32% and 61% of the colonies registered in San Andrés, Este Sudeste cay, Alburquerque cay and Providencia were affected by coral bleaching and the prevalence of the SCTLD disease ranged between 2 and 4%, being *Siderastrea siderea* (average of 42% of the colonies affected by SCTLD) and *O. annularis* (40.1%) the most affected species,

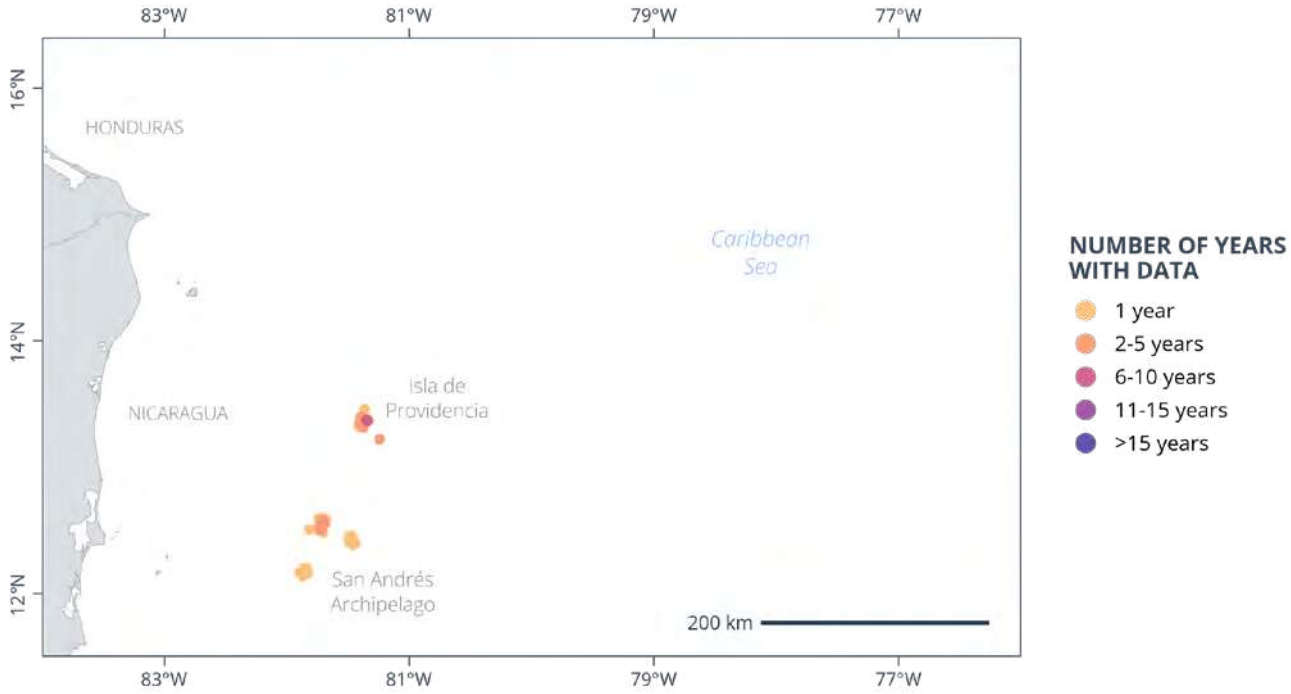
followed by *O. franksi, Montrastraea cavernosa,* and *O. faveolata*, with percentages ranging from 5 to 13.5% of infection (CORALINA, 2024). This disease has increased coral reef degradation in the area in ways rarely seen. In some locations, the disease has led to mortality rates of up to 90% in affected coral populations, with the potential to irreversibly alter reef composition and ecosystem function (Alvarez-Filip et al., 2022). Given its scale and impact, SCTLD poses one of the most urgent and complex challenges currently facing coral reef conservation efforts in the Caribbean.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 4 datasets (**Figure 2.38.3**) encompassing 90 monitoring sites. These data were collected through 137 surveys conducted between 1998 and 2024. Coral reef monitoring in the Seaflower is limited by its remote keys and atolls, with most data originating from San Andrés, Providencia, and Santa Catalina, while distant sites are surveyed only every 5–10 years.

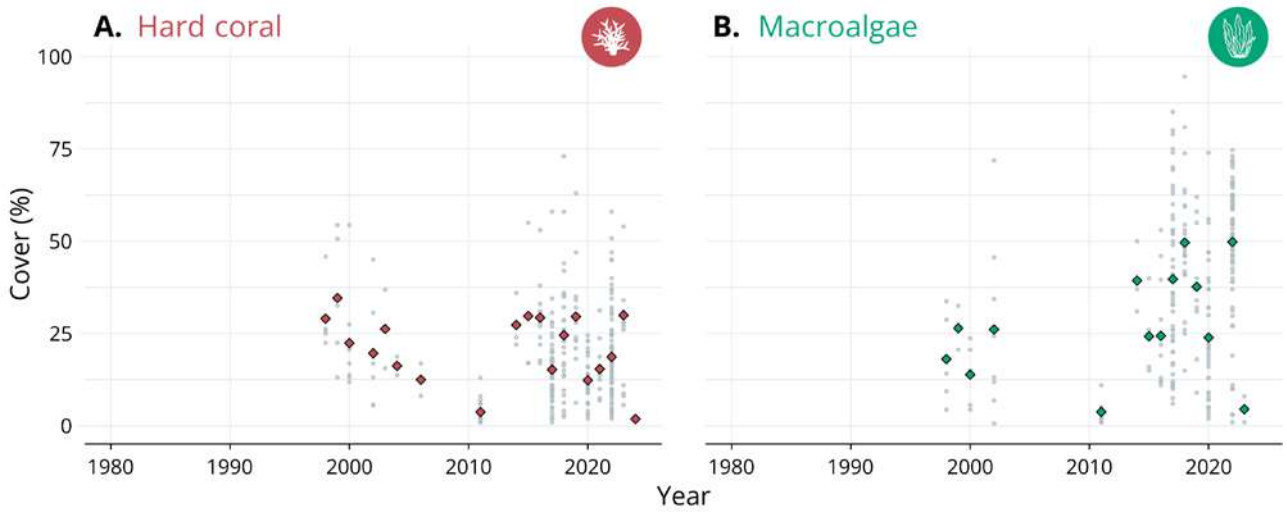


↑ **Figure 2.38.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.38.4.** Spatio-temporal distribution of benthic cover monitoring sites across Seaflower Biosphere Reserve. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Over the past decades, coral reefs in the SBR have been shaped by a combination of natural disturbances such as hurricanes and increasing anthropogenic pressures, including climate change, overfishing, and eutrophication. These stressors have led to a marked decline in hard coral cover and a strong increase in fleshy macroalgae (**Figure 2.38.5**.). For example, assessments conducted in 1995, 2003, and again in 2015–2016 at the atolls Roncador and Serrana revealed significant declines in reef-building coral species and benthic health, along with a reduction in crustose coralline algae (CCA) (mean decline of 10%), decreased structural complexity (rugosity), and early signs of benthic phase shifts (Sanchez et al., 2019). Additionally, the coral cover decrease between 2020 and 2024 was accelerated by the combined effect of the spread of the SCTLD to the reserve, the mass bleaching event and the devastation of the hurricanes Iota and Eta. Along with the physical damage, coral bleaching associated with the increased temperature previous to the hurricane and the presence of diseases such as black band, white band and dark spot diseases affected the coral reefs during that period.



↑ **Figure 2.38.5.** Points denote the annual means of observed data from each site (*i.e.* monitoring data) of benthic percentage cover for hard coral in red (A), and macroalgae in green (B) in Belize from 1980 to 2024.

Modeled temporal trends coincide with most recent studies, for example reefs around San Andrés island reflected an average coral cover around 8% in 2023 and lower than 4% in 2024 (Pizarro et al. in prep). The pressure of the 2023-2024 bleaching event combined with the impact of SCTLD and other threats led to a decrease of 59% of coral cover between 2023 and 2024. Unfortunately, coral loss has continued throughout 2025 due to coral diseases (SCTLD and yellow band predominately) and it is expected this deterioration affected coral reef functionality (Pizarro et al. in prep). Coral cover in shallow barrier reef areas experienced a severe coral cover loss of 86% to 94% (Prato et al., 2024b; Prato et al., *in prep*).

Reef restoration efforts have been carried out in the SBR since 2010 starting in the National Natural Park Old Providence McBean Lagoon, focusing mainly on Acroporid species. However, during the 2023-2024 bleaching event most of the outplanted corals and fragments in the nurseries died. Currently, efforts such as coral propagation using fragmentation and larval rearing, and artificial reefs, are being implemented in the islands of San Andres, Providencia and Santa Catalina. These efforts have the potential to positively impact the SBR's reef recovery, its success will depend on appropriate practices, effective restoration actions, strengthening long-term programs and coral reef management.

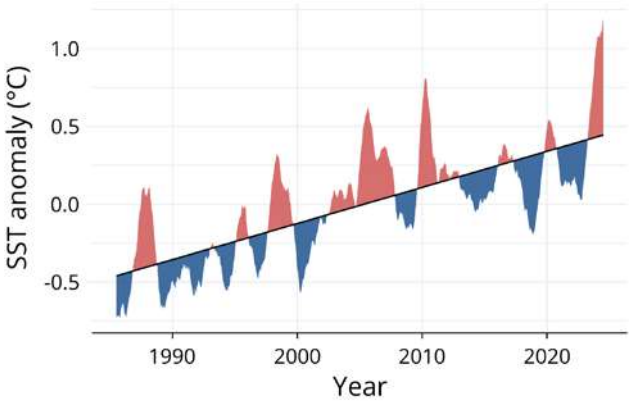
CHAPTER 39

Sint-Eustatius

Authors: Eleanor Butler, Bert W. Hoeksema, Kimani Kitson-Walters, Laëtitia Mathon.
Data contributors: Eleanor Butler, Alice Deschamps*, Kristi D. Klomp*, Chris Moses*, Sascha Steiner*.
*AGRRA data contributor(s).

INTRODUCTION

Sint Eustatius, a volcanic island of 21 km² with an Exclusive Economic Zone of 2,176 km², hosts approximately 23 km² of coral reefs, representing 0.094% of the Caribbean's total reef area. These reefs comprise both true calcareous formations and corals growing on volcanic substrates, with the most robust coral cover occurring mid-shelf near the island's southern tip. Since 1996, they have been managed within the St. Eustatius National Marine Park (27.5km²), which includes two no-take reserves (encompassing a total area of 4.9km²). Local fisheries remain small-scale, conducted primarily from open boats with outboard engines, and center on a trap fishery for Caribbean spiny lobster (*Panulirus argus*) that also yields a mix of reef fish. Fishing effort rose steadily through 2014, peaking in 2015 at an average of 79 trips per month, before declining to 25 trips per month in 2019 and partially rebounding to 42 trips per month in 2020 (Amelot et al., 2021). The island's population has grown from 1,400 in 1980 to around 3,200 in 2024.



↑ **Figure 2.39.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Sint-Eustatius. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

The reefs were severely affected by the 2005 bleaching event, which caused 70–80% of colonies to bleach and led to a decrease of live coral cover of 50% by 2008, accompanied by a sharp increase in macroalgal dominance (Bouchon et al., 2008).

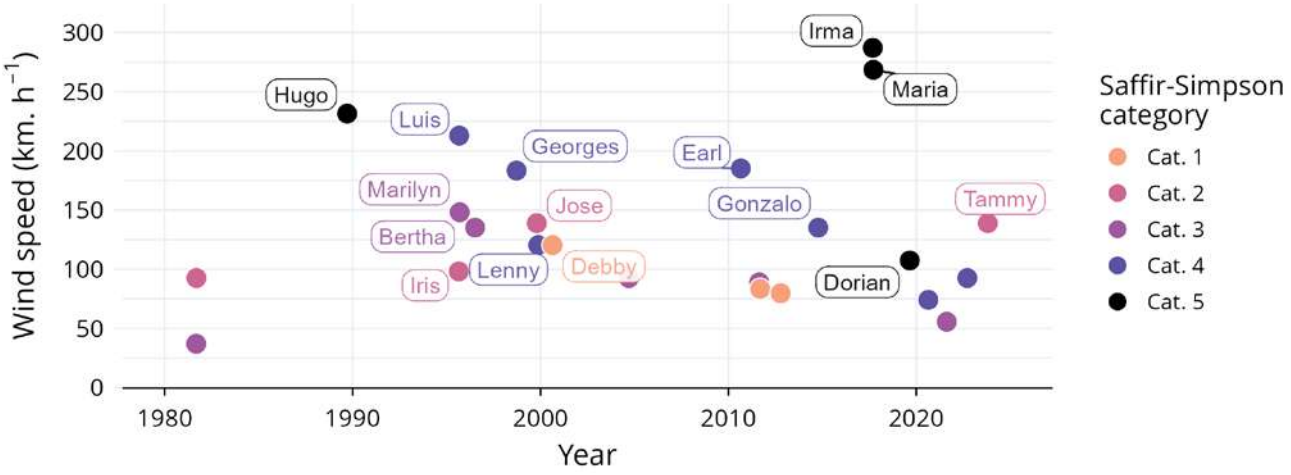
THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Sint-Eustatius was 27.76 °C. Over this period, SST increased by 1.06 °C, representing a warming rate of 0.027 °C per year (**Figure 2.39.1**).

Hurricanes

Between 1980 and 2024, a total of 33 tropical storms passed within 100 km distance from a coral reef of Sint-Eustatius, and of these 14 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.39.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed 51 km from a coral reef with sustained wind speed of 287 km.h^{-1}.

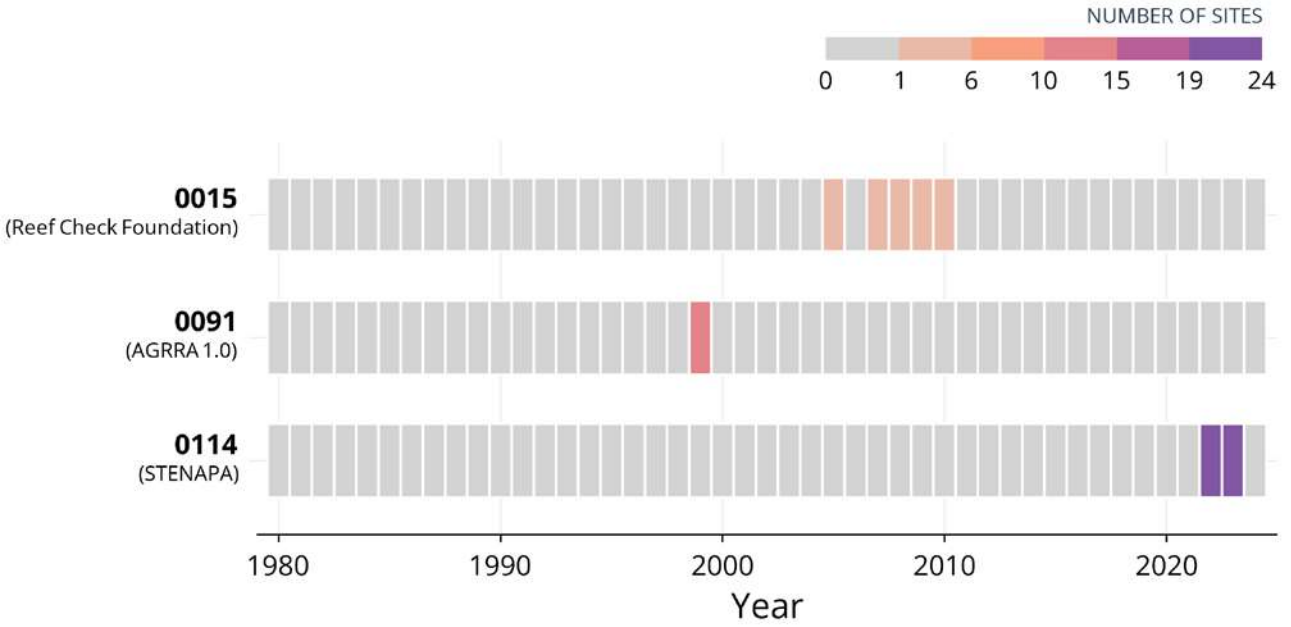


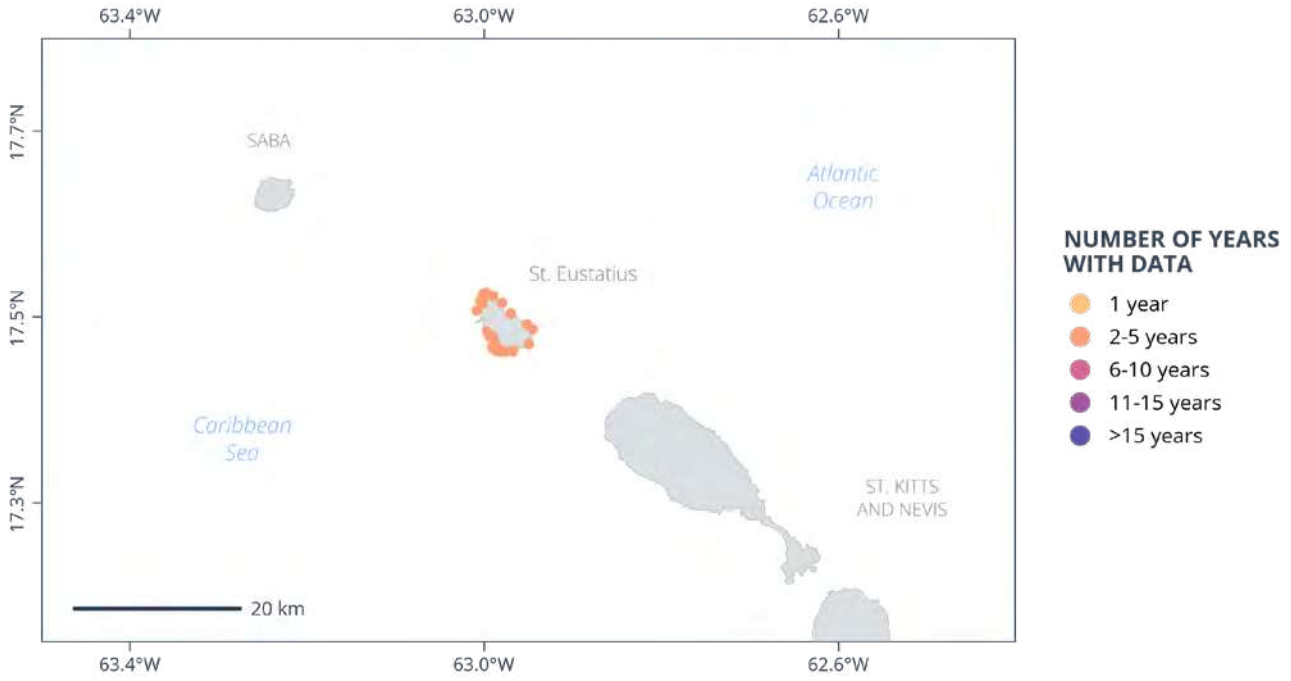
↑ **Figure 2.39.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Sint-Eustatius. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Sint-Eustatius are not represented, although they can have had an impact.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.39.3**) encompassing 37 monitoring sites. These data were collected through 72 surveys conducted between 1999 and 2023.

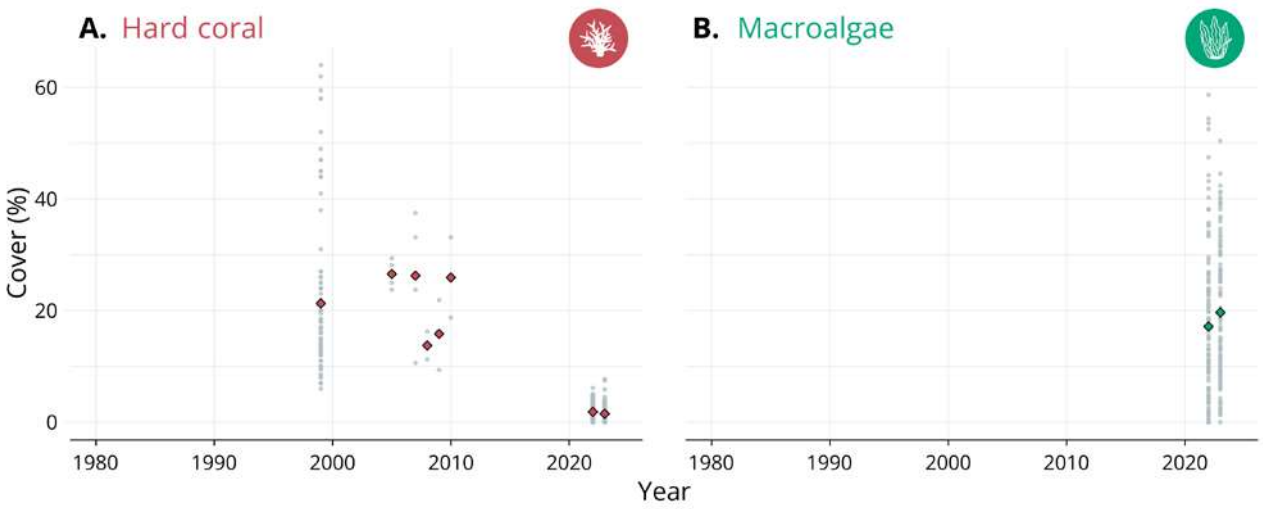
↓ **Figure 2.39.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.39.4.** Spatio-temporal distribution of benthic cover monitoring sites across Sint-Eustatius. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

The coral reefs of St. Eustatius support a high degree of biodiversity (Hoeksema et al., 2016). They have experienced a steady decline in health over the past two decades, shaped by bleaching events, hurricanes, and ecological shifts. In the late 1990s and early 2000s, reefs suffered from bleaching, with 24% of colonies affected in 1999, and significant damage following Hurricane Lenny in 1999, with the loss of large barrel sponges *Xestospongia muta* (Klomp and Kooistra, 2003). The 2005 mass bleaching event proved devastating, with 70–80% of colonies bleached and live coral cover halved from about 30% to less than 15% by 2008, while macroalgal cover surged from 40% to nearly 60% (Bouchon et al., 2008). By 2014, surveys indicated coral cover had fallen further to 4.9%, with macroalgae dominating and reefs characterized by low three-dimensional structure (Debrot et al., 2014). This degradation continued, and by 2015 coral cover reached a historic low of 5%, marking a shift to macroalgal dominance (>25%) with limited prospects for recovery due to low *Diadema antillarum* urchin densities, fishing pressure on herbivorous fish, and the near absence of large groupers (De Graaf et al., 2015). Subsequent assessments confirmed this trajectory: coral cover remained at ~5% from 2015 to 2017 while macroalgal dominance increased to 36.6%, and herbivorous fish biomass declined by 58% over 18 years. Reef-building corals like *Orbicella* spp, *Pseudodiploria strigosa* and *Montastrea cavernosa* were largely replaced by smaller, stress-tolerant species such as *Siderastrea siderea* and *Porites astreoides,* probably due to the cumulative impact of the 2005 bleaching event, structural damage by multiple hurricanes, and low grazing intensity (Kitson-Walters, 2017). Collectively, these trends illustrate a long-term shift from coral-dominated to macroalgae-dominated reefs, with the ecosystem now in critical condition. Conservation efforts of the reefs should focus on the St. Eustatius National Marine Park where the highest diversity of corals can be found (Hoeksema et al. 2022).



↑ **Figure 2.39.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Sint-Eustatius from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 40

Sint Maarten

Authors: Laëtitia Mathon, Ildiko Gilders, Bert W. Hoeksema.

Data contributors: Tadzio Bervoets, Kristi D. Klomp*, Alice Deschamps*, Leslie Hickerson, Laëtitia Mathon, Chris Moses*, Sascha Steiner*, Anna Venema.

*AGRRA data contributor(s).

INTRODUCTION

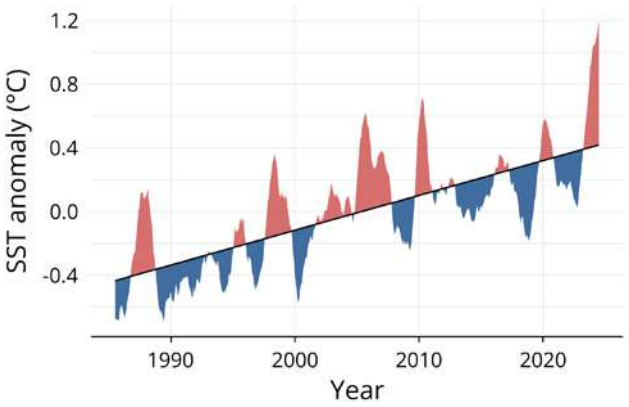
Sint Maarten, in the northeastern Caribbean's Lesser Antilles, shares its island with the French territory of Saint-Martin. The Dutch side covers 34 km², with territorial seas of 532.88 km² and an exclusive economic zone of 14.32 km² that hosts key ecosystems such as coral reefs, mangroves, and seagrass meadows, which provide coastal protection and carbon storage. Coral reef extent is about 4 km², only 0.015% of the GCRMN Caribbean total, although recent studies have estimated a slighter greater area (Mathon, 2025). Reefs, mainly along the southern coast at 8–15 m depths, are shaped by strong currents and dominated by gorgonians and massive hard corals, with fringing reefs also around nearby islets (Klomp and Kooistra, 2003). The marine environment supports 64 IUCN Red List species, 8 CITES Appendix I species, and 244 Appendix II species (MacRae and De Meyer, 2020).

Home to 42,000 people and with a density of 1,221/km², Sint Maarten is the most densely populated island in the Caribbean (Department of Statistics, 2021). Tourism dominates, generating 80% of economic activity, with 1.8 million cruise and 0.5 million stay-over visitors yearly. Reef-related tourism produces an estimated USD $53 million annually (Bervoets, 2010, updated), while the annual value of fisheries in Sint Maarten is estimated to be ~USD $23 million. When considering carbon sequestration, tourism, and fishery services of Sint Maarten's marine and coastal ecosystems, the total economic value amounts to approximately US$76 million annually (Grant Thornton, 2025).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Sint Maarten was 27.52 °C. Over this period, SST increased by 1.01 °C, representing a warming rate of 0.026 °C per year (**Figure 2.39.1**). The marine heatwave of 2005 led to large coral cover losses, due to a widespread bleaching event (Bouchon et al. 2008). Corals in Sint Maarten suffered from another strong bleaching event starting in 2023. Impacts of this event are still being observed, with *Acropora palmata*, *Porites astreoides*, *P. divaricata,* and *Millepora complanata* being the most affected species (Mathon, 2025).

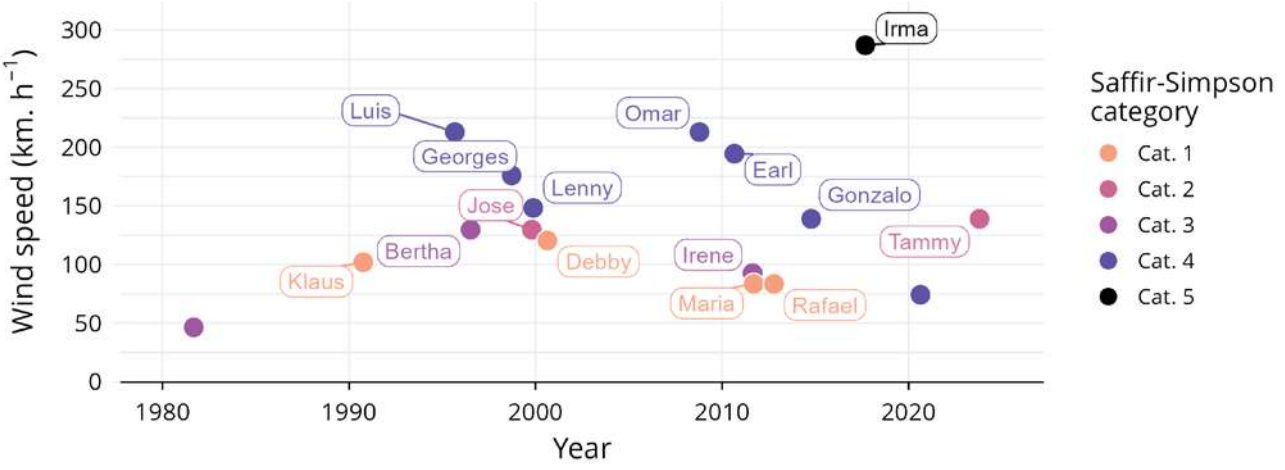


↑ **Figure 2.39.1.** Average Sea Surface Temperature (SST) anomaly from 1985 to 2024 over coral reefs of Sint Maarten. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Hurricanes

Between 1980 and 2024, a total of 25 tropical storms passed within 100 km distance from a coral reef of Sint Maarten, and of these 13 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.39.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed 1 km from a coral reef with sustained wind speed of 287 km.h^{-1}.

↓ **Figure 2.39.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Sint Maarten. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Sint Maarten are not represented, although they can have had an impact.



Coastal development and tourism

Coastal development in Sint Maarten has been closely tied to tourism, the island's main economic driver. Between 1956 and 2020, vegetation cover declined by 25% due to urbanization and expanding tourism infrastructure (De Freitas, 2020). Cruise tourism has been particularly significant: Sint Maarten received a peak of 2 million passengers on 692 vessels in 2014, before declining during COVID-19. Recovery has since been strong, with 1.3 million passengers from over 500 vessels in 2023 (Department of Statistics, 2023). Such growth has intensified habitat loss, erosion, and ecological degradation along the coast (**Figure 1.2.7**).

Pollution and nutrient runoff

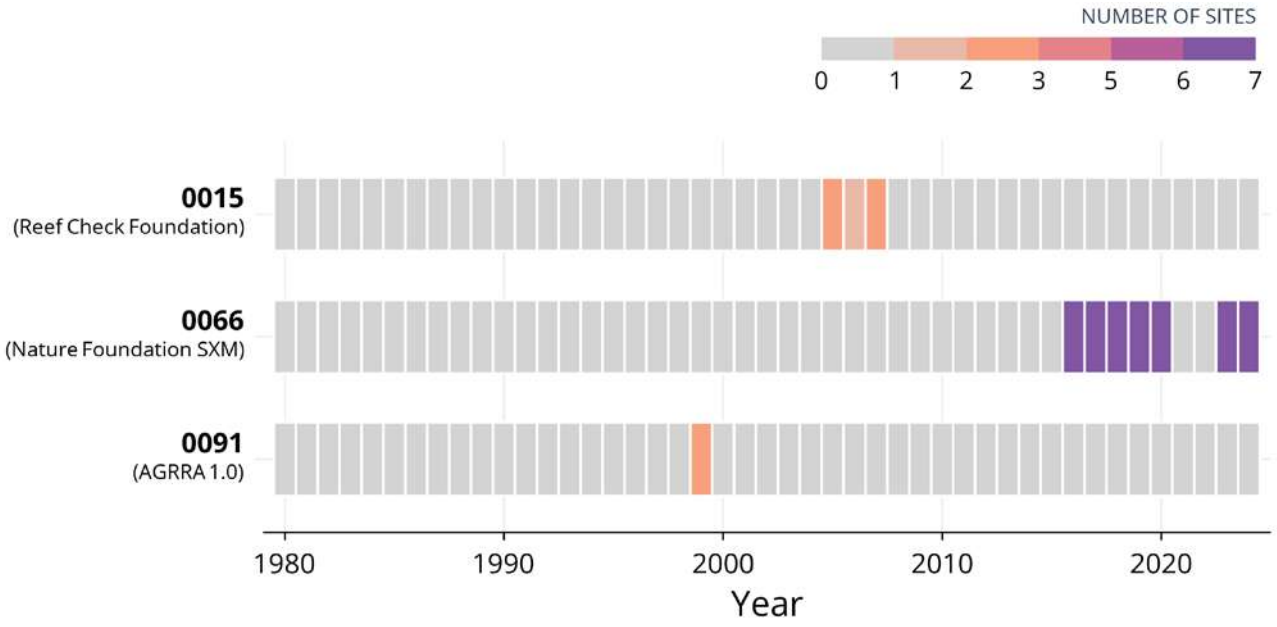
Pollution and nutrient runoff further stress marine environments. Elevated levels of heavy metals, nitrogen, and phosphorus at high-impact sites like Fresh Pond highlight the scale of urban pressures (Dungen, 2023). These inputs contribute to eutrophication, lowering water quality and disrupting ecosystem balance. Meanwhile, plastic pollution persists, driven by overburdened landfills, debris washed ashore, and littering by residents and tourists (de Bettencourt and Imminga-Berends, 2015; Bosker et al., 2018).

Fishing

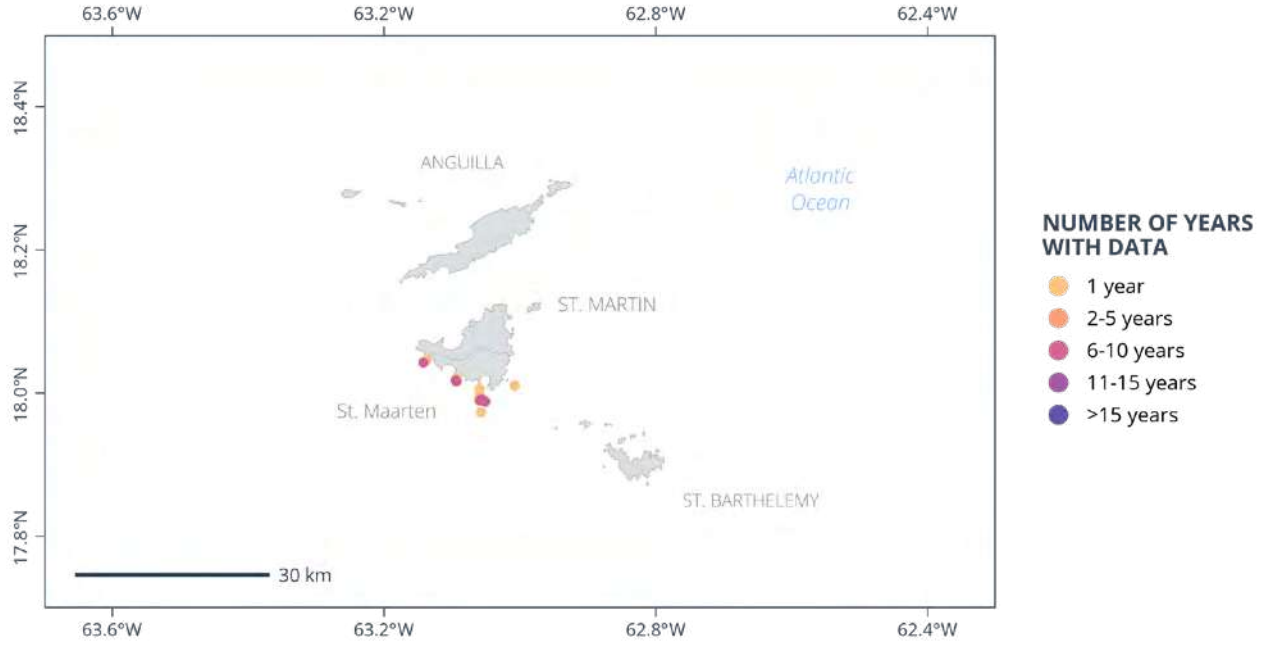
Fisheries also shape marine ecosystems in Sint Maarten. Artisanal and recreational fisheries target species from lobster and conch to groupers, snappers, and pelagics. This alters the abundance of carnivores and herbivores, impacting reef health, though the Man of War Shoal Marine Park, established in 2010, provides some refuge for vulnerable populations (MacRae, 2011; Mathon, 2024).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.39.3**) encompassing 14 monitoring sites within the Man of War Shoal Marine Park and in unprotected areas (**Figure 2.39.4**). These data were collected through 61 surveys conducted between 1999 and 2024. Annual coral reef monitoring with the GCRMN protocol started in 2016, but some previous studies described the status of coral reefs in Sint Maarten and their threats (Nijkamp et al., 1995, NFSXM, 2010).

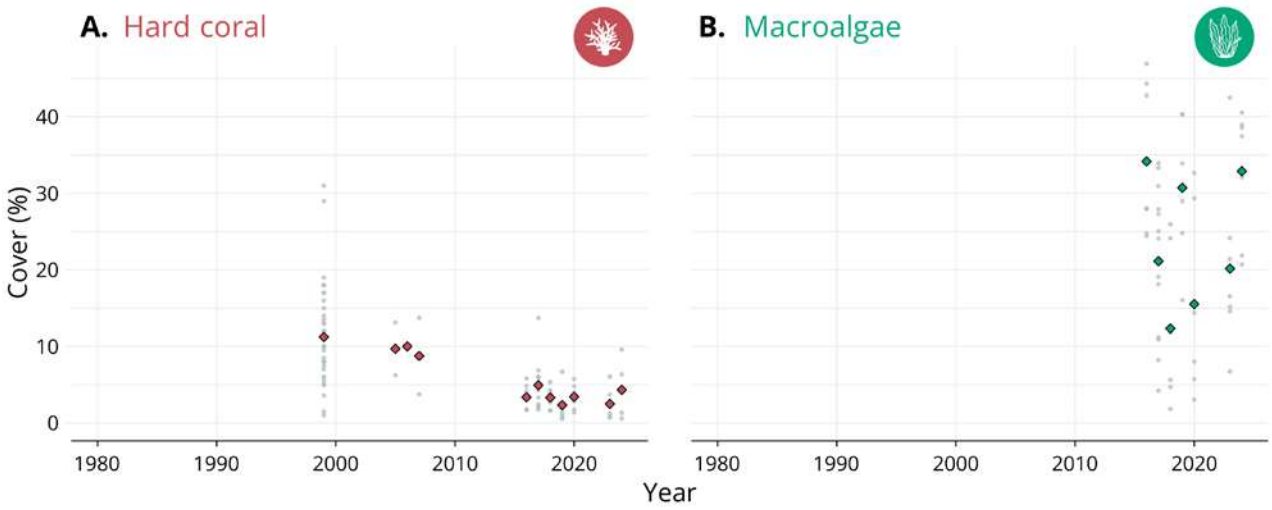


↑ **Figure 2.39.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.39.4.** Spatio-temporal distribution of benthic cover monitoring sites across Sint Maarten. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Over the past decades, coral reefs in Sint Maarten have been shaped by a combination of natural disturbances and increasing human pressures, marked by a decrease in hard coral cover and a strong increase in macroalgae and turf cover (**Figure 2.39.5**). Recent monitoring of Sint Maarten's coral reefs shows that overall species richness of fish and coral genera has remained relatively stable, but significant shifts in community structure and ecosystem health are evident. Inside the Man of War Shoal Marine Park, coral, sponge, and gorgonian cover, along with fish density and predatory fish biomass, are higher than in non-protected areas, reflecting the benefits of protection and greater habitat complexity. Outside protected zones, however, macroalgae dominate, indicating ecological imbalance.



↑ **Figure 2.39.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Sint Maarten from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

Post-hurricane assessments already documented early signs of stress: following Hurricane Lenny in 1999, the AGRRA protocol revealed widespread bleaching, with more than 23% of colonies affected throughout the then Netherland Antilles, and the highest levels on Sint Maarten (44%), alongside sedimentation and physical disturbance (Klomp and Kooistra, 2003). More recently, the catastrophic passage of Hurricane Irma in 2017 caused severe ecological impacts. Coral reefs experienced mechanical breakage, sediment redistribution, and sharp declines in coral cover, which fell from 6.13% to 3.74% after Irma and Maria (Mitchell, 2018). Monitoring showed shifts in fish biomass, species composition, and trophic groups, with carnivores increasing in biomass and moving into the Marine Park (Mathon, 2024). Widespread bleaching events in 2023 and 2024, driven by rising sea temperatures, have severely affected reefs from shallow to deeper sites. Notably, Indigo Bay supported one of the last resilient populations of the endangered elkhorn coral (*Acropora palmata*) in Sint Maarten, but most of the colonies recently died likely due to bleaching and heavy sedimentation from a resort development in the bay.

According to the Healthy Reefs Initiative indicators, live coral cover is critically low and macroalgae cover excessively high across all monitored sites, and herbivorous fish populations remain critically low (Mathon, 2024).

Reef restoration efforts, such as the deployment of Oceanite artificial reefs, show promising potential for enhancing biodiversity and supporting early reef recovery (Higgins et al., 2024), but long-term success depends on strengthening systematic monitoring programs, which currently lack the scope and standardized indicators needed to fully guide policy and management (Verweij, 2015).

CHAPTER 41

Trinidad and Tobago

Authors: Jahson B. Alemu I, Jennie Mallela, Jérémy Wicquart.
Data contributors: Jahson B. Alemu I, Jennie Mallela.

INTRODUCTION

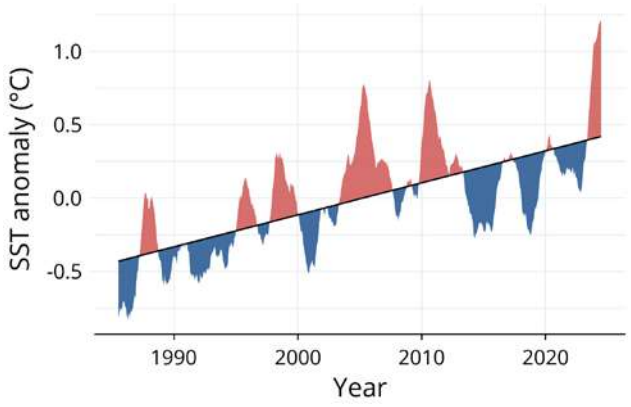
Trinidad and Tobago, the southernmost nation in the Caribbean, lies just off the northeastern coast of Venezuela and south of Grenada. The country is composed of two main islands - Trinidad, the larger, and Tobago, smaller - along with several smaller islets (**Figure 2.41.4**). Trinidad and Tobago has a population of 1.37 million people, with ~60,000 of these residents in Tobago. The two islands differ economically: Trinidad's workforce is dominated by the energy and industrial sectors, which includes petroleum, natural gas, and petrochemicals. Tobago, by contrast, relies heavily on reef-linked livelihoods such as tourism, small-scale agriculture, and fisheries, making the health of its coastal ecosystems central to both biodiversity and socio-economic wellbeing.

The coral reefs are relatively limited, covering an area of approximately 70 km², which represents nearly 0.3% of the Caribbean's total coral reef extent. However, these reefs are unique within the Caribbean. Their proximity to the South American mainland means that coastal waters are heavily affected by runoff from two of the world's largest rivers: the Orinoco and Amazon Rivers (Mallela and Harrod 2008). Their combined river runoff brings pulses of freshwater, sediment, and nutrients to the islands. As a result, local reefs have adapted to turbid water conditions and support a limited range of sediment-tolerant coral species and generally show lower diversity compared with other Caribbean coral reef systems (Mallela et al., 2010, Alemu and Mallela 2021). The reefs also provide a refuge to IUCN Red List coral species which include *Acropora cervicornis* (critically endangered), *Acropora palmata* (critically endangered) and *Dendrogyra cylindricus* (critically endangered).

THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of Trinidad and Tobago was 27.72 °C. Over this period, SST increased by 1.01 °C, representing a warming rate of 0.026 °C per year (**Figure 2.41.1**).



↑ **Figure 2.41.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Trinidad and Tobago. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

Coral core records from Tobago (1989-2008) have documented site specific (localised) differences in bleaching susceptibility. Coral colonies of *Colpophyllia natans* on reefs with better water quality proving to be more resilient, and less prone to bleaching and/or growth anomalies, during periods of thermal stress (Mallela et al., 2016). Other studies in Tobago have also documented significant coral recruitment decline following thermal anomalies/ heat stress events (*e.g.* bleaching) and tropical storms (Mallela and Crabbbe, 2009) followed by

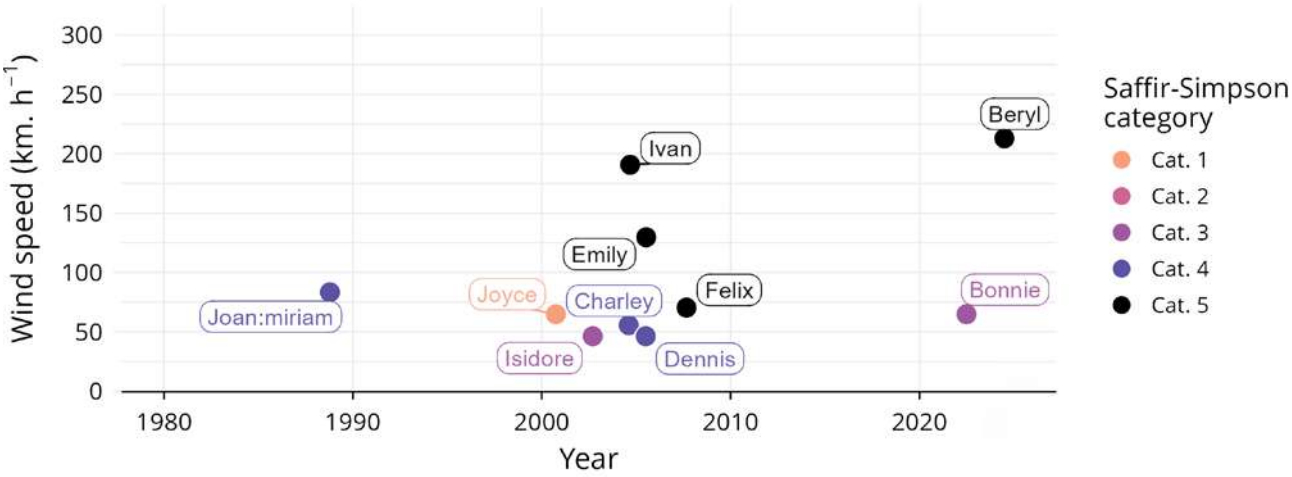
an increased prevalence of coral disease on major framework builders after bleaching events (Alemu and Clemente 2016; Mallela et al., 2010, Mallela and Parkinson 2008). Indeed, settlement plate studies demonstrated that coral recruitment rates remained low more than two years after bleaching events.

Hurricanes

Between 1980 and 2024, a total of 20 tropical storms passed within 100 km distance from a coral reef of Trinidad and Tobago, and of these 3 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.41.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Beryl, in 2024, which passed 67 km from a coral reef with sustained wind speed of 213 km.h^{-1}. An assessment of the impact of hurricanes and tropical storms from 1980-2006 on coral recruitment patterns and the survival of juvenile coral colonies on Tobago's reefs demonstrates significant declines following these events (Mallela and Crabbe, 2009).

The combined impacts of bleaching and hurricanes are also of concern for critical reef building species. *Orbicella faveolata* (formerly *Montastraea faveolata*) is a primary framework-building coral on Tobago's reefs. However, its live cover and recruitment potential declined sharply after the 2005 bleaching event and was further reduced by an outbreak of yellow band disease (Mallela et al 2010). No recruitment of *O. faveolata* was observed after the combined impacts of Hurricane Ivan (2004) and the 2005–2006 bleaching event (Mallela and Crabbe 2009). Subsequently, assessment of juvenile colonies of *Orbicella* following the 2010 bleaching event also revealed low numbers (Buglass et al., 2016).

↓ **Figure 2.41.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Trinidad and Tobago. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Trinidad and Tobago are not represented, although they can have an impact (*e.g.* Mallela and Crabbe, 2009).



Coastal Water Quality

Trinidad and Tobago's coral reef development reflects strong land–sea interactions. These southern Caribbean reefs experience seasonal inputs from the Orinoco–Amazon plumes (freshwater, sediments, nutrients), with reef communities dominated by sediment-tolerant coral assemblages (Muller-Karger et al 1988, Laydoo et al., 1998, Mallela, Parkinson, and Day 2010). Historically across southwest Tobago, nitrogen-driven eutrophication within the Buccoo Reef Complex, has been linked to a proliferation of turf and macroalgae overgrowing framework building corals and suppression of hard coral cover (Lapointe et al., 2003; Lapointe et al., 2010). Stable-isotope tracers ($\delta^{15}N$) have identified sewage-derived nitrogen with values highest near outfalls in Buccoo Reef, intermediate along the populated leeward coast, and lowest at oligotrophic reference sites off Little Tobago (*e.g.* Black Jack Hole). Spatial gradients strengthen in the wet season, and $\delta^{15}N$ correlates negatively with coral cover, linking wastewater enrichment to reduced coral growth (Lapointe et al. 2003, Mallela and Harrod 2008). Early studies flagged how macroalgal dominance persisted despite abundant herbivores inside the Marine Protected Area, highlighting nutrient

enrichment as the algal driver. Island wide stable isotope analyses further demonstrated distinct coastal food-web baselines around Trinidad versus Tobago, consistent with differing terrestrial inputs and human pressures (Mallela and Harrod 2008). Recent reef assessments confirm decades of deteriorating water quality, algal phase shifts, reduced recruitment, and disease outbreaks as key threats to ecosystem resilience (Alemu I and Mallela, 2021). Management should prioritise managing watershed runoff. In particular, reef nutrient and sediment inputs from agricultural runoff, land clearance, and coastal development, which reduce water clarity and cause high levels of sedimentation on reef benthos. These pressures are compounded by inadequate wastewater treatment, with many households relying on "soakaways" that release nutrient-rich effluent to nearshore waters (Lapointe et al., 2010). Management should include the implementation of long-term water-quality monitoring with trigger values that initiate real world management outcomes. Additional reefs that would benefit from targeted protection include the relatively oligotrophic reefs (*e.g.*, Speyside/Little Tobago) (Mallela, Hetzinger, and Halfar, 2016).

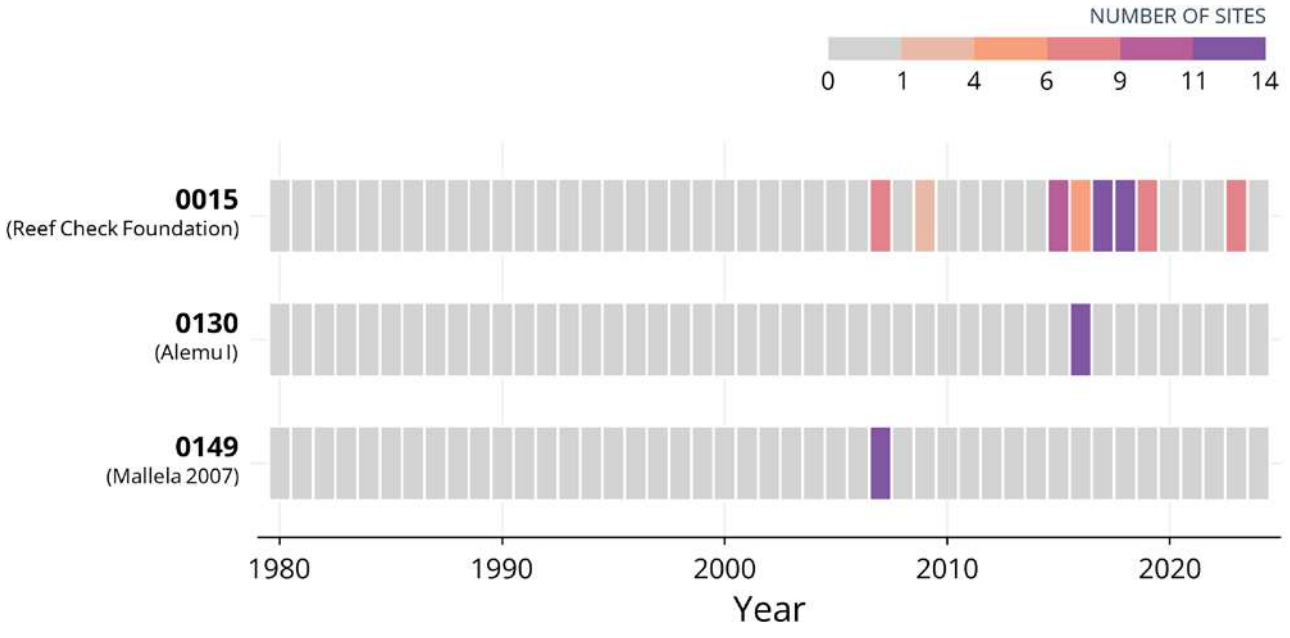
Invasive Species

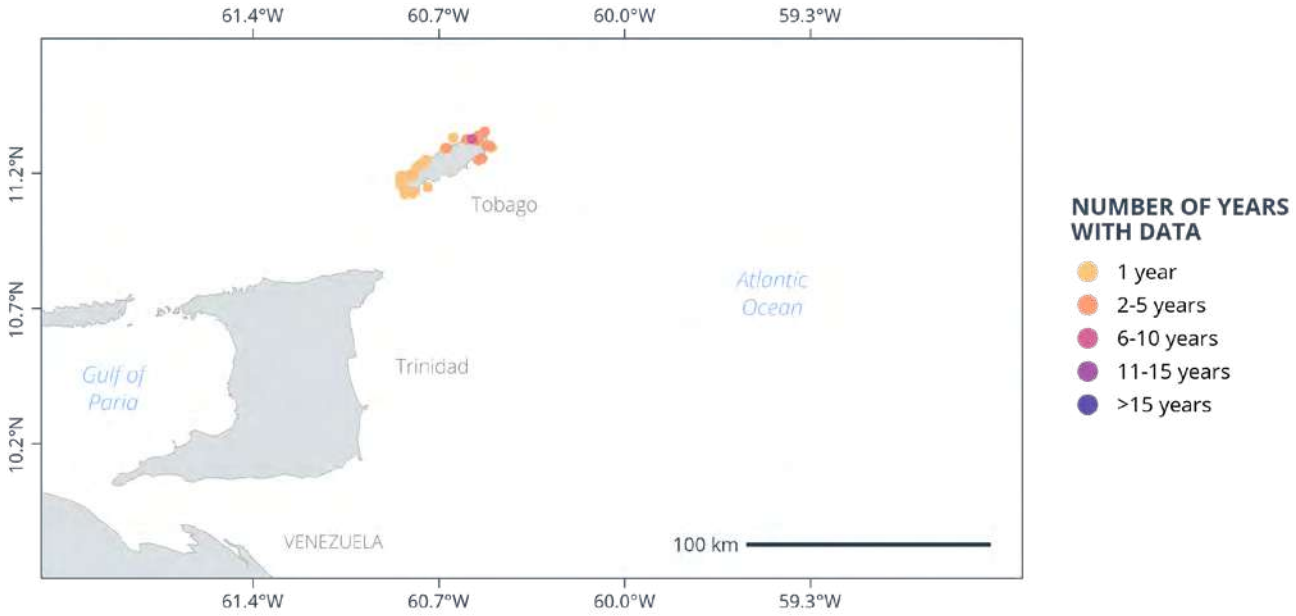
The introduction of the Indo-Pacific lionfish (*Pterois volitans* and *P. miles*) across Tobago's reefs represents one of the most recent and significant invasive pressures to reef resilience in recent decades. First documented locally in the mid-2010s (Alemu I, 2016), lionfish populations expanded rapidly due to their high reproductive output, lack of natural predators, and generalist diet. Stomach content analyses from the wider Caribbean reveal predation on more than 70 fish and invertebrate taxa, including ecological and commercially important species (Morris and Akins 2009; Green et al., 2011; Green and Côté 2014). While Tobago lacks long-term empirical datasets on reef fish population trends beyond initial baseline surveys, the aforementioned diet studies from the wider region demonstrate that lionfish readily consume juvenile herbivores, with the potential to weaken herbivory pressure on coral reefs, promote algal proliferation, and by extension, constrain coral recovery following disturbance. In addition to ecological effects, lionfish also pose risks to fishers and divers through venomous spines, complicating management responses. While culling programs and public awareness campaigns were initiated locally (Alemu I 2016), sustained monitoring and targeted removals in the last years have waned but remain critical to prevent long-term shifts in reef community structure.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.41.3**) encompassing 52 monitoring sites. These data were collected through 115 surveys conducted between 2007 and 2023.

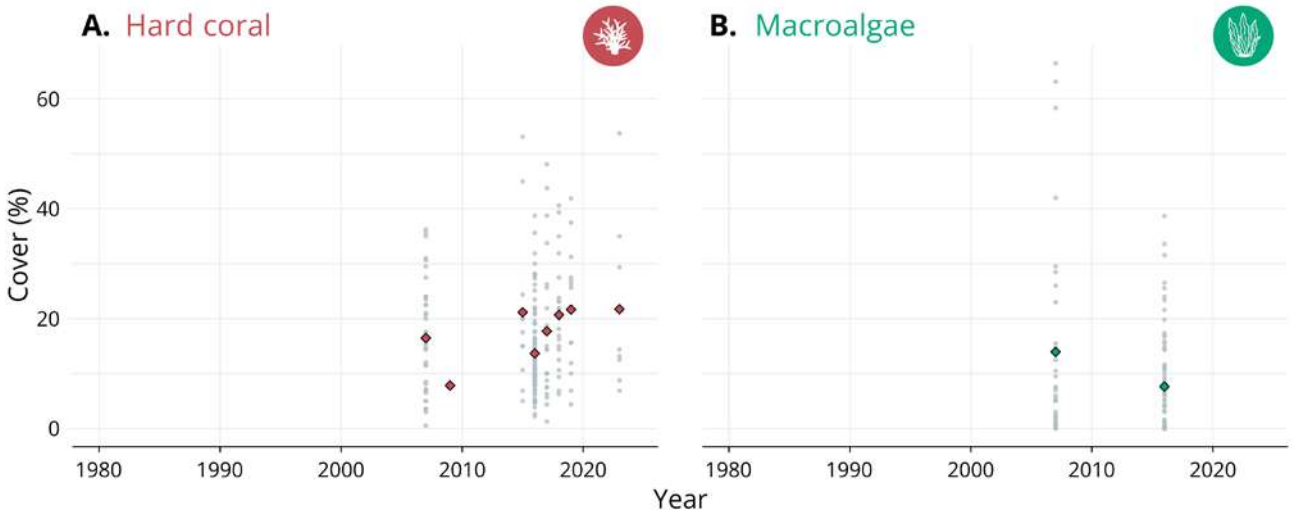
↓ **Figure 2.41.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.41.4.** Spatio-temporal distribution of benthic cover monitoring sites across Trinidad and Tobago. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

Multi-decadal monitoring of Tobago's reefs show that hard coral cover has declined from historical baselines of the 1980s, with the most pronounced losses following the 1998, 2005, and 2010 mass bleaching events (Alemu I and Mallela 2021). While some reefs have demonstrated partial recovery (Ganase and Lochan 2022; Buglass et al 2016), overall coral cover has remained low to moderate, with strong site level variability (Alemu I and Mallela 2021). Octocorals have become increasingly prominent on several reefs, while sponge cover has persisted at low levels across most sites, with only localized exceptions (Ganase and Lochan 2022; Alemu I and Mallela 2021).



↑ **Figure 2.41.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Trinidad and Tobago from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

CHAPTER 42

Turks and Caicos Islands

Authors: John DeBuysser, Louise Anderson, Jérémy Wicquart.

Data contributors: John DeBuysser, Casey Hermoyian*, Heidi Hertler, Carrie Manfrino*, Alexander R. McLeod, Rebecca Hamel Pepin, Bernhard Riegl*.

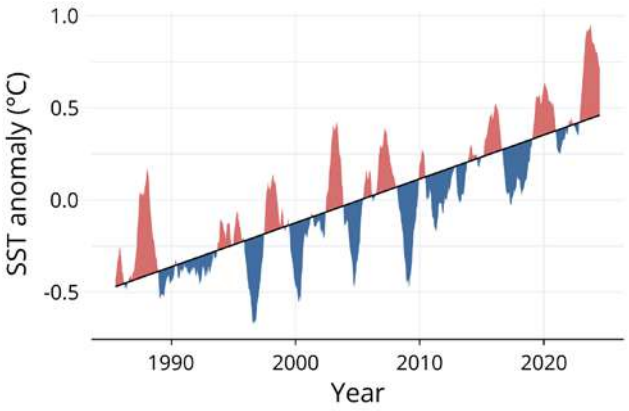
*AGRRA data contributor(s).

INTRODUCTION

The Turks and Caicos Islands, a British Overseas Territory in the Atlantic Ocean, lie southeast of the Bahamas and north of Hispaniola. The archipelago is composed of two island groups - the larger Caicos Islands and the smaller Turks Islands - comprising around 40 low-lying islands and cays, of which only a few are inhabited (**Figure 2.42.4**). Coral reef extent is about 343 km², which represents 1.4% of the Caribbean's total coral reef extent. Turks and Caicos Islands are characterized by clear, shallow waters overlying white carbonate sands (Logan and Sealey, 2013).

THREATS

Thermal regime



↑ **Figure 2.42.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Turks and Caicos Islands. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

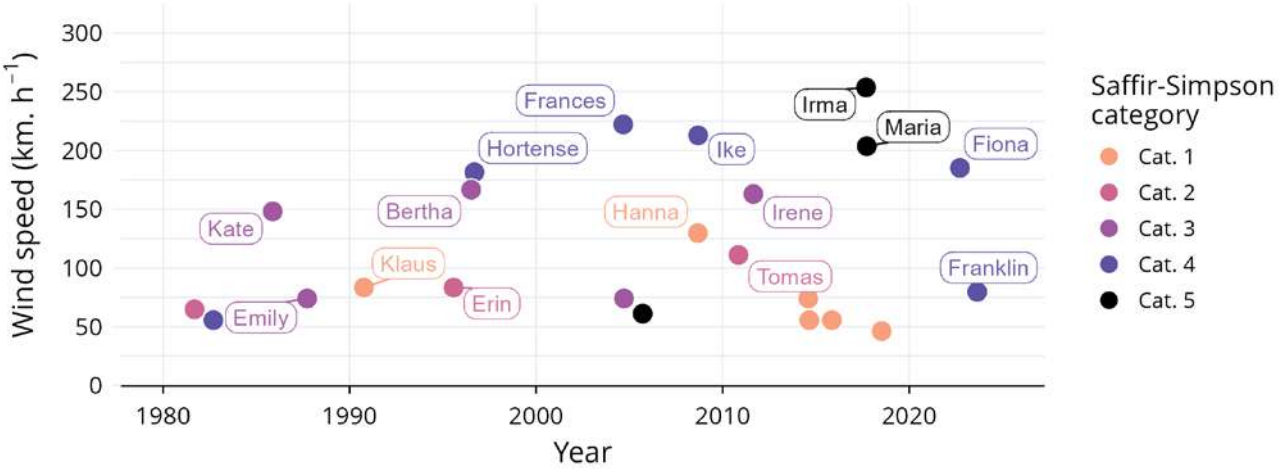
The long-term average in SST from 1985 to 2024 over coral reefs of Turks and Caicos Islands was 27.25 °C. Over this period, SST increased by 1.05 °C, representing a warming rate of 0.027 °C per year (**Figure 2.42.1**).

Regional and local conditions affect coral bleaching outcomes on South Caicos. The coral reefs fringe coastline, sand banks, and offshore cays which create a range of thermal conditions based on season and tidal flow. The main reef formation along the eastern and southern shores of the island closely matches the regional ocean temperature trends, whereas inshore reefs that are tidally connected with the sand bank system can experience temperatures 2°C warmer than ambient with an ebbing tide. Variability in local thermal regime can in part explain some of the areas coral bleaching response in the four global bleaching events, locally observed in 1998, 2005, 2015-2016, and 2023-2024 (Knipp et al., 2020). The first three events recorded stress and some mortality within the coral assemblage, and the last event in 2023-2024 had the greatest mortality effect, especially noted with *Acropora sp., Orbicella sp., Agaricia sp.,* and *Millepora sp.*

Hurricanes

Between 1980 and 2024, a total of 28 tropical storms passed within 100 km distance from a coral reef of Turks and Caicos Islands, and of these 11 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.42.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Irma, in 2017, which passed 13 km from a coral reef with sustained wind speed of 254 km.h^{-1}.

↓ **Figure 2.42.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Turks and Caicos Islands. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Turks and Caicos Islands are not represented, although they can have had an impact.



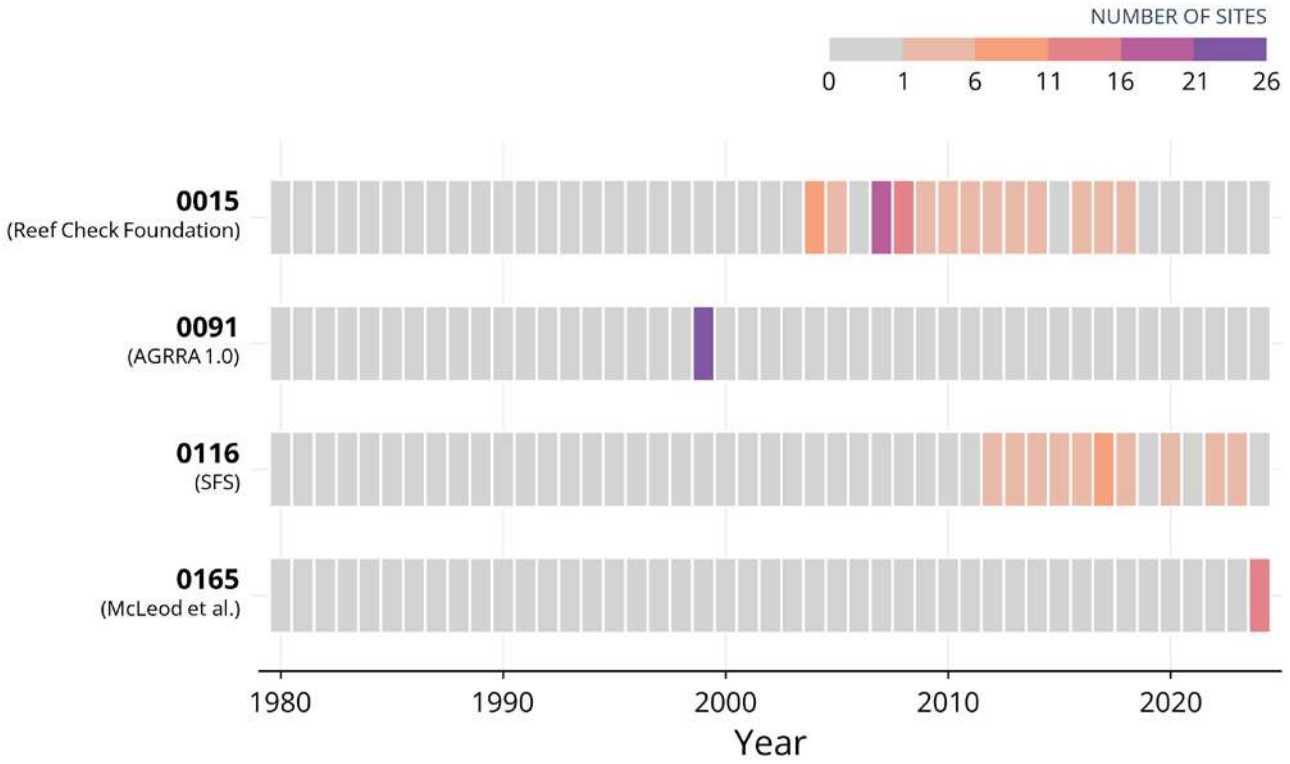
Coral diseases

Stony Coral Tissue Loss Disease (SCTLD) was observed in South Caicos in the fall of 2019 and quickly decimated the hard coral population in 2020 (Heres et al., 2021). By 2021, no living colonies of *D. cylindrus, M. meandrites, C. natans, and E. fastigiata* were observed. Some *Orbicella sp.* colonies were mildly affected, but overall the initial disease outbreak did not threaten this genus as much as others. From 2022 to 2025 there were few reported or observed cases of SCTLD in South Caicos, potentially due to previous mortality of susceptible individuals. In 2024, *M. meandrites* and *E. fastigiata* recruits and juvenile colonies began reappearing in appreciable numbers across all reef sites, and in 2025 the first *D. cylindrus* recruit and juvenile colony were observed.

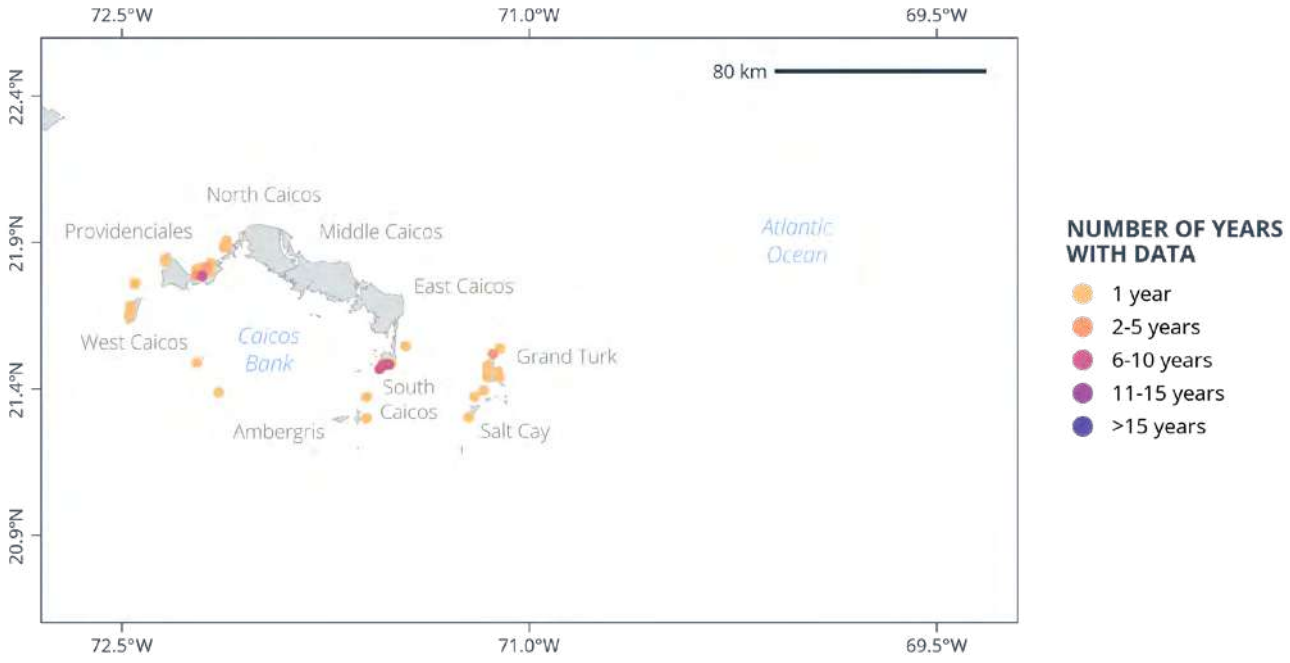
TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 4 datasets (**Figure 2.42.3**) encompassing 87 monitoring sites. These data were collected through 145 surveys conducted between 1999 and 2024.

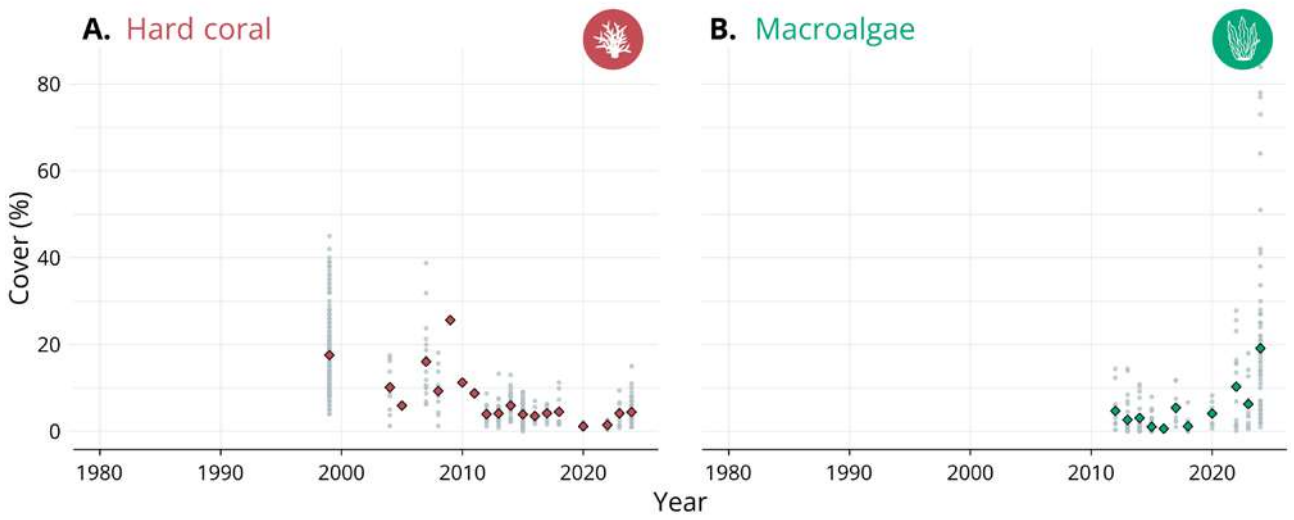
The School for Field Studies (SFS) operated a field research station on South Caicos, from 1990 to present. Visiting scientists and researchers have collected discrete surveys, with initial publications including Mumby (1998), Steiner (1999), and Riegl (2003). Heidi Hertler, Ph.D. and Center Director for the past 13 years, has compiled a reef benthic assessment data series beginning in 2012 at four South Caicos reef sites, with each site surveyed at 10m, 20m, and 30m depths (**Figure 2.42.3**). Surveys are typically conducted twice per year, in spring and fall, though hurricanes and the COVID pandemic did prevent data collection in some semesters. Overall, South Caicos reefs have lower coral cover compared to other islands of the country, potentially due to its windward exposure and consistent hurricane impacts (Mumby, 1998; Steiner, 1999, Riegl, 2003).



↑ **Figure 2.42.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).



↑ **Figure 2.42.4.** Spatio-temporal distribution of benthic cover monitoring sites across Turks and Caicos Islands. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.



↑ **Figure 2.42.5.** Annual means of observed data (*i.e.* monitoring data) for hard coral (A), and macroalgae (B) in Turks and Caicos Islands from 1980 to 2024. Grey dots in the background represent the mean values of individual transects conducted each year.

CHAPTER 43

United States Virgin Islands

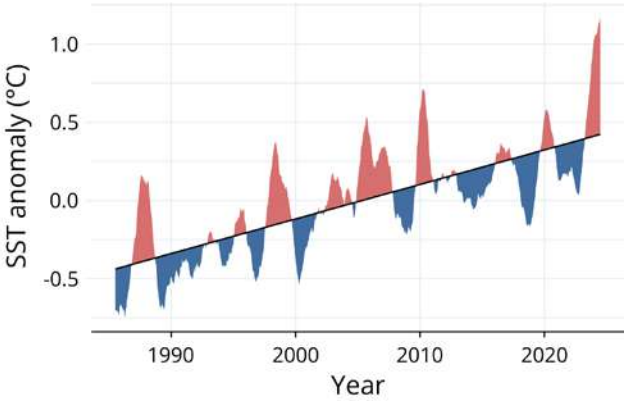
Authors: Tyler B. Smith, Richard S. Nemeth, Erica K. Towle, Shay Viehman, Lauren Olinger, Jérémy Wicquart.

Data contributors: Peter J. Edmunds, Sarah Heidmann, Lonnie Kaczmarsky*, Nicole Kellum*, Nicole Krampitz, Brianne Loran*, Peiro Mannoni*, Richard S. Nemeth*, Lauren Olinger, Jennifer Osborn*, Adam Quandt*, Laurie Requa*, Paige Rothenberger*, Tyler B. Smith, Mayra Suarez-Velez*, Erica K. Towle.

*AGRRA data contributor(s).

INTRODUCTION

The United States Virgin Islands (USVI), an unincorporated territory of the United States located in the Lesser Antilles, lie east of Puerto Rico and west of the British Virgin Islands. The territory is composed of three main islands - St. Thomas, St. Croix, and St. John - along with several smaller islands and cays such as Water Island, Hassel Island, and Buck Island (**Figure 2.43.4**). Coral reef extent is about 134 km², which represents nearly 0.5% of the Caribbean's total coral reef extent. When mesophotic reefs are included, Smith et al. (2019) estimated the total coral reef extent of the USVI at 259 km².



↑ **Figure 2.43.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of the United States Virgin Islands. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

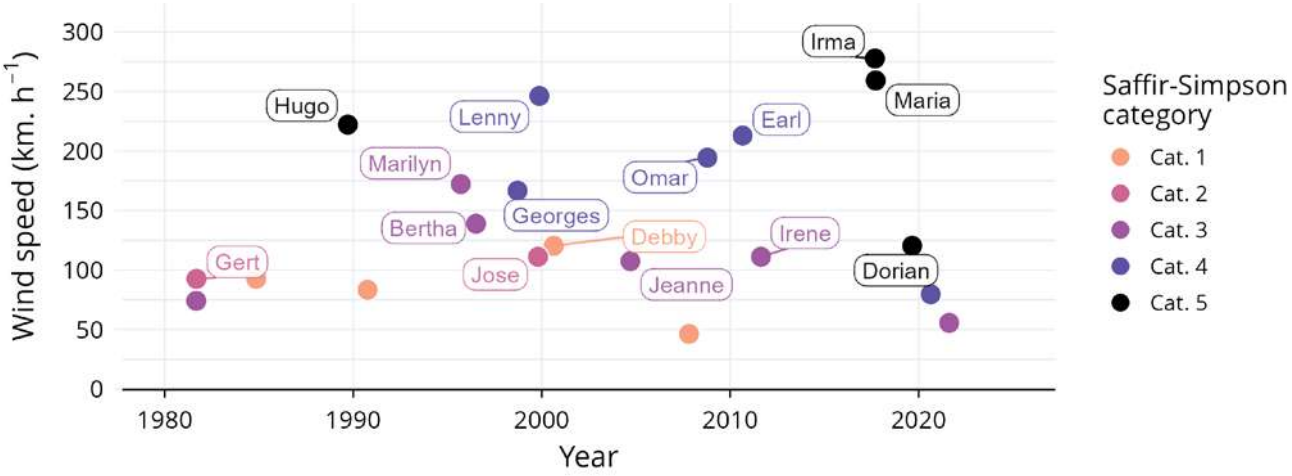
THREATS

Thermal regime

The long-term average in SST from 1985 to 2024 over coral reefs of the United States Virgin Islands was 27.66 °C. Over this period, SST increased by 1.02 °C, representing a warming rate of 0.026 °C per year (**Figure 2.43.1**). Mass bleaching events leading to substantial coral mortality occurred in 2005 (15.4 Degree Heating Weeks; Smith et al. 2013), 2023 (18.7 DHW; Krampitz et al. 2023), and 2024 (22.33 DHW; Skirving et al., 2020).

Hurricanes

Between 1980 and 2024, a total of 32 tropical storms passed within 100 km distance from a coral reef of the United States Virgin Islands, and of these 14 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.43.2**). The hurricane with the highest sustained wind speed over the studied period was hurricane Irma, in 2017, which passed 24 km from a coral reef with sustained wind speed of 278 km.h^{-1}.



↑ **Figure 2.43.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in the United States Virgin Islands. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of the United States Virgin Islands are not represented, although they may have impacted reef conditions.

Coral diseases

Epizootics and other diseases have impacted corals directly or indirectly by affecting ecologically important reef associates. Notable coral-specific disease epizootics include white band diseases that affected elkhorn and staghorn (Acropora) corals beginning in the late 1970s (Gladfelter 1982), white plague or white syndrome diseases affecting large head corals intermittently for the past 30 years (Chaves-Fonnegra et al. 2021) and acutely after the 2005 bleaching event (Miller et al. 2009), and stony coral tissue loss disease which impacted a majority of coral species and caused large losses of colony density (Brandt et al. 2021). Epizootics also caused large reductions in a critical shallow water herbivore, the black-spined sea urchin (*Diadema antillarum*) across the Western Atlantic in 1982-83 and 2022 (Hewson et al. 2023).

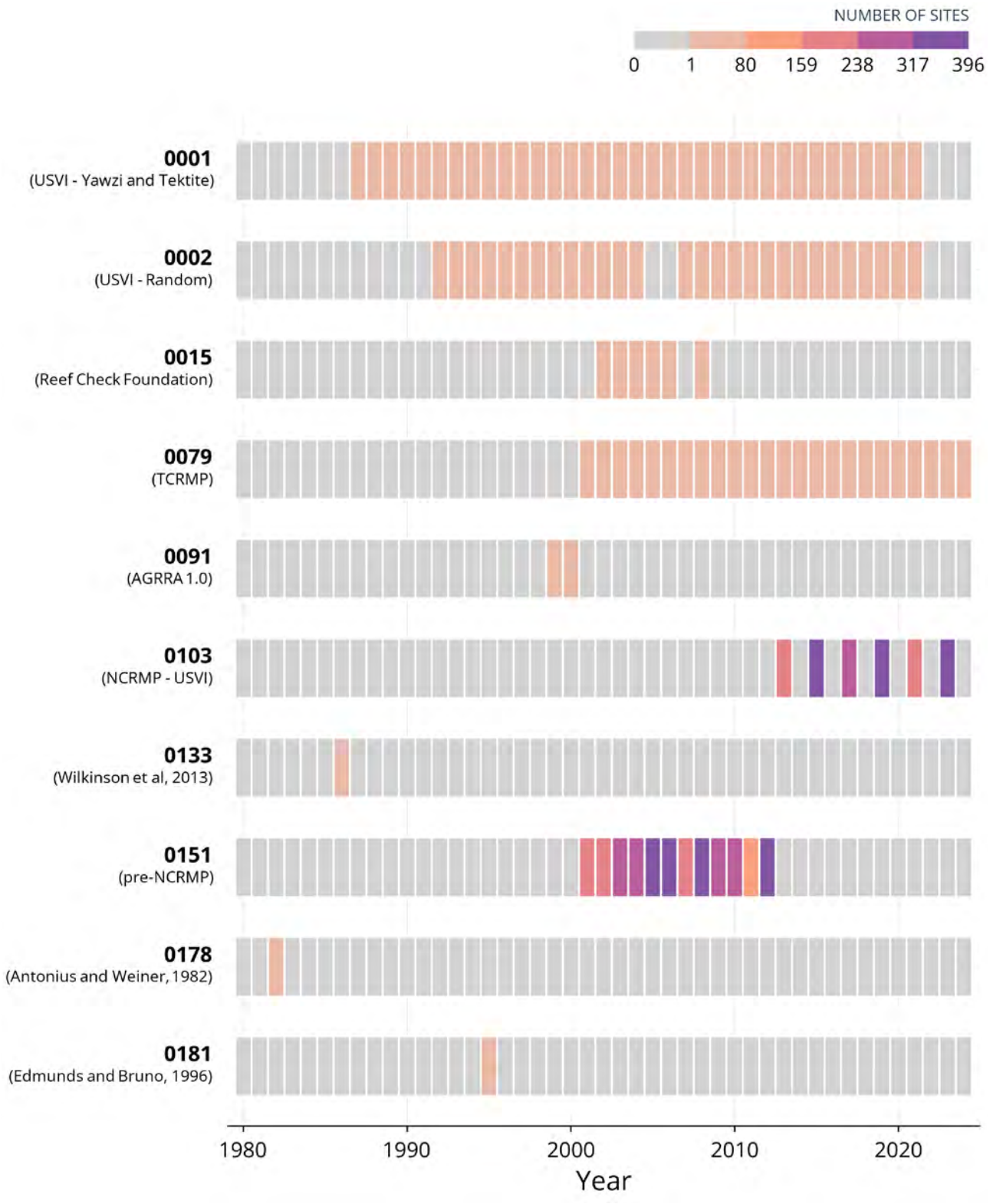
NCRMP data (included in this analysis) show that coral species that are highly susceptible to SCTLD, such as *D. cylindrus* and *M. meandrites*, have become increasingly rare, and for some species, as large colonies have died, population size distributions have shifted to higher proportions of smaller colonies than larger colonies (Viehman et al., 2025).

Nuisance alga *Ramicrusta*

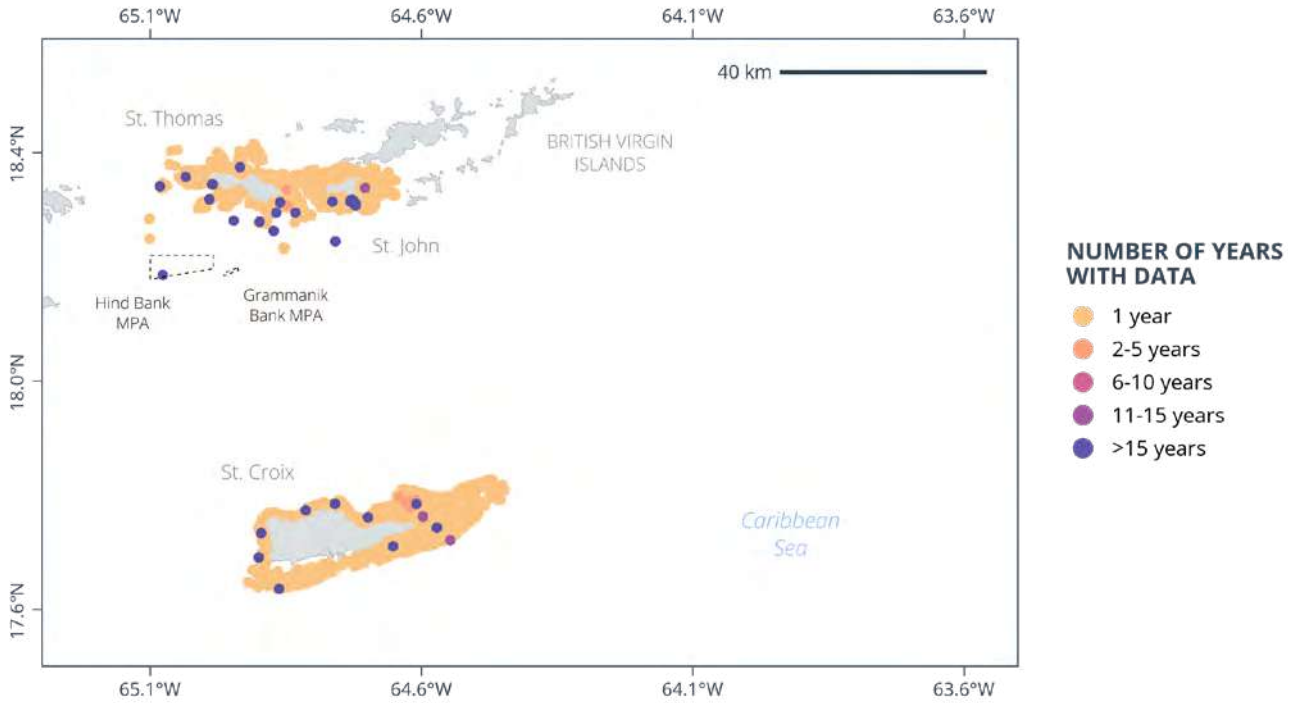
Since as early as 2004, the encrusting red algal genus *Ramicrusta* has been present and expanding on reef surfaces across coral reef surfaces from shallow water to as deep as 30 m (Smith et al. 2018). This alga can rapidly expand over reef surfaces, including living stony corals (Edmunds et al. 2023). In some sites coverage of the alga exceeds half the seafloor with growth that is largely unchecked by modern levels of reef herbivory (Williams 2022).

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 10 datasets (**Figure 2.43.3**) encompassing 5,183 monitoring sites. These data were collected through 6,071 surveys conducted between 1982 and 2024.



↑ **Figure 2.43.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).

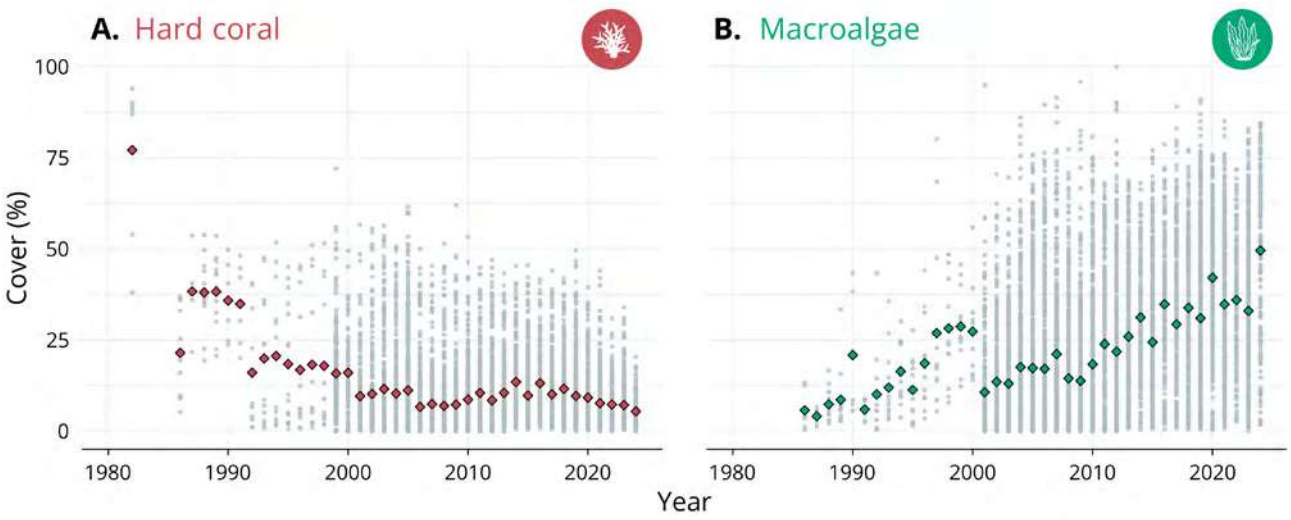


↑ **Figure 2.43.4.** Spatio-temporal distribution of benthic cover monitoring sites across the United States Virgin Islands. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset.

From 1980 to 2024, stony coral cover has declined from over 50% to less than 10%, with the more modern data having high confidence. Causes of the decline can primarily be attributed to acute thermal stress and coral bleaching in 2005 and 2023-24, the emergence and spread of stony coral tissue loss disease in 2019, and chronic stressors such as increasing mean ocean temperatures, disease, nuisance species, land-based sources of pollution, and overfishing of ecologically important species. In this same period macroalgal cover on reefs has increased from a minor contribution of less than 20% to over 30% across habitats.

The relative composition of coral reefs in the US Virgin Islands has therefore moved far away from baseline conditions at the start of modern ecological study. These reefs represent fundamentally changed habitats with likely altered ecological functions and provision of ecosystem services.

Although early indicators of the most recent global bleaching event were observed in many surveys in 2023-2024, continued population declines from this event are detectable in 2025, though 2025 data is not included in this report.



↑ **Figure 2.43.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in the United States Virgin Islands from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

TRENDS IN FISH POPULATIONS

The U.S. Virgin Islands contain some of the largest and oldest marine protected areas in the Caribbean and provide useful examples of trends in fish population characteristics over time. Three MPA's with long-term data include the St. Croix Buck Island Reef National Monument (BIRNM, est. 1961), the Virgin Islands National Park (VINP) on St. John (est. 1962), and the Virgin Islands Coral Reef National Monument (VICRNM) on St. John (est. 2001). BIRNM and VICRNM are not-take while VINP allows recreational and traditional subsistence fishing (Rogers and Beets 2001). In the past 40 years studies were conducted of fish populations inside and outside these MPA's. The studies showed that target fish families (*i.e.*, groupers, snappers, parrotfish) in all three MPAs either declined or showed no change in density and biomass relative to areas outside MPAs (Beets and Rogers 2002, Pittman et al. 2014). The lack of improvement may be due, in part, to MPA boundaries being established without suitable ecological design criteria (Monaco et al. 2007), no enforcement, differences in reef structure or viability of fish spawning aggregations (FSA) (Kadison et al. 2017, Heidmann et al. 2024a). Other assessments of MPA's that specifically protected FSA sites in the USVI (*i.e.* Red Hind Bank and Grammanik Bank in St. Thomas, and Mutton Snapper and Lang Bank in St. Croix) have shown mixed success. St. Thomas FSA's had significant improvements in abundance and biomass of spawning adults (Nemeth 2005, Rosemund et al. 2022), while St. Croix FSA's showed no change or continued decline due to poaching (Nemeth et al. 2007, Heidmann et al. 2024b).

Case study n°8

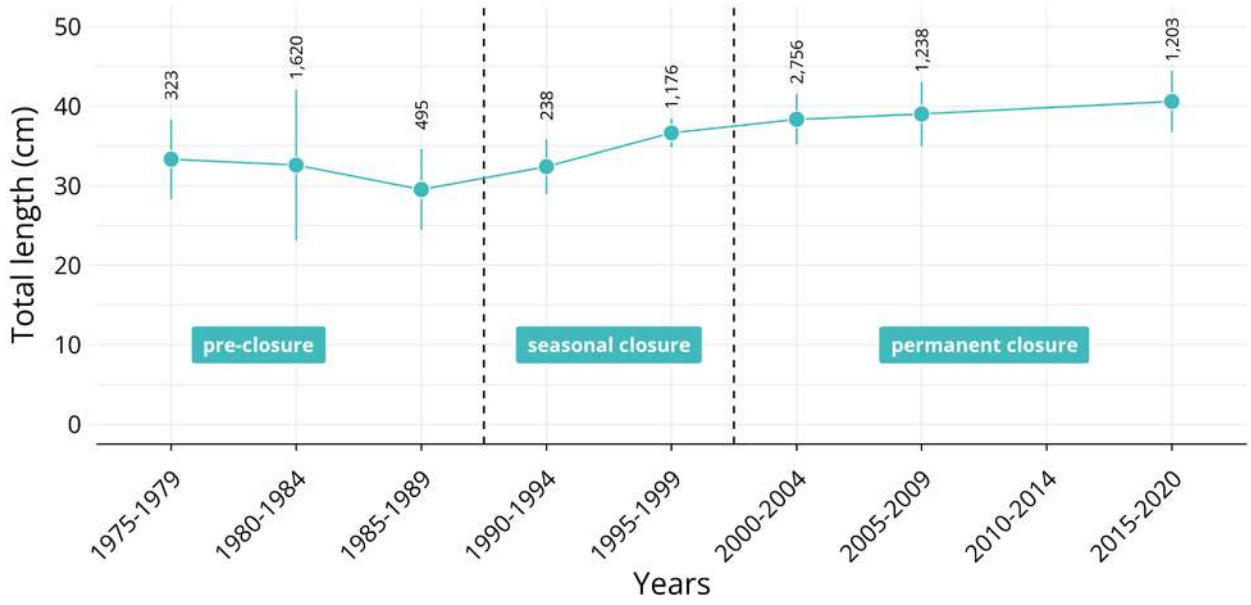
Forty years of research and monitoring documents the success of a large marine protected area in rebuilding a red hind spawning aggregation in the U.S. Virgin Islands

Authors: Richard S. Nemeth, James Beets, Alan M. Friedlander, Elizabeth Whiteman, Cecil Jennings, R. Claire Rosemond, and Scott Heppell

Many species of reef fishes form large fish spawning aggregations (FSA) during specific lunar periods for annual reproductive events. The predictability of the timing and location of FSAs makes them extremely vulnerable to fishing pressure. Several management approaches can be used to protect FSA sites including market closures, gear restrictions, closed seasons, and seasonal or permanent fishery closed areas.

In the United States Virgin Islands, the Red Hind Bank Marine Conservation District (MCD, **Figure 2.43.4**), which encompasses an FSA site, was closed seasonally in 1990 and then permanently in 1999 in response to declines in the mean size and landings of red hind grouper (*Epinephelus guttatus*) (Beets and Friedlander, 1992). Red hind replaced Nassau grouper (*E. striatus*) in the fishery after the collapse of the Nassau grouper population a decade earlier (Olsen et al., 1978; Nemeth et al., *in review*). Both fishers and managers agreed that protection of the red hind FSA site was needed. The large (42 km²) MCD was designed to protect the red hind spawning population (**Figure 1**) from fishing pressure but also protected critical spawning habitat of the extirpated Nassau grouper. The research and monitoring that led to these regulations and subsequent understanding of management effectiveness occurred over nearly a 40-year period.

↑ **Figure 1.** Red hind (*Epinephelus guttatus*) aggregating for spawning. Credit: R. Nemeth



↑ **Figure 2.** Change in mean length (± SE) of spawning red hind (Epinephelus guttatus) from 1975 to 2019 in response to seasonal and permanent closed area that protected a spawning aggregation site in St. Thomas, U. S. Virgin Islands. Data are aggregated by 5-year periods when one to four annual samplings were conducted. The number above the points indicates sample size of red hind.

Early indications of red hind population decline were observed in fishery landings as a change in the size structure of the population (Beets and Friedlander, 1992). The Red Hind Bank MCD was established in 1990 in response to significant declines in mean red hind length and loss of the largest size classes (predominately males) which resulted in a highly skewed population sex ratio. Within seven years, mean red hind total length (TL) increased by 7.1 cm (**Figure 2**; Beets and Friedlander, 1999). By 2003, after 10 years of seasonal protection and three years of permanent protection, red hind density and biomass both increased by >60% (Nemeth, 2005). By 2020, following 30 years of protecting the spawning population, the overall TL of red hinds had increased by >35% (**Figure 2**) and mean TL of females was 39.6 cm (Rosemond et al., 2022), indicating an increase in potential fecundity (Whiteman et al., 2005). Overall, the length of red hinds increased significantly (p-value < 0.0001) during the 10 years of seasonal closure from 1990 to 2000 and continued to increase in length (p-value < 0.0001) during 10 years of permanent protection. A length-based stock assessment of the red hind spawning population in 2020 showed that the spawning potential ratio increased to its highest observed level (0.49) and above the level considered sustainable for most fisheries (Rosemond et al., 2022). The management decisions to protect spawning aggregation sites have greatly benefited the red hind population and local fisheries. These studies emphasize the need for long-term data sets and their importance for justifying management actions to the fishing community. In the U.S. Virgin Islands, fish are traditionally eaten as whole plate-sized fish. Today, one can hear fishers complain that many red hinds they catch have outgrown the size of plates and must be fileted, which is a good problem to have regarding red hind population dynamics.

Case study n°9

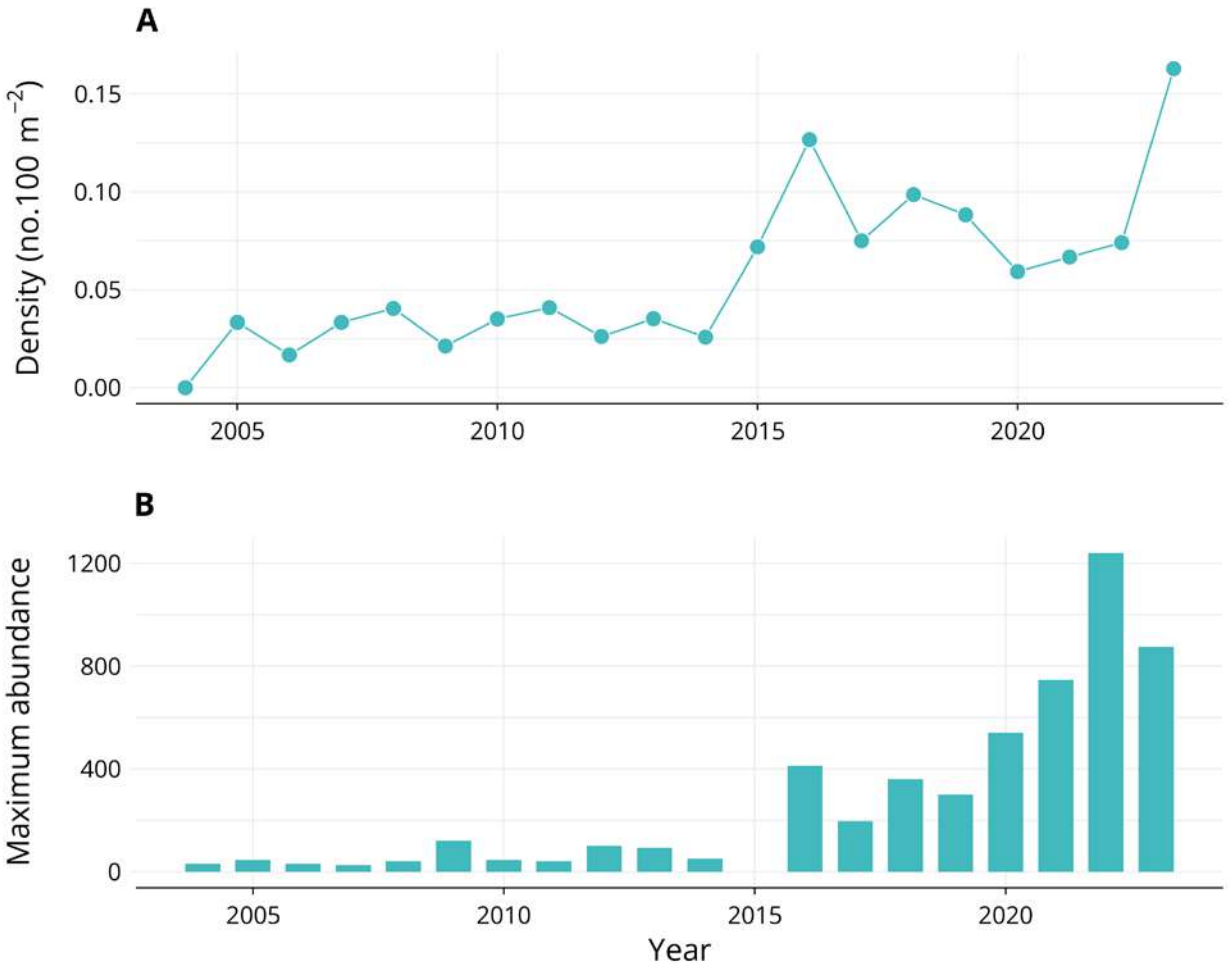
Road to recovery for the endangered Nassau grouper in the US Virgin Islands

Authors: Richard S. Nemeth, Elizabeth Kadison, Sarah Heidmann and Tyler B. Smith

Nassau grouper (*Epinephelus striatus*) were once the most common large grouper inhabiting coral reefs of the greater Caribbean region. Starting in the 1970's Nassau grouper populations collapsed due to intensive fishing on their spawning aggregations (Olsen et al., 1978, Aguilar-Perera, 2006), a practice that is still common throughout the world (Sadovy de Mitcheson et al., 2013). Fish spawning aggregation (FSA) sites are used by many species where 100's to 1000's of individuals gather to spawn at specific times and locations (Domeier and Colin, 1997). The hyperstability of FSA sites makes harvests seem sustainable until the spawning population collapses (Sadovy de Mitcheson and Domeier, 2005). Nassau grouper has shown few signs of population recovery throughout its range (Waterhouse et al., 2020) and only a few FSA sites remain (Kobara et al., 2013). Nassau groupers are considered critically endangered by IUCN (Sadovy de Mitcheson et al., 2018).

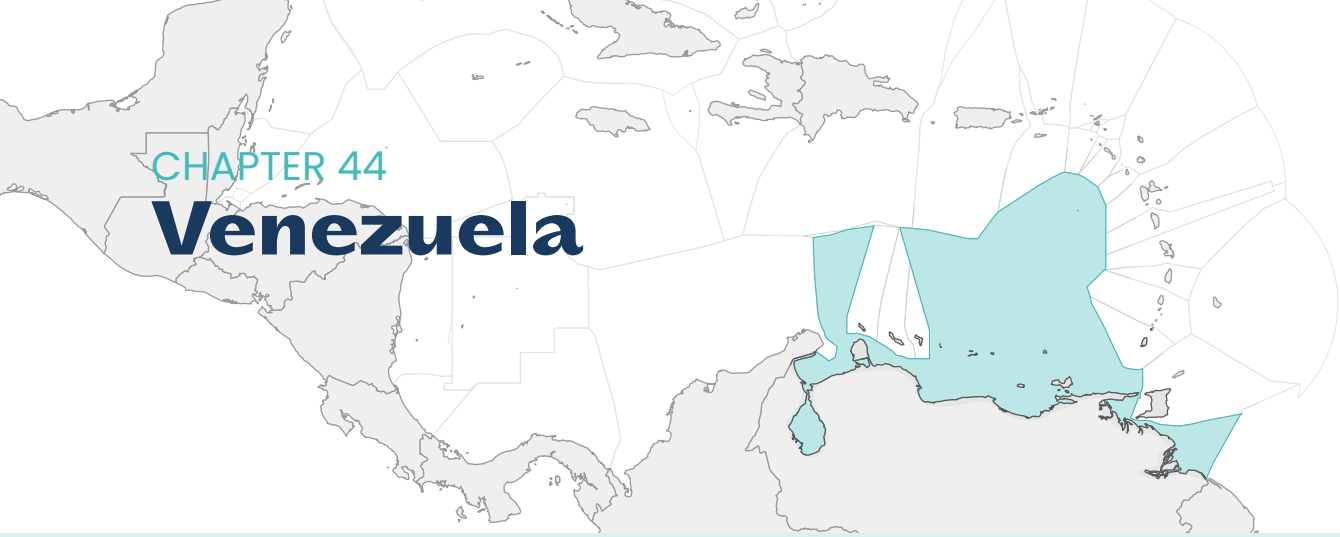
In the United States Virgin Islands (USVI) strategic conservation, management and monitoring efforts have resulted in signs of Nassau grouper population recovery. Regulations include a prohibition on harvest or possession in US federal waters (1996) and in USVI territorial waters (2006). Critical to their recovery was the discovery of a small Nassau grouper spawning aggregation in 2004, which facilitated the establishment in 2005 of the Grammanik Bank, a 3-month seasonal closed area (February to April) (**Figure 1**). The Grammanik Bank is only 1.5 km² in area and located on the edge of the Puerto Rican shelf 12 km south of St. Thomas, USVI (**Figure 2.43.4**). Yearly surveys on the Grammanik Bank FSA showed the abundance of Nassau grouper increased from 30 fish to over 1000 fish in about 20 years (**Figure 2**). The increase in abundance is largely attributed to fisher support of conservation and management actions. The recovery of the Nassau grouper spawning population is also evident in our long-term coral monitoring program data. Annual surveys around St. Thomas USVI over 20 years at 16 long-term coral reef monitoring sites have shown a concurrent increase in Nassau grouper (**Figure 2**).

↑ **Figure 1.** The photographs show the Nassau grouper spawning aggregation in January 2022 on the Grammanik Bank MPA. The lower photo shows male Nassau grouper in bicolor courtship coloration following a dark phase female with distended abdomen from hydrated ovaries.



↑ **Figure 2.** Trends in Nassau grouper density for St. Thomas (A) Territorial Coral Reef Monitoring Program transects (N = 16 sites) and maximum abundance of Nassau grouper at the Grammanik Bank fish spawning aggregation (B) site from 2004 to 2023.

Despite these population increases, a seven-year acoustic telemetry study identified a mismatch between the boundaries and timing of the seasonal closed area and showed that the Grammanik Bank seasonal closed area protected Nassau grouper only 65% of the time during the spawning season (Nemeth et al., 2023). Because the Nassau grouper spawning population is still highly vulnerable to bycatch mortality, the study recommended that the seasonal closed area be increased from 1.5 to 6.5 km² and extended from 3 to 5 months (January to May). These adaptive management changes would improve conservation goals by reducing bycatch and disturbance of spawning fish due to fishing gear.



CHAPTER 44

Venezuela

Authors: Aldo Croquer, Estrella Villamizar, Anaurora Yranzo-Duque, Jérémy Wicquart.

Data contributors: Esteban Agudo-Adriani, Alfredo Ascanio, Aldo Croquer, Philip Kramer*, Patricia R. Kramer*, Gloria Marino, Emy Miyazawa, Luis Miguel Montilla, Maria Antonieta Panama Lopez, Mariano Palazzo*, Andreina Rivera, Estrella Villamizar*, Alejandra Verde, Anaurora Yranzo-Duque, Jeannette Pérez.

*AGRRA data contributor(s).

INTRODUCTION

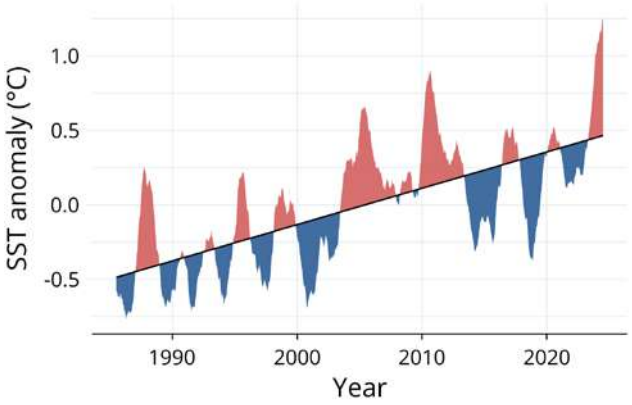
Venezuela, located on the northern coast of South America, faces the Caribbean Sea to the north. Coral reefs and coral communities occur along latitudinal and longitudinal environmental gradients determined by the presence of upwelling spots and contrasting oceanographic and coastal settings (Miloslavich et al. 2003, Chollet et al. 2012, Miyazawa et al. 2020). All coral reef and marginal coral formations in Venezuela occur within the southern ecoregion (Chollett et al. 2012) accounting for about 696 km², which represents 2.8% of the Caribbean's total coral reef extent.

The country includes a wide array of highly-diverse bottom types, coral reef habitats including extensive oceanic reefs, banks and archipelagos semi enclosed by large reef barriers such as the Los Roques, La Blanquilla, Aves Archipelago and Isla de Aves (Weil 2003, Schweizer et al. 2005, Yranzo et al. 2014, Villamizar et al. 2014, Debrot et al. 2019). Fringing and barrier reefs are also found along Morrocoy National Park, Cuare Wildlife Refuge and Cienaga de Ocumare (Villamizar et al. 2003, Croquer et al., 2010). Coral reefs in Venezuela have been shown to be structurally complex, a feature that supports highly diverse fish assemblages (Agudo-Adriani et al. 2019). However, in the western coast of the country and the western Venezuelan Gulf, reefs are less diverse and developed (Barrios-Garrido et al. 2016).

Scattered coral marginal communities established on rocky substrates occur along the central and eastern coast of Venezuela (Ortiz 2002). To the east, reef formation is limited by upwelling conditions (Chollett and Mumby 2013), both along the Cariaco Gulf coast (*e.g.* Mochima National Park) and offshore in Margarita and Los Testigos Islands (**Figure 2.44.4**, Weil 2003).

THREATS

Thermal regime



↑ **Figure 2.44.1.** Average Sea Surface Temperature (SST) anomaly from July 1985 to July 2024 over coral reefs of Venezuela. The black line is the long-term trend in SST anomaly, values below this line are negative SST anomalies (*i.e.* cooler than long-term trend), and values above this line are positive SST anomalies (*i.e.* warmer than long-term trend).

The long-term average in SST from 1985 to 2024 over coral reefs of Venezuela was 27.22 °C. Over this period, SST increased by 1.10 °C, representing a warming rate of 0.028 °C per year (**Figure 2.44.1**).

Coral reefs in Venezuela have been repeatedly affected by major bleaching events, notably in 1998, 2005, 2010 (Wilkinson and Souter, 2008, Weil

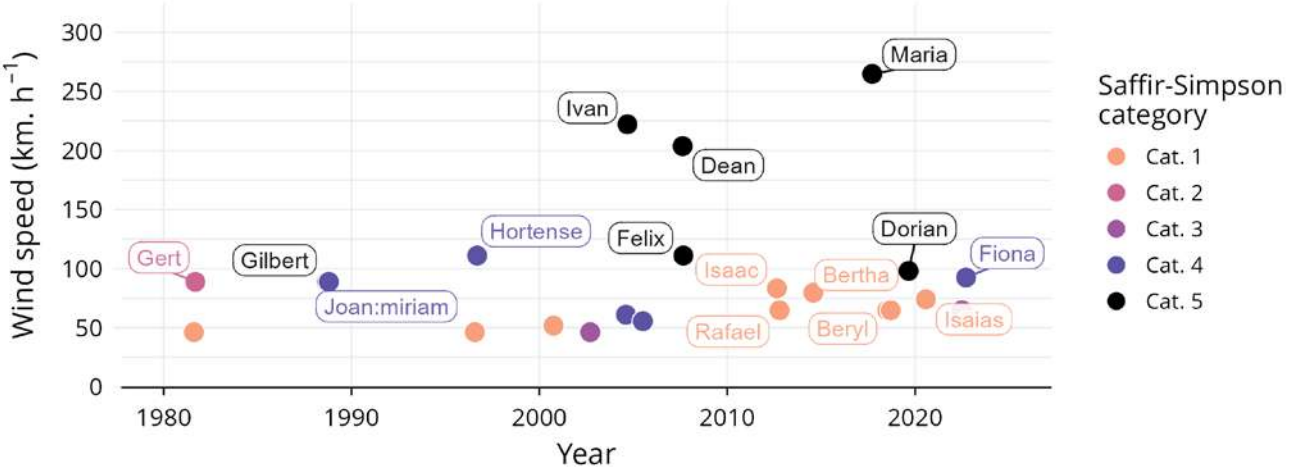
2003, Villamizar et al. 2008, Rodriguez et al. 2010, Bastidas et al. 2012), and more recently during the 2023-2024 marine heatwave. Notably, these massive bleaching events had differential impacts on Venezuelan reefs. In 1998, reefs of Morrocoy National Park were severely affected, in 2005, both oceanic and offshore reefs were affected (Villamizar et al. 2008) but mortality did not reach the levels observed in the eastern and northern Caribbean. In 2010, oceanic reefs of Los Roques suffered unprecedented impacts from extensive, intensive and prolonged thermal stress which reduced live coral cover by 40-60% (Bastidas et al. 2012, Rodriguez et al. 2010, Cróquer et al. 2010).

In 2023-2024, a new massive bleaching event was recorded across the Caribbean. For 12 months, a marine heatwave led to unprecedented thermal stress levels, triggering widespread bleaching and mortality in the Caribbean (Hoegh-Guldberg et al., 2023). AGRRA surveys done in November 2023, October and November 2024 in a reef at the central coast of Venezuela (Aragua state), portrays the impact of this heatwave in the country. The proportion of species and colonies affected by paleness and bleaching was notably higher in 2024 compared to 2023. In 2023, 24.83% of the coral colonies were pale, but only 1.38% of *Meandrina meandrites* and *Siderastrea siderea* were severely bleached. In 2024, up to 9 coral species were bleached; 24.86% of colonies in October and 21.55% in November being *Agaricia agaricites, Montastraea cavernosa, Meandrina meandrites, Siderastrea sidereal, Diploria labyrinthiformis* and *Orbicella faveolata* the most susceptible species.

The presence of upwelling spots along the Venezuelan coast might protect reefs from these heatwaves (Chollet and Mumby 2013), however, during the past few years, there is evidence suggesting that this process has weakened (Taylor et al. 2012) which suggest that upwelling alone might fail to protect reefs from heatwaves (Chollett et al. 2010). However, upwelling areas in the Caribbean have conservation utility because rates of thermal warming are lower (Chollett and Mumby 2013). In Venezuela, like in other Caribbean countries, massive bleaching events are followed by severe coral disease epizootic events that lead to coral mortality (Cróquer et al. 2010, Bastidas et at. 2012).

Hurricanes

Venezuela is relatively safe from the devastation of category 5 hurricanes as the country is normally away from the hurricane track (Schweizer et al. 2005), with the exception of Aves Island, in the central-eastern Caribbean. However, between 1980 and 2024, a total of 37 tropical storms passed within 100 km distance from a coral reef of Venezuela, and of these 5 were characterized by sustained wind speed greater than 100 km.h^{-1} (**Figure 2.44.2**). The hurricane with the highest sustained wind speed over the studied period was the hurricane Maria, in 2017, which passed 87 km from a coral reef with sustained wind speed of 265 km.h^{-1}.



↑ **Figure 2.44.2.** Maximum sustained wind speed of tropical storms passing within 100 km of a coral reef between 1980 and 2024 in Venezuela. Colors correspond to the cyclone's Saffir-Simpson category along its entire track. However, the values of sustained wind speed are extracted from the nearest tropical storm position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir-Simpson scale (*i.e.* 119 km.h^{-1}). Note that cyclones passing more than 100 km away from coral reefs of Venezuela are not represented, although they can have had an impact.

Planktonic blooms

The worst mass mortality event ever recorded in Venezuela occurred during the first quarter of 1996 (Labboy Nieves et al., 2001). In a few weeks, average water temperatures dropped to 10°C lower than usual, reduced concentrations of dissolved oxygen, salinities nearing 5 psu at the bottom-water interface, and lower than normal surficial current speeds were recorded over 160 km² (Labboy-Nieves et al., 2001). These conditions may have triggered complex processes that included planktonic blooms and mucilaginous aggregates and finally induced widespread mortalities due to organic matter decomposition and anoxia (Labboy-Nieves et al., 2001). In less than 3 months, the largest coastal coral reef system of Venezuela collapsed with dramatic loss of coral cover (Villamizar, 2000). Three decades after this event, some reefs in Morrocoy have failed to recover (Mariño-Briceño et al., 2024).

Coral diseases

Coral diseases are a major threat for populations of reef-building corals in Venezuelan reefs. The first formal report of a coral disease affecting the reef building coral *Orbicella annularis* was published in 1983 (Ramos-Flores 1983). During the 80's and the late 90's white band disease, plague and other pathologies were reported affecting acroporids and other species in oceanic and coastal reef systems (Weil 2003). However, the first quantitative assessment of the impact of a coral disease outbreak in Venezuela was conducted in the early 2000s in Los Roques National Park, when a wide-spread white plague mortality event significantly reduced live coral cover from 35 to 29% at Madrizqui key (Cróquer et al. 2003a, 2005). Ever since, coral diseases have been shown to be widely distributed and highly prevalent from east (Rodriguez and Cróquer 2008) to west and offshore and coastal reef systems (Weil et al. 2002, Weil 2003, Cróquer et al. 2003b, García et al. 2003, Villamizar et al. 2014, Yranzo-Duque et al. 2025). Most prevalent diseases are yellow band disease, particularly in *Orbicella* spp reefs (*e.g.* Cróquer et al. 2003b, Yranzo-Duque et al. 2025), black band disease, in shallow habitats (Rodriguez and Cróquer 2008), white plague disease, particularly in offshore reefs and/or reefs with high coral diversity (Cróquer et al. 2003a, 2005, Yranzo-Duque et al. 2005), dark spot disease (García et al. 2003, Sweet et al. 2013), and ciliate infections (Cróquer et al. 2006 a.b, Verde et al. 2016) among others. Furthermore, these diseases have also been observed at Aves Island, the northernmost territory of the country (Villamizar and Yranzo-Duque, pers. com.). To date, no official report of SCTLD massive mortalities have been observed in Morrocoy National Park and Cuare Wildlife Refuge (Yranzo-Duque et al. 2025) and Los Roques National Park (Jeannette Perez, *pers. obs.*). However, one colony of *Dendrogyra cylindrus* with obvious macroscopic signs of SCTLD have been observed in Boca de Cote reef, in Los Roques National Park (J. Pérez, pers. obs.).

Oil spills

Oil spills constitute another critical threat, and their impact on the reef systems is not easy to estimate due to the latter´s structural and functional complexity (Villamizar 2021). During the past decades increasing number of oil spills in the country´s refineries represents a high-risk factor to the conservation of coral reefs and other marine-coastal ecosystems of Venezuela (Villamizar 2020, Yranzo-Duque et al. 2025). In addition, spills that occurred in other countries have also reached coastal areas of Venezuela. These oil spills have been recorded reaching areas with extensive reef formations where reefs are recovering from past disturbances (*e.g.* Morrocoy National Park) and/or in areas where reefs have been shown to be resilient to bleaching and diseases (*e.g.* Cuare Wildlife Refuge) (Cróquer et al. 2012, Yranzo-Duque et al. 2025). Two cases can illustrate it. Since 2020 oily hydrocarbon stains produced at the El Palito Refinery on the central-western coast (very close to these Marine Protected Areas), have been frequently observed on the water surface of reef areas in Morrocoy National Park and the Cuare Wildlife Refuge. On the other hand, in April 2017, the Venezuelan coast was affected by an oil spill that occurred in Trinidad. It reached the beaches of the Gulf of Paria, the northern coast of the Paria Peninsula, Nueva Esparta State, the islands of Los Testigos, Coche, La Tortuga, La Blanquilla, La Orchila, and the Los Roques Archipelago National Park. In the latter, the presence of hydrocarbons was determined in two forms: as microparticles and as fragments in the form of fistulas on the seabed at one of the sites (Francisquí Islet) (Villamizar, 2021).

Terrestrial runoff, sedimentation and chemical pollution

Sedimentation and heavy metal pollution from terrestrial runoff has been historically acknowledged as a serious threat for coral reefs in Venezuela, particularly in coastal reefs that occur nearby large rivers displaying high levels of human interventions (Bastidas et al. 1999 a,b, Garcia et al. 2011). Heavy metals such as mercury have been historically imprinted in coral skeletons in western Venezuelan reefs (Ramos et al. 2009). Sedimentation smooths and suffocates corals and they may also transport pathogens (Croquer et al. 2003c) and introduce organic/inorganic xenobiotics with proven toxicity for corals (Bastidas et al. 1999a, 2004).

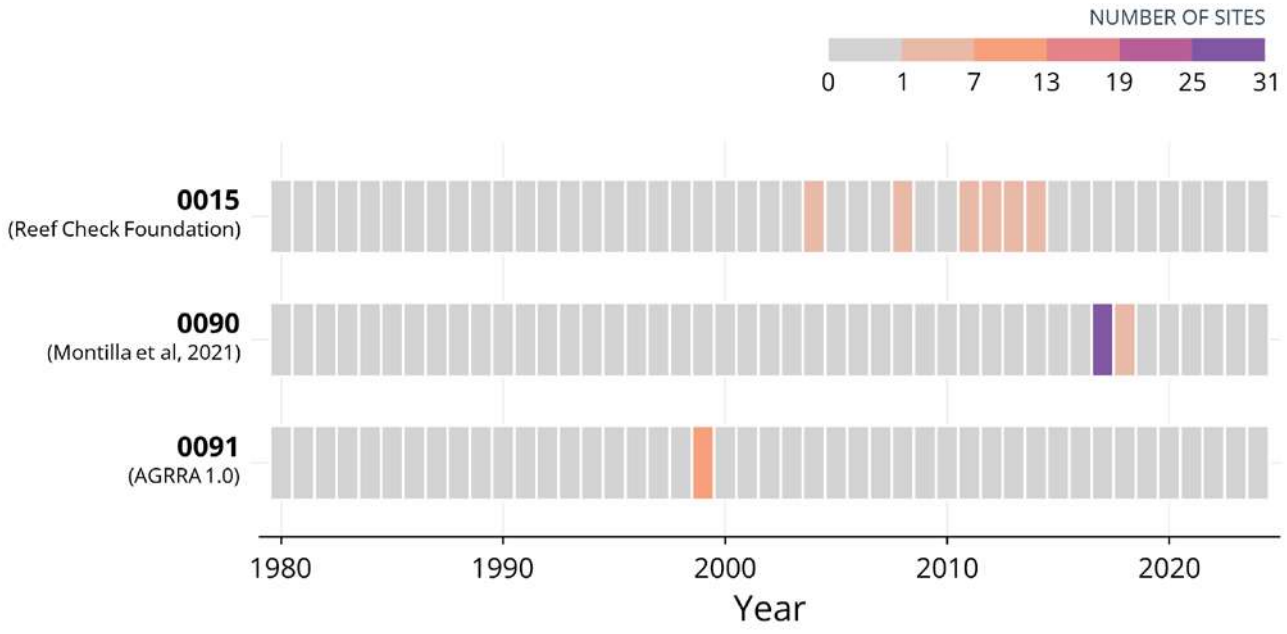
Invasive species

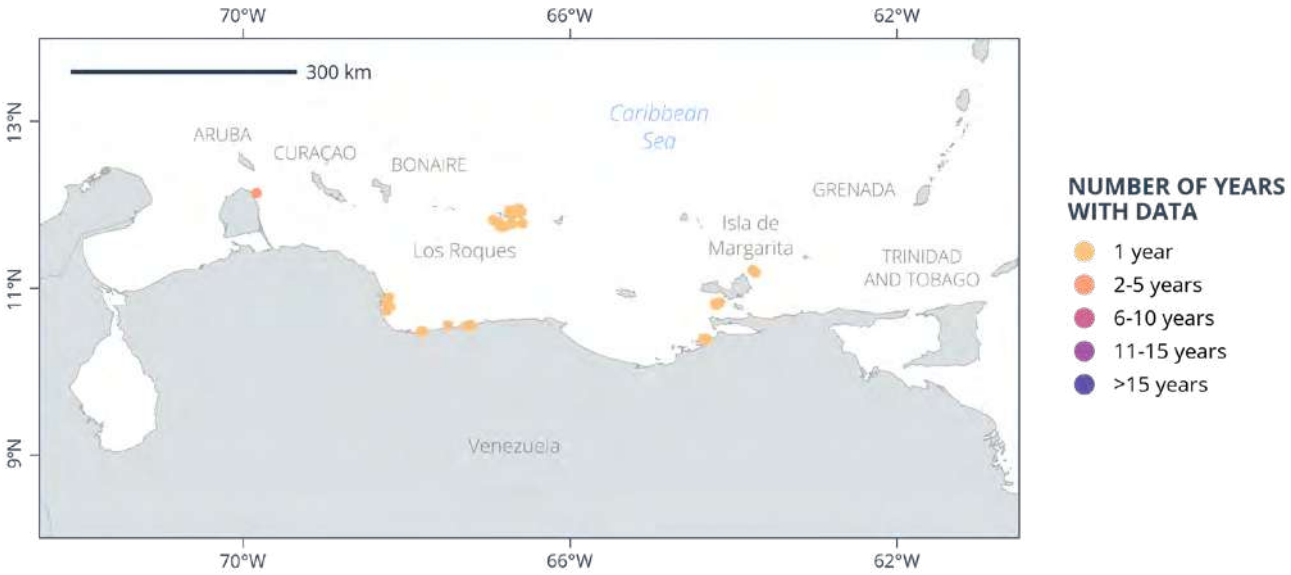
Perhaps the most relevant threat for Venezuelan reefs is the recent invasion of coral exotic species (Yranzo-Duque et al. 2025). The first exotic coral species, *Unomia stolonifera*, was observed in Mochima National Park in 2007, but first reported in 2014 (Ruiz-Allais et al. 2014). Likely introduced by aquarium trades, coral communities across the eastern coast of Venezuela have been undertaken by *U. stolonifera* (Ruiz-Allais et al. 2021). This Indopacific Xeniid has been shown to rapidly spread out to significantly reduce live coral cover of native coral species and associated biodiversity, while suffocating sponges, octocorals and other local benthic organisms (Ruiz-Allais et al. 2021). The species colonizes hard and soft bottom substrates, seagrass beds and artificial substrates across wide range depth intervals (0.5-20 m). The key of this successful invasion is two-fold: (1) effective dispersion mechanisms and (2) deterrence mechanism (*e.g.* allelopathy) to displace benthic competitors (Ruiz-Allais et al. 2021). In approximately 11 years, the species has invaded almost the entire eastern region of Venezuelan coast, increased its coverage in two bays of the Aragua State, central coast, and in the southwest of Cayo Sur, in Cuare Wildlife Refuge, in the central-western region of Venezuela. Recent surveys demonstrate that two other Alcyonacea of the Xeniidae family, and one scleractinian coral of the *Acropora tenuis* group (Villamizar et al., in review) have been found in the central western coast of Venezuela. However, the impacts of these invasions have not been demonstrated yet.

TRENDS IN BENTHIC COVER

Temporal trends in the percentage cover of benthic categories from 1980 to 2024 were estimated using data from 3 datasets (**Figure 2.44.3**) encompassing 50 monitoring sites. These data were collected through 57 surveys conducted between 1999 and 2018.

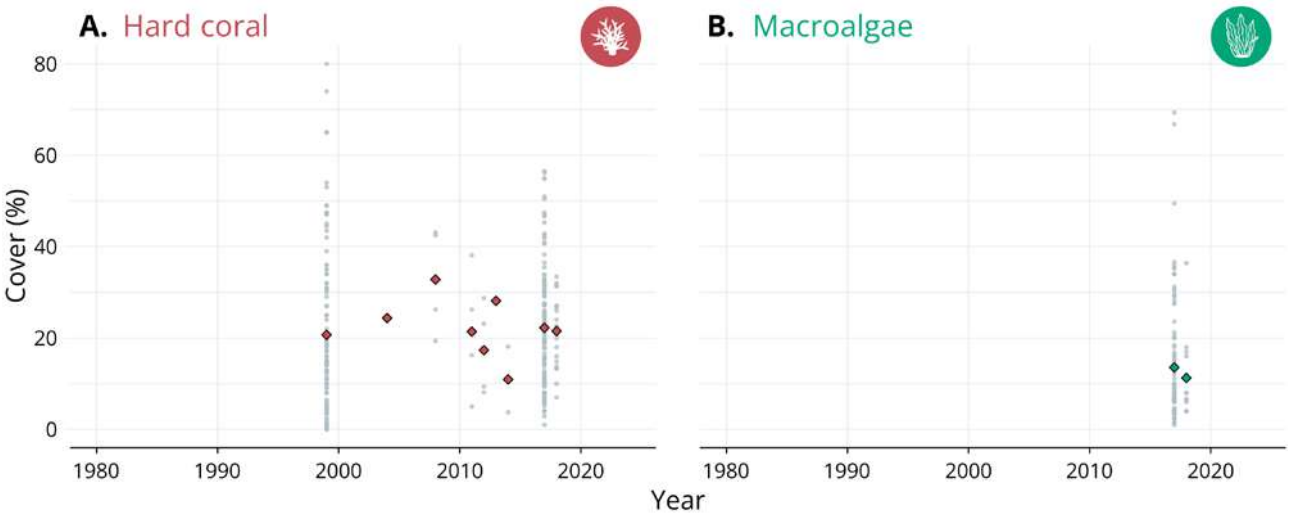
↓ **Figure 2.44.3.** Temporal distribution of surveys from datasets included in the analyses to model temporal trends of the three benthic categories. The y-axis numbers denote individual dataset identifiers, and the text below, shown in parentheses, specifies the corresponding dataset names as defined in gcrmndb_benthos (see Materials and Methods).





↑ **Figure 2.44.4.** Spatio-temporal distribution of benthic cover monitoring sites across Venezuela. Sites monitored for the greatest number of years are displayed above those monitored for fewer years. For the purposes of this report, a site is defined as a unique set of GPS coordinates. Consequently, small year-to-year variations in these coordinates may have resulted in lower "number of years with data" in this figure compared to those reported in other publications based on the same dataset. Aves Island, situated to the west of Dominica, is not represented on this map.

During the past decades, coral reefs in Venezuela have been shifting from coral-dominated to algal-dominated communities. In 35 years, live coral cover has dropped by half (38% to less than 20%) whereas algae (including macroalgae, turf algae and other algal functional groups) have duplicated (**Figure 2.44.5**). The benthic cover of other sessile organisms (*e.g.* hydrocorals, octorals, sponge) has remained stable within this time period. While coral reefs in Venezuela have been declining, the country still has reefs with higher coral cover compared to the regional context. Despite the impacts of massive bleaching andcoral disease epizootic events reported in the past and the demise of reefs in Morrocoy, there are still strongholds for iconic coral engineering species such as *Acropora palmata* (Zubillaga et al. 2008, Croquer et al. 2016), *A. cervicornis* (Martinez et al. 2021), *Dendrogyra cylindrus* (Cavada-Blanco et al. 2020), and *Orbicella faveolata* (Porto-Hannes et al., 2015, Yranzo-Duque et al. 2025)



↑ **Figure 2.44.5.** Modeled temporal trends of benthic percentage cover for hard coral (A), and macroalgae (B) in Venezuela from 1980 to 2024. The bold lines indicate the modeled mean cover, while the shaded ribbon represents the 95% confidence intervals. Grey points denote the annual means of observed data (*i.e.* monitoring data).

MATERIALS & METHODS

Bleached soft corals © Valentina Cucchiara

1. SPATIAL DATA

Several spatial datasets were used to perform the analyses and generate the maps presented in this report. The sources of these datasets, along with the processing steps and modifications applied, are detailed below.

1.1. Maritime areas

Maritime areas corresponding to countries and territories (hereafter referred to as "areas") were used as the main spatial units for the analyses and for the 44 chapters presented in **Part 2 - Syntheses for countries and territories**. These areas were primarily derived from the Exclusive Economic Zones (EEZ) shapefile provided by the World EEZ v12 dataset (Flanders Marine Institute, 2023). EEZs were selected as the reporting units for temporal trends in coral reefs, as they generally align with the jurisdictional boundaries where management and conservation measures are implemented. However, certain EEZs were modified or removed, and additional zones not corresponding to official legal entities were added when necessary.

For the United States, the Flower Garden Banks and Florida Keys areas were used instead of the entire U.S. continental EEZ. This choice allowed the separation of coral reef systems located hundreds of kilometers apart, ensuring that their temporal trends could be analyzed independently. Shapefiles for both areas were obtained from the Office of National Marine Sanctuaries (2021). A buffer of 0.3° (coordinate reference system: EPSG:4326) was applied to merge the multiple polygons of each area into a single feature. For the Florida area, buffer zones overlapping the EEZs of The Bahamas and Cuba were removed.

The Mexican EEZ was divided into two distinct units—"Mexico (Gulf of Mexico)" and "Mexico (Caribbean Sea)"—to reflect the separation between two marine ecoregions, as defined by Spalding et al. (2007).

Several islands and banks located in the central Caribbean Sea (*e.g.*, Serranilla Bank) host coral reefs that experience different environmental pressures compared to those near continental margins. To address this, coral reefs in the central Caribbean were grouped together using the delineation of the Seaflower Biosphere Reserve, kindly provided by INVEMAR. Portions of EEZs overlapping this reserve were excluded, resulting in reduced EEZ areas for several countries (*i.e.*, Colombia, Jamaica, Nicaragua, and Costa Rica).

Because Navassa Island is a disputed territory claimed by the United States, Haiti, and Jamaica, it was excluded from the chapters in **Part 2 – Syntheses for countries and territories**. Nevertheless, the coral reefs within this area were retained for the regional-scale analyses of the entire Caribbean.

1.2. Land boundaries

To represent inland boundaries on regional maps (*i.e.*, for the entire Caribbean region), we used the Admin 0 – Boundary Lines v5.1.0 dataset from Natural Earth (Natural Earth, 2024a). However, the spatial resolution of this dataset was insufficient for maps at the country or territory level. For those, we used the Global Administrative Areas v2.8 dataset (Hijmans, 2015), which provides finer boundary delineations.

1.3. Background maps

For the background representation of the wider Caribbean region, we used the high-resolution "Cross-Blended Hypsometric Tints" (Hypso + Shaded Relief + Ocean Bottom + Drainages) v6 dataset from Natural Earth (Natural Earth, 2024b).

2. GEOGRAPHIC INDICATORS

2.1. Coral reef extent

The coral reef extent of each area was estimated using the Tropical Coral Reefs of the World dataset (Institute for Marine Remote Sensing et al., 2011), developed by the World Resources Institute (WRI). This dataset provides a shapefile of coral reef locations at a 500 m resolution. Notably, it does not include reefs in Guatemala, despite their documented presence (Kjerfve et al., 2021). To address this gap, we incorporated coral reef distribution data for Guatemala provided by the Healthy Reefs for Healthy People initiative (Purkis, 2021), after reprojecting them to match the 500 m resolution of the WRI dataset.

We then performed a spatial intersection between this coral reef shapefile and the maritime areas previously defined (see section **Maritime areas**). For each area, we estimated the total coral reef extent using the EPSG:4326 coordinate reference system. We also calculated the extent of coral reefs relative to (1) the global total and (2) the GCRMN Caribbean region total, based on the GCRMN regions shapefile (Global Coral Reef Monitoring Network, 2024).

Differences between our estimates and those derived from other datasets primarily reflect variations in how coral reefs are defined. For example, Lyons et al. (2024), using the Allen Coral Atlas dataset, estimated the global extent of shallow coral reefs at 348,361 km², compared with our estimate of 249,636 km² from the WRI dataset. Discrepancies can be particularly pronounced at the national or territorial scale. In the Caribbean, for instance, estimates for The Bahamas range from 107,449 km² (Allen Coral Atlas; Lyons et al., 2024) to 4,081 km² (WRI dataset; this study), depending on whether sandy areas are included in the definition of coral reefs.

2.2. Human population

Indicators related to human population were derived from the Gridded Population of the World v4.11 – Population Count dataset (CIESIN, 2018), which provides estimates of human population counts for the years 2000, 2005, 2010, 2015, and 2020 at a spatial resolution of 30 arc-seconds (approximately 1 km at the equator).

A spatial intersection was performed between the Tropical Coral Reefs of the World dataset (Institute for Marine Remote Sensing et al., 2011) and the maritime areas defined for this report (see Section **Maritime areas**), to associate each coral reef polygon with its corresponding country or territory. Using the resulting spatial layer, we generated 20 km buffers around coral reefs within each area using Google Earth Engine (GEE). Although arbitrary, the 20 km distance was selected as a reasonable threshold representing the spatial extent over which human populations are likely to exert direct (*e.g.*, fishing, coastal development) or indirect (*e.g.*, sedimentation, nutrient runoff) pressures on coral reefs.

To ensure accuracy, the buffer polygons were corrected in R to remove overlapping zones between adjacent maritime areas. For example, the 20 km buffer created around the reefs of Belize slightly overlapped with the Mexico (Caribbean Sea) area due to coral reefs located near the maritime boundary between the two countries.

Using GEE, we then computed the spatial intersection between each corrected buffer and the Gridded Population of the World dataset. The total human population was summed within each buffer per maritime area and for each reference year (2000, 2005, 2010, 2015, and 2020), thereby estimating the number of people living within 20 km of a coral reef in each country or territory of the Caribbean region.

We also extracted the total human population of each country and territory by performing a spatial intersection between the EEZ land union dataset (Flanders Marine Institute, 2024) and the Gridded Population of the World dataset (CIESIN, 2018). Following this intersection, the total population within each EEZ was calculated by summing the population counts for each reference year (2000, 2005, 2010, 2015, and 2020).

Based on these extracted data, we computed three indicators of human population, for each area:

1. The number of inhabitants in 2020 living within 20 km of coral reefs.
2. The relative change in the number of inhabitants living within 20 km of coral reefs between 2000 and 2020 (**Eq. 3.1**).
3. The percentage of the total population of each area living within 20 km of coral reefs in 2020.

Eq. 3.1

$$Pop_{change} = \frac{\left(pop_{2020} - pop_{2000}\right)}{pop_{2000}} * 100$$

It should be noted that the values derived using this approach may differ from national population census data (see Andrew et al., 2019). However, we used the Gridded Population of the World dataset to ensure methodological consistency across all countries and territories in the region. In addition, the raster format of this dataset was essential for calculating total human population values based on spatial intersections with shapefile layers.

3. HURRICANE'S INDICATORS

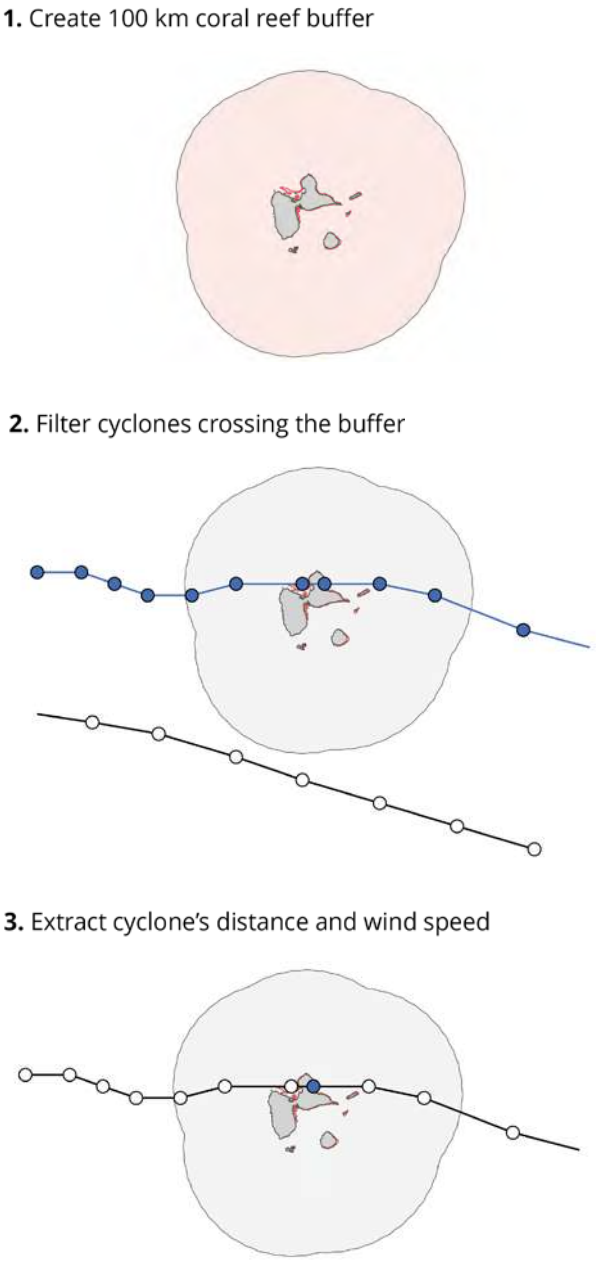
Tropical storms, and more particularly hurricanes (also named cyclones in the Pacific and the Indian Ocean or typhoons in South East Asia), are major disturbance events affecting coral reefs (Puotinen et al., 2020). To examine their occurrence on Caribbean coral reefs over the past four decades, we used version 4.01 of the International Best Track Archive for Climate Stewardship (IBTrACS) dataset (Knapp et al., 2010; Gahtan et al., 2024). This dataset provides 3-hourly storm positions at a spatial resolution of 0.1° (approximately 10 km).

We applied four successive filters to the IBTrACS dataset. First, we retained only hurricanes, defined as events with maximum sustained wind speeds exceeding 119 $km.h^{-1}$ (*i.e.*, Category 1 or higher on the Saffir–Simpson scale; see **Table 3.1**), across their entire tracks. It should be noted, however, that wind speeds may fall below this threshold at certain points along a hurricane's trajectory. Second, we restricted the dataset to hurricanes occurring after 1 January 1980, corresponding to the reporting period of interest. Third, we excluded hurricanes lacking sustained wind speed records for all positions along their tracks. Finally, using the coral reef distribution shapefile from the Tropical Coral Reefs of the World dataset (Institute for Marine Remote Sensing et al., 2011), we generated a 100 km buffer around Caribbean reefs within Google Earth Engine (GEE). Only hurricanes intersecting this buffer were retained, while acknowledging that hurricane impacts on coral reefs may extend beyond this distance (Beeden et al., 2015; Puotinen et al., 2020).

↓ **Table 3.1.** The different categories of the Saffir-Simpson scale and their associated sustained wind speed. Recently, Wehner and Kossin (2024) proposed the addition of a sixth category to the Saffir-Simpson scale.

Type	Category	Sustained wind speed ($km.h^{-1}$)
Tropical storm		64 to 118
Hurricane	1	119 to 153
	2	154 to 177
	3	178 to 210
	4	211 to 251
	5	> 251

Following these filtering steps, each hurricane was assigned a Saffir–Simpson scale category (**Table 3.1**) based on the maximum sustained wind speed recorded along its track. For each hurricane, we then extracted (1) the minimum distance between its trajectory and coral reefs, and (2) the sustained wind speed at the point on the track closest to a coral reef. These extractions were conducted separately for each area (see section **Maritime areas**). The main steps of the hurricane data extraction process are summarized in **Figure 3.1**.



↑ **Figure 3.1.** Schematic representation of the main steps in the hurricane data extraction process, illustrated using Guadeloupe as an example. Coral reefs are shown in red, while points along hurricane tracks represent 3-hourly storm positions. The final step is repeated for each hurricane intersecting the coral reef buffer.

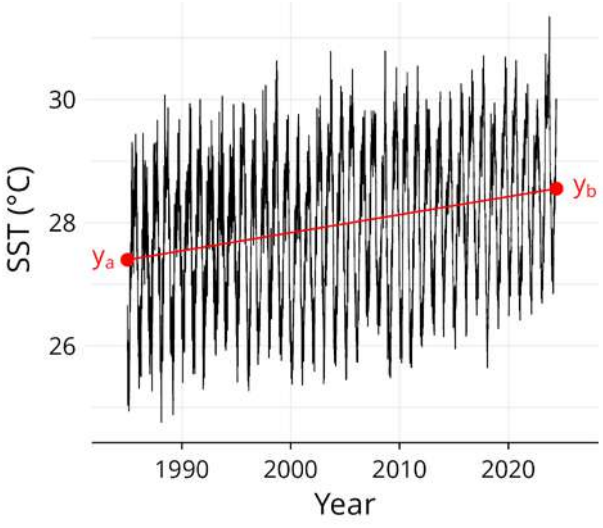
4. THERMAL REGIME INDICATORS

To characterize the thermal regime of coral reefs in the Caribbean, we used the CoralTemp v3.1 dataset (Skirving et al., 2020), developed by the NOAA Coral Reef Watch program. CoralTemp provides global daily Sea Surface Temperature (SST) data at a spatial resolution of 5 km, available from January 1, 1985, to the present. The dataset is distributed as one NetCDF4 file per day, which we downloaded and processed to derive a series of thermal regime metrics and to generate graphical representations of SST values, long-term SST trends, and SST anomalies.

4.1. Sea Surface Temperature

We extracted daily SST values over coral reef areas using the coral reef distribution shapefile from the *Tropical Coral Reefs of the World* dataset (Institute for Marine Remote Sensing et al., 2011). The extracted SST values were then averaged per day and per maritime area (see section **Maritime areas**). The same procedure was applied to compute a regional daily average for the entire Caribbean. The resulting time series were used to represent the daily average SST over coral reefs from January 1, 1985, to December 31, 2024 (see **Supplementary Figures 1**). To better visualize intra- and inter-annual SST variability, we also expressed the extracted time series using Julian days, with months displayed on the x-axis instead of years (see **Supplementary Figures 2**).

4.2. Long-term SST trends



↑ **Figure 3.2.** Illustration of the linear regression performed on the SST time-series, and the points (y_a = first point of the long-term trend, y_b = last point of the long-term trend) used to estimate the average change in SST and the average annual warming rate. The data used in this example correspond to coral reefs in Belize.

We used the previously extracted daily SST values, averaged for each area (see section **Maritime areas**) and for the entire Caribbean region, to assess long-term SST trends from January 1, 1985, to December 31, 2024. To do so, the dates in these time series were converted to a numeric format (*e.g.*, "1985-01-01" was replaced by 1), and ordinary linear regressions were performed to estimate the trends, as illustrated in **Figure 3.2**.

We then used the slope and intercept of the linear regressions to calculate the first (y_A) and last (y_B) SST values of the long-term trends (**Eqs. 3.2 and 3.3**). The average change in SST between 1985 and 2024 was obtained by subtracting the first SST value from the last (**Eq. 3.4**). Finally, the annual rate of SST change over the study period was estimated by dividing the average change in SST by 39 years, corresponding to the time interval from 1985 to 2024 (**Eq. 3.5**).

Eq. 3.2 $$y_A = slope * x_A + intercept$$

Eq. 3.3 $$y_B = slope * x_B + intercept$$

Eq. 3.4 $$SST\ change = y_B - y_A$$

Eq. 3.5 $$Annual\ rate\ of\ change = \frac{y_B - y_A}{2024 - 1985} = \frac{SST\ change}{39}$$

4.3. SST anomaly

In addition to SST, we also used SST anomalies to describe the occurrence, intensity, and duration of anomalous temperatures from 1986 to 2024. For this purpose, we used the previously extracted daily SST values and subtracted, for each area (see section **Maritime areas**) and for the entire Caribbean region, the long-term mean SST calculated over 1985–2024. We then applied a centered 365-day moving average to the daily SST anomaly time series for each area and for the entire region to smooth the data and enhance visualization.

These time series were first plotted using a zero anomaly value (0 °C) as the reference line (see **Supplementary Figures 3**). In addition, we also represented them relative to the long-term trend in SST anomaly (*e.g.*, **Figure 2.1.1**) to account for gradual warming of coral reef environments under climate change. This latter visualization approach allows clearer identification of periods

of temperature anomalies corrected for the long-term increase in SST, following a shifting baseline approach (see Amaya et al., 2023).

We opted to use SST anomalies rather than Degree Heating Weeks (DHW) - a commonly used indicator of thermal stress in coral reef studies (*e.g.*, Eakin et al., 2010; Muñiz-Castillo et al., 2019) - for four main reasons:

- **Accessibility**. SST anomalies are expressed in a familiar unit (°C) and are easier to interpret for non-scientific audiences.
- **Generality**. Unlike DHW, which are specifically designed to assess bleaching risk in hard corals, SST anomalies provide a broader indication of thermal stress for coral reef organisms in general.
- **Trend correction**. SST anomalies allow correction for the long-term warming trend in SST.
- **Bidirectional anomalies**. SST anomalies can capture both anomalously high and low temperature events, whereas DHW focus exclusively on heat stress. Indeed, it has been shown that cold waves can also cause mortality phenomena on hard corals (Collela et al., 2012).

5. BENTHIC COVER INDICATORS

In addition to the geographic and climatic indicators described above, we incorporated biological indicators related to benthic cover.

Although numerous ecological variables are measured and monitored across the world's coral reefs (Flower et al., 2017), benthic cover is arguably the most widely assessed, both spatially and temporally. This is largely because most benthic cover monitoring methods are relatively simple, cost-effective, and require limited taxonomic expertise (Aronson et al., 1994; Hill and Wilkinson, 2004). Measurements of benthic cover began prior to 1980 (Connell et al., 1997; Dustan and Halas, 1987), and advances in survey techniques have had minimal effects on comparability with historical data. For instance, different approaches such as point-intercept transects and photo-quadrats generally yield consistent estimates of benthic cover (Jokiel et al., 2015).

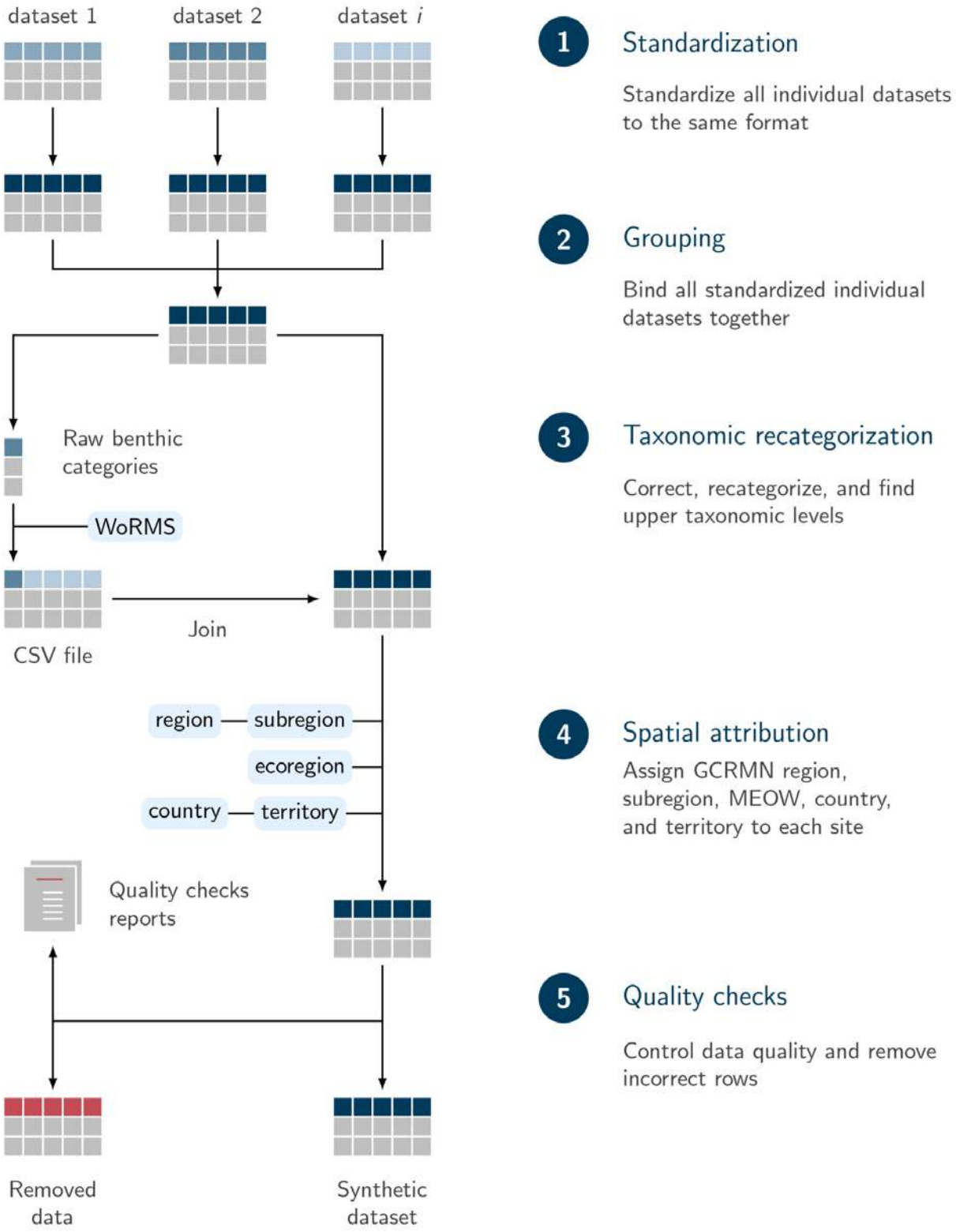
The comparability of methods enables the integration of datasets from multiple monitoring programs, allowing temporal trends in benthic cover to be estimated at large spatial scales.

This approach, known as full data analysis, provides greater precision than other forms of synthesis such as meta-analyses or systematic reviews (Spake et al., 2020). Unlike meta-analysis, which typically produces a single effect size, full data analysis facilitates the estimation of complex temporal dynamics. Furthermore, because it relies directly on raw data, it incorporates a larger quantity of information than meta-analyses or systematic reviews, which are constrained to published results.

Among the wide range of benthic categories used by monitoring programs, we chose to report five major benthic categories (hard coral, macroalgae, turf algae, coralline algae, and other fauna), and three hard coral genera (*Acropora*, *Orbicella*, *Porites*). Unlike the "*Status of Coral Reefs of the World: 2020*" GCRMN report (Souter et al., 2021), we separated algae into three functional groups - macroalgae, turf algae, and coralline algae (see section **Taxonomic re-categorization**, for definitions). This decision was made to better capture the ecological complexity of coral reefs and to reflect the distinct functional roles of different algal groups (Tebbett et al., 2023a). For hard corals, *Orbicella* and *Porites* were selected because they represented the greatest proportion of total hard coral cover across surveys conducted between 1980 and 2024. Although less abundant in terms of average benthic cover, *Acropora* was included due to its critical functional role in Caribbean coral reefs (McWilliam et al., 2018).

5.1. Data integration

A full data analysis of benthic cover in coral reefs is only feasible when a homogeneous dataset is available. As no such dataset existed for the GCRMN Caribbean region, we first conducted a data integration step prior to analysis. The objective was to construct a consistent synthetic dataset of coral reef benthic cover by aggregating multiple heterogeneous individual datasets. This data integration relied on an improved version of the workflow developed by Wicquart et al. (2022) for the "*Status of Coral Reefs of the World: 2020*" GCRMN report (Souter et al., 2021). Key improvements to the workflow included the addition of quality checks, which provide stronger guarantees of data reliability. The data integration workflow applied in this report consists of five main steps, summarized in **Figure 3.3** and described in detail in the following sections.



↑ **Figure 3.3.** Schematic representation of the five main steps of the data integration workflow used to construct the synthetic dataset on coral reef benthic cover. MEOW = Marine Ecoregions of the World (Spalding et al., 2007).

5.1.1. Data collation

We recontacted data contributors from the Caribbean region who had participated in the "*Status of Coral Reefs of the World: 2020*" GCRMN report (Souter et al., 2021) to obtain authorization for data reuse and to update the datasets they had previously shared.

In parallel, we issued an open call for data contributions in July 2024, disseminated through the GCRMN-Caribbean and SPAW-RAC networks. Data collation efforts were further strengthened by a dedicated workshop on data collection and analysis for the present report, held in Puerto Morelos from 1-4 October 2024. The workshop, hosted by the *Universidad Nacional Autónoma de México* (UNAM) and the *Instituto de Ciencias del Mar y de Limnología*, facilitated exchanges among diverse formal and informal networks across the region, substantially enhancing data mobilization.

Finally, data collation was complemented by searching for datasets available in scientific articles, data papers, and coral reef data platforms (*e.g.*, MERMAID), and by inviting dataset owners to contribute. Particular efforts were devoted to collating benthic cover data from the 1980s through extraction from published studies, as raw datasets could not be obtained directly from the researchers.

5.1.2. Data standardization

The datasets compiled during the data collation step were heterogeneous in terms of format (*e.g.*, CSV, Excel), structure, variable names, and measurement units (*e.g.*, meters vs. feet for depth). To enable integration into a single homogeneous synthetic dataset, a data standardization step was required. For each dataset, we developed an R script to apply dataset-specific modifications according to a predefined structure and set of variables (**Table 3.2**). These modifications included merging Excel spreadsheets, linking primary datasets with associated metadata (*e.g.*, site coordinates, benthic codes), selecting and renaming variables, converting units, harmonizing Coordinates Reference Systems (CRS) and date formats, and deriving benthic cover values from raw observations such as point-intercept transects or photo-quadrats (**Figure 3.3**, *Step 1*).

All data standardization was performed using R scripts, ensuring transparency of modifications and allowing for subsequent corrections if errors are identified. Because all R scripts are publicly available, dataset owners can also verify the changes applied to their data (see https://github.com/GCRMN/gcrmndb_benthos).

Following standardization, each dataset was exported in CSV format, and all CSV files were then merged into a single raw synthetic dataset (**Figure 3.3**, *Step 2*).

↓ **Table 3.2.** Description of variables included in the gcrmndb_benthos synthetic dataset. Variable names (except region, subregion, ecoregion, category, subcategory, and condition) follow DarwinCore terms. Variables 2-13 correspond to spatial dimensions, variables 14-17 to temporal dimensions, and variables 20-28 to taxonomic information.

Nb	Variable	Type	Description
1	datasetID	Factor	ID of the dataset
2	region	Factor	GCRMN region
3	subregion	Factor	GCRMN subregion
4	ecoregion	Factor	Marine Ecoregion of the World (Spalding et al., 2007)
5	country	Factor	Country
6	territory	Character	Territory
7	locality	Character	Site name
8	habitat	Factor	Habitat
9	parentEventID	Integer	Transect ID
10	eventID	Integer	Quadrat ID
11	decimalLatitude	Numeric	Latitude (decimal, EPSG:4326)
12	decimalLongitude	Numeric	Longitude (decimal, EPSG:4326)
13	verbatimDepth	Numeric	Depth (m)
14	year	Integer	Four-digit year
15	month	Integer	Integer month
16	day	Integer	Integer day
17	eventDate	Date	Date (YYYY-MM-DD, ISO 8601)
18	samplingProtocol	Character	Method used to acquire the measurement
19	recordedBy	Character	Person who acquired the measurement
20	category	Factor	Benthic category
21	subcategory	Factor	Benthic subcategory
22	condition	Character	Condition for hard corals
23	phylum	Character	Phylum
24	class	Character	Class
25	order	Character	Order
26	family	Character	Family
27	genus	Character	Genus
28	scientificName	Character	Species
29	measurementValue	Numeric	Percentage cover

5.1.3. Taxonomic re-categorization

During the data standardization step, the raw benthic categories - those originally defined by the data providers in each dataset - were stored in the temporary variable *organismID*. We then used the *taxize* R package to automatically retrieve taxonomic information for variables 23 to 28, from the World Register of Marine Species (WoRMS) database. Subsequently, a CSV file containing all unique raw benthic categories was exported. Each automatically completed entry was verified, and any missing taxonomic information was manually added for variables 20 to 28 (**Figure 3.3**, *Step 3*).

When a raw benthic category represented a broad, non-taxonomic grouping, we completed the variables category and, where possible, subcategory accordingly. For example, the raw category "turf" was assigned to category = "Algae" and subcategory = "Turf algae."

Several specific cases required particular attention during taxonomic reclassification:

- **Mixed categories**. In this case, we assigned the lowest common denominator of the mixed categories. For example, when the raw benthic category was "Macroalgae and turf algae," we classified it as "Algae".
- **Stacked categories**. In this case, we retained the category that was above the other. For instance, when the raw benthic category was "Algae on dead coral," we assigned the category to "Algae."
- **Homonym taxa names**. In this case, we either contacted the data provider to replace the ambiguous raw category with a precise one during the data standardization step, or we left the variable *category* blank, which subsequently led to the removal of the corresponding rows. For example, "Turbinaria" refers both to a genus of Scleractinia (corals) and to a genus of Fucales (macroalgae). If the data provider was able to clarify the intended meaning, the raw category was replaced with a specific denomination (e.g., "Turbinaria – Coral") during the standardization step.
- **Unrequired or unintelligible categories**. In this case, we did not complete the variable category, which also led to the removal of the corresponding rows. Examples include raw benthic categories such as "Shadow," "Other," or "Unknown."

When the raw benthic category referred to a taxon (*e.g.*, "Acropora") rather than a broad grouping (*e.g.*, "Hard coral"), we completed variables 23 to 28. To do so, we first verified the spelling of each raw benthic category using WoRMS and considered updates in taxonomic classification that may have occurred between the time of data entry by the providers and the integration process. Within the CSV file, we then filled in the taxonomic rank corresponding to the raw benthic category, as well as all higher taxonomic levels (variables 23 to 28 in **Table 3.2**). For example, for the raw benthic category "coral acropora," we entered "Acropora" for the variable genus, "Acroporidae" for family, and continued this process up to the variable phylum. When the raw benthic category was taxonomic, the variables category and subcategory were left empty. These variables were later completed in R by applying attribution rules derived from the taxonomic variables (23 to 28 in **Table 3.2**). The rules were as follows:

- **Hard coral**. We assigned *category* = "Hard coral" when the variable *order* was "Scleractinia" or when the variable *family* was "Milleporidae" or "Helioporidae." When contributors used the term "hard coral" without providing taxonomic information, we also assigned *category* = "Hard coral." Note that the "Hard coral" category excluded dead corals but may have included partially or fully bleached corals.
- **Macroalgae**. We assigned *category* = "Algae" and *subcategory* = "Macroalgae" when the variable *class* was "Phaeophyceae," "Florideophyceae," or "Ulvophyceae" and the variable *order* was not "Corallinales." When contributors used the term "macroalgae" without providing taxonomic information, we applied the same classification (*category* = "Algae" and *subcategory* = "Macroalgae").
- **Coralline algae**. We assigned *category* = "Algae" and *subcategory* = "Coralline algae" when the variable *order* was "Corallinales." When contributors used the term "coralline algae" without taxonomic detail, we again classified it as *category* = "Algae" and *subcategory* = "Coralline algae".
- **Turf algae**. When contributors used the term "turf algae," we assigned *category* = "Algae" and *subcategory* = "Turf algae".
- **Cyanobacteria**. We assigned *category* = "Algae" and *subcategory* = "Cyanobacteria" when the variable *phylum* was "Cyanobacteria".

- **Other fauna**. We assigned *category* = "Other fauna" when the variable *phylum* was "Porifera," "Chordata," "Echinodermata," "Bryozoa," "Annelida," "Mollusca," or "Arthropoda"; when the variable *order* was "Actiniaria," "Alcyonacea," "Zoantharia," "Corallimorpharia," or "Antipatharia"; or when the variable *subclass* was "Octocorallia".
- **Seagrass**. We assigned *category* = "Seagrass" when the variable *phylum* was "Tracheophyta".

Once the recategorization was completed, we merged the table containing the *organismID* variable and the newly created variables (20 to 28 in **Table 3.2**) with the main data table. We then removed the temporary variable *organismID* and deleted all rows with missing values in the variable *category*. Removing missing values in *category* did not affect percentage cover values, although it reduced the total percentage cover for a sampling unit (*e.g.*, quadrat, transect) to below 100%. Finally, we aggregated the variable *measurementValue* by grouping across all other variables, thereby preventing duplicate benthic categories within a given sampling unit.

The full recategorization of raw benthic categories into standardized categories (variables 20 to 28, **Table 3.2**) is publicly available in the file **"***03_taxo-recategorisation.csv***"** on the gcrmndb_benthos GitHub repository.

5.1.4. Spatial attribution

Using site coordinates (from the variables *decimalLatitude* and *decimalLongitude*), we added five spatially related variables to the dataset (**Figure 3.3**, *Step 4*). First, we performed a spatial intersection to assign the GCRMN "region" and "subregion" variables to each site, using the GCRMN regions shapefile (Global Coral Reef Monitoring Network, 2024). We then added the variable *ecoregion* by intersecting sites with the Marine Ecoregions of the World dataset (Spalding et al., 2007). Finally, we included the variables *country* and *territory* by referencing the fields *SOVEREIGN1* and *TERRITORY1* from the World EEZ v12 dataset (Flanders Marine Institute, 2023). Because of the limited spatial resolution of the World EEZ v12 dataset, some sites located near coastlines fell slightly outside the EEZ polygons. To minimize missing values for *country* and *territory*, we applied a two-step procedure: we first intersected sites with the World EEZ v12 dataset, then reprocessed sites with missing values by performing a second intersection with EEZ polygons buffered by 1 km.

5.1.5. Quality checks

Pooling large volumes of data without adequate verification risks compromising data quality. Previous studies have shown that insufficient quality control in large synthetic datasets can lead to biased interpretations of ecological questions (Augustine et al., 2024). To address this issue, we implemented a dedicated step for data quality assessment (**Figure 3.3**, Step 5).

↓ **Table 3.3**. List of quality checks used for the gcrmndb_benthos synthetic dataset.

Nb	Variable	Question
1	decimalLatitude decimalLongitude	Are the latitude and longitude available?
2	decimalLatitude	Is the latitude within its possible boundaries (*i.e.* between -90 and 90)?
3	decimalLongitude	Is the longitude within its possible boundaries (*i.e.* between -180 and 180)?
4	decimalLatitude decimalLongitude	Is the site within the coral reef distribution area (100 km buffer)?
5	decimalLatitude decimalLongitude	Is the site located within a GCRMN region?
6	decimalLatitude decimalLongitude	Is the site located within an EEZ (1 km buffer)?
7	year	Is the year available?
8	measurementValue	Is the sum of the percentage cover of benthic categories within the sampling unit greater than 0 and lower than 100?
9	measurementValue	Is the percentage cover of a given benthic category (*i.e.* a row) greater than 0 and lower than 100?

We established a set of nine quality checks, adapted from Vandepitte et al. (2015) (**Table 3.3**). Each quality check was formulated as a question linked to one or more variables, making them readily translatable into code.

Because of rounding decimal places, the sum of percentage cover values for benthic categories within a sampling unit can slightly exceed 100% (Razak et al., 2024). To avoid unnecessary data loss, we normalized rows where the total percentage cover was between 100 and 101 to a maximum of 100. This adjustment was applied prior to implementing quality check 8 (**Table 3.3**).

We evaluated each quality check as either "True" or "False," and removed all rows that failed at least one check. However, correcting errors is preferable to deleting data, in order to preserve sufficient data for analysis. To support error correction, we generated summary reports for each dataset integrated into the synthetic dataset. These reports detailed the percentage of rows removed per quality check, allowing us to identify and correct errors by updating the data standardization code (**Figure 3.3**, *Step 1*). For example, a common and easily corrected error was the inversion of *decimalLatitude* and *decimalLongitude*.

In addition, the individual dataset reports provided descriptive information such as the proportion of missing values per variable, the percentage of rows by benthic category and taxonomic level, and the spatial and temporal distribution of observations.

5.1.6. Code availability

The data integration process, comprising data collation, standardization, and cleaning, was implemented through the gcrmndb_benthos GitHub repository. The full code used for the data integration of benthic cover monitoring data is publicly available at https://github.com/GCRMN/gcrmndb_benthos. This repository serves as a shared resource for GCRMN reports and was not created or maintained specifically for the present study. All data integration steps were carried out in R version 4.5.1 (R Core Team, 2025), primarily using the packages *tidyverse* v2.0.0 (Wickham et al., 2019), *mermaidr* v1.2.5 (Gelfand, 2025), *taxize* v0.10.0 (Chamberlain and Szöcs, 2013), and *sf* v1.0-21 (Pebesma et al., 2023). The complete list of R packages used and their associated versions is provided in the README file of the GitHub repository.

5.2. Data analysis

Over the past three decades, numerous studies have assessed temporal trends in the percentage cover of hard corals and other benthic categories at broad spatial scales. Most of these studies aggregated data from multiple monitoring programs and interpreted trends by plotting mean annual hard coral cover together with associated confidence intervals (*e.g.* Gardner et al., 2003; Obura et al., 2017). Other studies applied statistical models to investigate the factors driving temporal variation in percentage cover (Bruno and Selig, 2007; De'ath et al., 2012; Jackson et al., 2014; Tebbett et al., 2023b).

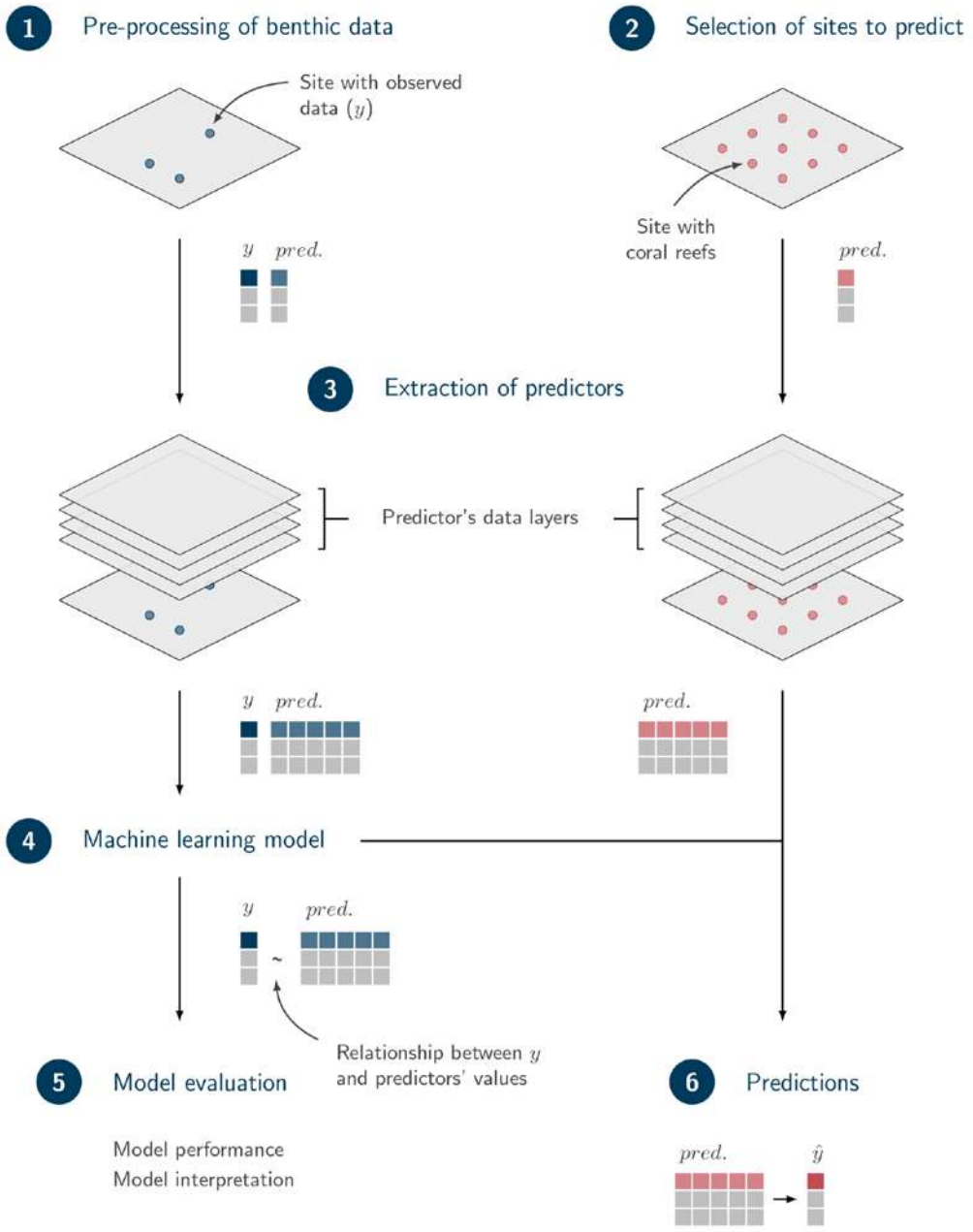
The main assumption underlying these approaches is that the aggregated data are representative of the coral reefs within the region of interest. However, coral reef monitoring is highly heterogeneous in both space and time (Souter et al., 2021). Some reefs, often located near research facilities, have been intensively monitored, whereas others, particularly those in remote areas, remain largely unmonitored. For instance, Cuba contains 20.3% of the total coral reef extent in the Caribbean, yet only 0.9% of all surveys conducted in the region between 1970 and 2024 were carried out there (**Tables 1.1.1** and **1.3.1**). Conversely, Florida hosts just 4.8% of the total coral reef extent in the Caribbean but accounted for 14.5% of all surveys conducted in the region during the same period (**Tables 1.1.1** and **1.3.1**). As a result, calculating mean annual cover at the regional scale risks producing benthic cover trends that are not representative of the region's coral reefs as a whole, but instead reflect changes occurring at monitoring sites.

One way to address this bias is to use predictors to estimate benthic cover temporal trends in areas with limited or no observed data. Predictors - also referred to as covariates or explanatory variables - are variables (*e.g.*, sea surface temperature) that can be incorporated into statistical models to estimate a response variable (*e.g.*, hard coral cover). By modeling the relationship between benthic cover measured at monitoring sites and the corresponding predictor values, it becomes possible to predict benthic cover at unmonitored coral reef sites based on their predictor values. Achieving accurate predictions requires using a large number of predictors, each of which must contain different information.

For many years, the use of predictors in statistical models for macro-ecological studies was constrained by their limited availability at large

spatial scales and by coarse spatio-temporal resolution. In recent years, however, numerous data products—often derived from satellite observations—have become publicly accessible (Gorelick et al., 2017). Combined with advances in computational power and Machine Learning (ML) algorithms (Rubbens et al., 2023), this has enabled the integration of multiple predictors to estimate benthic cover. Accordingly, ML-based modeling approaches are increasingly being applied in coral reef studies (*e.g.*, Bakker et al., 2023; McClanahan et al., 2024).

In this study, we applied a ML approach to estimate temporal trends in the percentage cover of five major benthic categories (hard coral, macroalgae, turf algae, coralline algae, and other fauna), and three hard coral genera (*Acropora*, *Orbicella*, *Porites*) on Caribbean coral reefs from 1980 to 2024. The six main steps of the ML workflow are summarized in **Figure 3.4** and described in detail in the following sections. These sections follow the REFORMS checklist (Kapoor et al., 2024), which provides guidance for reporting ML-based research.



↑ **Figure 3.4.** Main steps of the machine learning (ML) modeling approach used to estimate temporal trends in the percentage cover of five major benthic categories (hard coral, macroalgae, turf algae, coralline algae, and other fauna), and three hard coral genera (*Acropora*, *Orbicella*, *Porites*) on Caribbean coral reefs from 1980 to 2024. *y* denotes the observed percentage cover values (*i.e.*, *measurementValue* in **Table 3.2**) for a given benthic category; *pred.* represents the set of predictors used in the ML model; and ŷ indicates the predicted percentage cover values for a given benthic category across coral reef sites.

5.2.1. Pre-processing

5.2.1.1. Data selection

We extracted benthic cover monitoring data from the *gcrmndb_benthos* synthetic dataset (see section **Data integration**) restricted to sites within the GCRMN Caribbean region.

As this report focuses on the status and trends of shallow coral reefs, we retained surveys conducted at depths of 0–30 m (Rocha et al., 2018). Surveys with missing depth information were retained to minimize data loss.

In addition, some sites not flagged as erroneous during the quality checks of the data integration step (see section **Quality checks**) were removed at this stage based on recommendations from data contributors.

After applying these selection criteria, the dataset used for modeling comprised 72 individual datasets, representing 23,742 surveys conducted across 13,864 sites between 1973 and 2024 (**Table 1.3.1**). Although we modeled benthic cover trends from 1980 to 2024, we included observed data from 1970 to 2024 for model training in order to maximize the dataset size. The number of sites varied among benthic categories, as not all monitoring programs surveyed all categories included in this report. The full list of datasets used per country and territory is provided in **Supplementary Table 1**.

5.2.1.2. Exploratory data analysis

We conducted an exploratory data analysis on the benthic cover dataset to identify potential inconsistencies that could bias model predictions.

We plotted and visually inspected the distribution of percentage cover values (from the variable "measurementValue") for each benthic category across both datasets (datasetID) and maritime areas. When distributions differed notably from those observed in other datasets or areas, we systematically reviewed the corresponding R scripts used for data standardization to ensure that these discrepancies were not the result of data integration errors.

Particular attention was given to the presence of zero values in the measurementValue variable. In the context of coral reef benthic cover, removing zero values can lead to biased temporal trends by overestimating the mean percentage cover of certain categories.

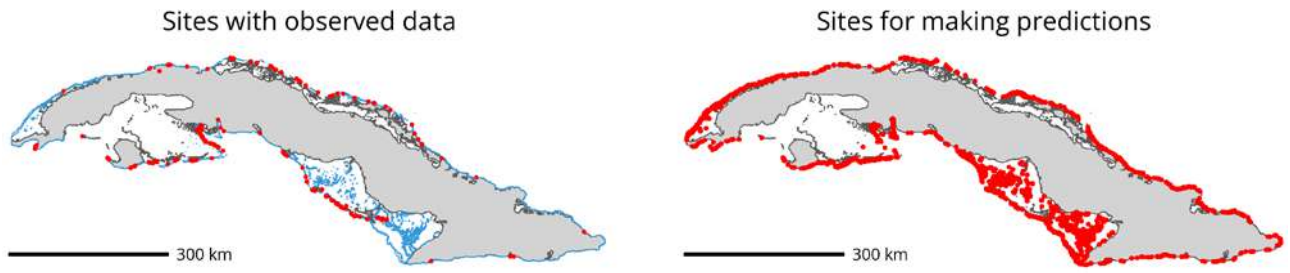
Together with the quality control procedures applied during data integration (see section **Quality checks**), this exploratory data analysis provided an additional means of verifying the overall reliability and consistency of the data.

5.2.1.3. Data transformation

The final pre-processing step consisted of preparing the dataset for use in the modeling analyses. An "area" variable was added to each site through a spatial intersection with the "maritime areas" shapefile specifically created for this report (see section **Maritime areas**). This step was essential to enable the estimation of temporal trends in the percentage cover of benthic categories at the scale of each maritime area (see section **Predictions**). For each sampling unit (*i.e.*, site, transect, or quadrat), percentage cover values were summed for each of the five major benthic categories and the three hard coral genera. To ensure consistency across datasets, the average benthic cover per transect was then calculated, thereby reducing the influence of semi-quantitative photo-quadrat datasets (*e.g.*, those with a limited number of observation points).

5.2.2. Selection of sites to predict

To be able to get estimations of benthic categories' percentage cover over sites representative of coral reefs of the Caribbean, and not on sites where monitoring data were available, it was necessary to select sites on which predictions would be made. To this end, within GEE, we randomly sampled 10,000 sites across the Caribbean coral reefs using the Tropical Coral Reefs of the World dataset (Institute for Marine Remote Sensing et al., 2011). Although the number of sites selected may seem low, each site was later repeated for each year from 1980 to 2024, which means that each model must have predicted almost half a million rows (see section **Predictions**). **Figure 3.5** compares the spatial distribution of sites with observed data with those selected for making predictions using the example of Cuba. Although we assumed that the coral reef distribution shapefile used for this purpose provides a representative depiction of the true spatial distribution of coral reefs within the region, we acknowledge that this working hypothesis may not hold true, particularly at finer spatial scales.



↑ **Figure 3.5.** Comparison of the spatial distributions of sites with observed data and sites used for making predictions, using Cuba as an example. In the left panel figure, coral reef distribution is represented in light blue.

5.2.3. Extraction of predictors

We derived a total of 26 predictors, encompassing various environmental and anthropogenic dimensions: six related to spatial characteristics, one to temporal variation, three to hurricane regime, twelve to thermal regime, two to water quality, and two to direct human influence. These predictors were used both to train and test the models based on sites with observed data and to generate predictions for selected sites (see section **Selection of sites to predict**). The complete list of predictors included in the modeling framework, along with their corresponding data sources, is provided below.

decimalLatitude

- **Category**: Spatial
- **Data source**: gcrmndb_benthos (training) / Tropical Coral Reefs of the World (Institute for Marine Remote Sensing et al., 2011) (prediction)
- **Spatial resolution**: Unknown, but variable (training) / 500 m (prediction)
- **Temporal resolution**: None
- **Description**: The latitude of the site
- **Percentage of missing values**: 0% (training/ testing) and 0% (prediction)

decimalLongitude

- **Category**: Spatial
- **Data source**: gcrmndb_benthos / Tropical Coral Reefs of the World (Institute for Marine Remote Sensing et al., 2011)
- **Spatial resolution**: Unknown, but variable / 500 m
- **Temporal resolution**: None
- **Description**: The longitude of the site
- **Percentage of missing values**: 0% (training/ testing) and 0% (prediction)

verbatimDepth

- **Category**: Spatial
- **Data source**: gcrmndb_benthos / None
- **Spatial resolution**: Unknown, but variable
- **Temporal resolution**: None
- **Description**: The depth at which the survey was conducted
- **Percentage of missing values**: 14.5% (training/ testing) and 100% (prediction)

year

- **Category**: Temporal
- **Data source**: gcrmndb_benthos / Generated
- **Spatial resolution**: None
- **Temporal resolution**: None
- **Description**: The year at which the measurement of percentage cover was done
- **Percentage of missing values**: 0% (training/ testing) and 0% (prediction)

windspeed_y1

- **Category**: Hurricane regime
- **Data source**: International Best Track Archive for Climate Stewardship (IBTrACS) Version 4 (Knapp et al., 2010)
- **Spatial resolution**: 0.1° (~10 km)
- **Temporal resolution**: 3 hourly (interpolated, most raw data are 6 hourly)
- **Description**: Maximum sustained wind speed of cyclones passed within 1° (~ 111 km) from the site during the year preceding year *i*
- **Percentage of missing values**: 11% (training/ testing) and 7% (prediction)

nb_cyclones_y1

- **Category**: Hurricane regime
- **Data source**: International Best Track Archive for Climate Stewardship (IBTrACS) Version 4 (Knapp et al., 2010)
- **Spatial resolution**: 0.1° (~10 km)
- **Temporal resolution**: 3 hourly (interpolated, most raw data are 6 hourly)
- **Description**: Number of cyclones passed within 1° (~ 111 km) from the site during the year preceding year *i*
- **Percentage of missing values**: 11% (training/testing) and 7% (prediction)

cyclones_freq

- **Category**: Hurricane regime
- **Data source**: International Best Track Archive for Climate Stewardship (IBTrACS) Version 4 (Knapp et al., 2010)
- **Spatial resolution**: 0.1° (~10 km)
- **Temporal resolution**: 3 hourly (interpolated, most raw data are 6 hourly)
- **Description**: Annual frequency of cyclones that passed within 1° (~ 111 km) of the site during the period 1980-2024
- **Percentage of missing values**: 11% (training/testing) and 7% (prediction)

elevation

- **Category**: Spatial
- **Data source**: Shuttle Radar Topography Mission (SRTM) Digital Elevation Data Version 4
- **Spatial resolution**: 90 m
- **Temporal resolution**: None (one-shot)
- **Description**: Mean (above-sea) elevation from a 30 km radius from the site
- **Percentage of missing values**: 0.03% (training/testing) and 0% (prediction)

reefextent

- **Category**: Spatial
- **Data source**: Allen Coral Atlas (ACA) Geomorphic Zonation and Benthic Habitat Version 2.0 (Lyons et al., 2024)
- **Spatial resolution**: 5 m
- **Temporal resolution**: None (one-shot)
- **Description**: Total area of coral reef habitat from a 10 km radius from the site
- **Percentage of missing values**: 0.03% (training/testing) and 0% (prediction)

land

- **Category**: Spatial
- **Data source**: Shuttle Radar Topography Mission (SRTM) Digital Elevation Data Version 4
- **Spatial resolution**: 90 m
- **Temporal resolution**: None (one-shot)
- **Description**: Total land area from a 30 km radius from the site
- **Percentage of missing values**: 0.03% (training/testing) and 0% (prediction)

gravity

- **Category**: Human influence
- **Data source**: Cinner et al. (2018)
- **Spatial resolution**: 10 km
- **Temporal resolution**: None (one-shot)
- **Description**: Gravity value on the site. Gravity can be seen as a proxy of direct human impact on coral reefs and is calculated from human population and reef accessibility
- **Percentage of missing values**: 22% (training/testing) and 8% (prediction)

enso

- **Category**: Thermal regime
- **Data source**: Nino 3.4 Index
- **Spatial resolution**: None
- **Temporal resolution**: Monthly
- **Description**: Average Nino 3.4 Index monthly values over the last two years from the year *i*
- **Percentage of missing values**: 2.3% (training/testing) and 4.4% (prediction)

sst_sd

- **Category**: Thermal regime
- **Data source**: NOAA CDR Optimum Interpolation Sea Surface Temperature (OISST) Version 2.1
- **Spatial resolution**: 27,830 m
- **Temporal resolution**: Daily (from 1981-09-01)
- **Description**: Long-term SST standard deviation on the site, derived from 1981 to 2024 daily SST values

- **Percentage of missing values**: 5.1% (training/testing) and 6.3% (prediction)

sst_skewness

- **Category**: Thermal regime
- **Data source**: NOAA CDR Optimum Interpolation Sea Surface Temperature (OISST) Version 2.1
- **Spatial resolution**: 27,830 m
- **Temporal resolution**: Daily (from 1981-09-01)
- **Description**: Long-term SST skewness on the site, derived from 1981 to 2024 daily SST values
- **Percentage of missing values**: 5.1% (training/testing) and 6.3% (prediction)

sst_max

- **Category**: Thermal regime
- **Data source**: NOAA Coral Reef Watch (Skirving et al., 2020)
- **Spatial resolution**: 5,000 m
- **Temporal resolution**: Yearly (from 1985)
- **Description**: Maximum annual SST values on the site for the year *i*
- **Percentage of missing values**: 10.1% (training/testing) and 11.3% (prediction)

sst_max_y1

- **Category**: Thermal regime
- **Data source**: NOAA Coral Reef Watch (Skirving et al., 2020)
- **Spatial resolution**: 5,000 m
- **Temporal resolution**: Yearly (from 1985)
- **Description**: Maximum annual SST values on the site for the year i-1. This lagged version of the predictor "sst_max" was included to take into account the time needed for the percentage cover to be impacted by the event(s).
- **Percentage of missing values**: 10.1% (training/testing) and 13.5% (prediction)

sst_mean

- **Category**: Thermal regime
- **Data source**: NOAA Coral Reef Watch (Skirving et al., 2020)
- **Spatial resolution**: 5,000 m
- **Temporal resolution**: Yearly (from 1985)
- **Description**: Mean annual SST values on the site for the year i
- **Percentage of missing values**: 10.1% (training/testing) and 11.3% (prediction)

sst_mean_y1

- **Category**: Thermal regime
- **Data source**: NOAA Coral Reef Watch (Skirving et al., 2020)
- **Spatial resolution**: 5,000 m
- **Temporal resolution**: Yearly (from 1985)
- **Description**: Mean annual SST values on the site for the year i-1. This lagged version of the predictor "sst_mean" was included to take into account the time needed for the percentage cover to be impacted by the event(s).
- **Percentage of missing values**: 10.1% (training/testing) and 13.5% (prediction)

sst_min

- **Category**: Thermal regime
- **Data source**: NOAA Coral Reef Watch (Skirving et al., 2020)
- **Spatial resolution**: 5,000 m
- **Temporal resolution**: Yearly (from 1985)
- **Description**: Minimum annual SST values on the site for the year *i*
- **Percentage of missing values**: 10.1% (training/testing) and 11.3% (prediction)

ssta_mean

- **Category**: Thermal regime
- **Data source**: NOAA Coral Reef Watch (Skirving et al., 2020)
- **Spatial resolution**: 5,000 m
- **Temporal resolution**: Yearly (from 1985)
- **Description**: Mean annual SST anomaly values on the site for the year *i*
- **Percentage of missing values**: 10.1% (training/testing) and 11.3% (prediction)

ssta_max

- **Category**: Thermal regime
- **Data source**: NOAA Coral Reef Watch (Skirving et al., 2020)
- **Spatial resolution**: 5,000 m
- **Temporal resolution**: Yearly (from 1985)

- **Description**: Maximum annual SST anomaly values on the site for the year *i*
- **Percentage of missing values**: 10.1% (training/testing) and 11.3% (prediction)

dhw_max

- **Category**: Thermal regime
- **Data source**: NOAA Coral Reef Watch (Skirving et al., 2020)
- **Spatial resolution**: 5,000 m
- **Temporal resolution**: Yearly (from 1986)
- **Description**: Maximum Degree Heating Week (DHW) value on the site for the year *i*
- **Percentage of missing values**: 10% (training/testing) and 13.5% (prediction)

dhw_max_y1

- **Category**: Thermal regime
- **Data source**: NOAA Coral Reef Watch (Skirving et al., 2020)
- **Spatial resolution**: 5,000 m
- **Temporal resolution**: Yearly (from 1986)
- **Description**: Maximum Degree Heating Week (DHW) value on the site for the year i-1. This lagged version of the predictor "dhw_max" was included to take into account the time needed for the percentage cover to be impacted by the event(s).
- **Percentage of missing values**: 10% (training/testing) and 15.7% (prediction)

chla_mean

- **Category**: Water quality
- **Data source**: Ocean Color SMI: Standard Mapped Image MODIS Aqua Data
- **Spatial resolution**: 4,616 m
- **Temporal resolution**: Unknown
- **Description**: Long-term average of chlorophyll *a* concentration (in mg/m³) from a 10 km radius from the site from 2002-01-01 to 2022-12-31.
- **Percentage of missing values**: 2.6% (training/testing) and 4.3% (prediction)

chla_sd

- **Category**: Water quality
- **Data source**: Ocean Color SMI: Standard Mapped Image MODIS Aqua Data
- **Spatial resolution**: 4,616 m
- **Temporal resolution**: Unknown
- **Description**: Long-term standard deviation of chlorophyll *a* concentration (in mg/m³) from a 10 km radius from the site from 2002-01-01 to 2022-12-31.
- **Percentage of missing values**: 2.6% (training/testing) and 4.3% (prediction)

population

- **Category**: Human influence
- **Data source**: Gridded Population of the World (GPW) Version 4.11
- **Spatial resolution**: 927.67 m
- **Temporal resolution**: Yearly (for years 2000, 2005, 2010, 2015, and 2020). Missing values for the years 2000–2024 were estimated using a linear model.
- **Description**: Total number of inhabitants living within 5 km from the site on the year *i*
- **Percentage of missing values**: 4.8% (training/testing) and 44.4% (prediction)

After extracting the predictor values, we examined the number of missing values for each predictor and each year. This control step aimed to identify potential issues in the extraction process that could result in an abnormally high number of missing values, thereby reducing the amount of usable information available for models development (*i.e.* training and testing) and prediction.

The performance of GBM models is generally unaffected by the inclusion of non-informative predictors, as these models inherently perform feature selection during training (Kuhn and Johnson, 2013). Feature selection involves identifying and removing highly correlated predictors, which represent redundant sources of information (Boehmke and Greenwell, 2020). Although this step is not strictly necessary for GBM models, we examined the Pearson correlation coefficients among all predictors using pairwise linear regressions and excluded variables with correlation coefficients greater than 0.90 or lower than −0.90. This optional procedure was implemented primarily to reduce model computation time. As a result of this process, the predictor "sst_mean_y1" was removed, leaving a total of 25 predictors used in the final models.

The high predictive potential of ML models is often constrained by data availability, particularly the ability to integrate a sufficient number of predictors at adequate spatio-temporal resolutions. Not all factors known to influence the percentage cover of benthic categories can be incorporated into the modeling framework due to data limitations. For instance, although Stony Coral Tissue Loss Disease (SCTLD) has been identified as a major driver of hard coral mortality (Alvarez-Filip et al., 2022), the scarcity of monitoring programs quantifying SCTLD prevalence prevents its inclusion as a predictor variable.

Beyond data availability, the spatio-temporal resolution of predictors also represents a critical limitation. We assumed that the extracted predictor values accurately reflected in situ conditions—those that would have been measured directly at a given site and time. However, we acknowledge that the resolution of some predictors is insufficient to capture small-scale variability. For example, substantial differences in sea surface temperature (SST) may occur within a single 5 km x 5 km pixel (Green et al., 2019), which corresponds to the spatial resolution of NOAA's CoralTemp dataset (Skirving et al., 2020). Moreover, by definition, SST data do not account for temperature variations with depth, which can nevertheless be considerable (Leichter et al., 2006).

5.2.4. Machine learning models

Machine learning encompasses a wide range of methods (*e.g.*, support vector machines, random forests) that can be applied to classification or regression tasks (Boehmke and Greenwell, 2020). Among these, we selected Gradient Boosting Machines (GBM, also referred to as Boosted Regression Trees), and specifically the XGBoost algorithm, due to its high predictive performance (Boehmke and Greenwell, 2020; Chen and Guestrin, 2016).

Contrary to other modelling approaches such as Bayesian models, like the one used for the "*Status of Coral Reefs of the World: 2020*" GCRMN report (Souter et al., 2021), ML models are not intrinsically built to consider uncertainty (Rubbens et al., 2023). However, confidence intervals can be estimated for ML models using bootstrap. Bootstrapping consists of sampling randomly the initial dataset with replacement, to produce variability within the new dataset (Boehmke and Greenwell, 2020). This means that the number of initial rows remained the same, but that some rows were either removed, present one time, or duplicated (one or several times) into the new dataset, due to the sampling. For the purpose of this report, we used a stratified bootstrap per datasetID and year. This was done to obtain more representative confidence intervals and stabilize estimates.

A key challenge in ML is overfitting, where a model performs extremely well on the training dataset but poorly on new, unseen data (Kuhn and Johnson, 2013). A common strategy to reduce overfitting is to partition the dataset into a training set and a testing set. The training set is used to fit the model, while the testing set is used to evaluate predictive accuracy on independent data. For this study, we applied a 75/25 split, assigning 75% of observations (*i.e.*, rows) to the training dataset and 25% to the testing dataset.

In addition, GBM models include multiple hyperparameters that must be tuned to optimize predictive performance and limit overfitting. Model tuning involves identifying the best combination of hyperparameters that maximize predictive accuracy. For each bootstrap iteration, we tested 10 combinations of five hyperparameters: learning rate, number of trees, tree depth, minimum observations per leaf, and loss reduction. The learning rate (also referred to as shrinkage) controls the step size during gradient descent; the number of trees specifies the ensemble size; tree depth defines the maximum number of levels in each tree; minimum observations per leaf sets the smallest number of observations required to form a terminal node; loss reduction (also known as gamma) sets the structural complexity of the trees. The 10 hyperparameter combinations were selected to ensure broad coverage of the hyperparameter space (*i.e.*, maximizing entropy). Model performance during hyperparameter tuning was assessed using 3-fold cross-validation on the training dataset.

We fitted 20 ML models, one corresponding to a bootstrap iteration, for each of the five major benthic categories (hard coral, macroalgae, turf algae, coralline algae, and other fauna), and three hard coral genera (*Acropora*, *Orbicella*, *Porites*), by training the model on a new dataset, generated through bootstrapping. Although we could have kept the same hyperparameters from one bootstrap iteration to another, which would have saved a lot of computation time, we decided to rerun the hyperparameter tuning at each iteration to stabilize the model results and make them less sensitive to the inclusion or removal of new data.

5.2.5. Model evaluation

5.2.5.1. Model performance

The Root Mean Square Error (RMSE) is the most common metric used to evaluate the performance of ML regression models. Since RMSE is expressed in the same unit as the variable predicted, it represents an easy-to-interpret metric (Kuhn and Johnson, 2013). For example, a RMSE value of 5 means that the value predicted by the model differed, on average, by 5 percent of hard coral cover. Hence, the lower the RMSE value, the better the model's predictive performance. We calculated the RMSE as follows:

Eq. 3.6

$$RMSE = \sqrt{\sum_{i=1}^{n} \frac{\left(\hat{y}_i - y_i\right)^2}{n}}$$

where n is the number of observations, y_i is the observed value i, and $\hat{y}_i$ the predicted value i. In addition to the RMSE, we also calculated the R^2 (also known as R-squared or coefficient of determination), which is a widely used metric in statistics, particularly for linear regression. R^2 is usually interpreted as the proportion of the information from the data that is explained by the model (Kuhn and Johnson, 2013). For example, an R^2 of 0.80 indicates that the model explains 80% of the variation of the data. We calculated R^2 as follows

Eq. 3.7

$$R^2 = 1 - \frac{\sum_i \left(y_i - \hat{y}_i\right)^2}{\sum_i \left(y_i - \bar{y}\right)^2}$$

where n is the number of observations, y_i is the observed value i, $\hat{y}_i$ the predicted value i, and $\underline{y}$ the mean of observed values. RMSE and R^2 represent two complementary metrics of model performance. Indeed, RMSE describes how well the model performs on average, while R^2 provides a measure of correlation (Kuhn and Johnson, 2013).

Metrics for regression models such as RMSE and R^2 are not sufficient alone to estimate the performance of a ML model. Indeed, these metrics provide a single value to evaluate the overall performance of the ML model and can mask poor predictions over certain ranges of variation of the variable to be predicted. For this reason, we plotted predicted versus observed values on the testing dataset and we checked for the distribution of residuals.

5.2.5.2. Model interpretation

Although ML models are sometimes described as "black box" models, there exists in fact different tools to interpret these models (Biecek and Burzykowski, 2021; Boehmke and Greenwell, 2020). We used variable importance plots (VIMP, see page 207 in Kuhn and Johnson, 2013) to compare the relative influence of each predictor on the percentage cover of each of the five major benthic categories and the three hard coral genera. However, we restricted our interpretations of the VIMP to model evaluation rather than using them for their explicative capabilities. Hence, interpretation of VIMP can also be seen as a third way to control data quality, along quality checks (see section **Quality checks**) and exploratory data analysis (see section **Exploratory data analysis**).

5.2.6. Predictions

We predicted values of percentage cover for each of the five major benthic categories and the three hard coral genera, across 10,000 sites over the Caribbean. For each site, values were predicted for 44 years (*i.e.* from 1980 to 2024), resulting in a total of 440,000 predictions of percentage cover values performed by each ML model. These predictions were obtained for the 20 bootstrap iterations. Because we were interested in temporal trends at a higher spatial scale than sites, we averaged the predictions per area or over the entire Caribbean region, for each year and each bootstrap iteration. Then, we calculated mean percentage cover for each of the five major benthic categories and the three hard coral genera, for each year across the 20 bootstrap iterations. In addition, we calculated the 95% upper and lower bounds of confidence intervals around the mean by using the 0.975 and 0.025 quantiles, respectively.

Finally, we computed Mann-Kendall statistical tests on the mean percentage cover values estimated by the ML models from 1980 to 2024 for each of the five major benthic categories and the three hard coral genera. These tests were assessed using a significance level of $\alpha = 0.05$, to evaluate if trends were present.

6. CODE AVAILABILITY

All scripts used to generate the values, tables, figures, and maps presented in this report are available at https://github.com/GCRMN/caribbean_2024. Data analyses relying on Google Earth Engine (Gorelick et al., 2017) were implemented in JavaScript. Machine learning models were executed using R scripts on an RStudio Server hosted on a Google Cloud virtual machine (50 GB storage, 64 GB RAM, 8 CPUs, Ubuntu). Other R scripts were run on a Dell Inspiron 7780 personal computer (32 GB RAM, Windows 11). Most analyses were conducted in R version 4.5.1 (R Core Team, 2025), primarily using the packages *tidyverse* v2.0.0 (Wickham et al., 2019), *tidymodels* v1.3.0 (Kuhn and Wickham, 2020), and *sf* v1.0-21 (Pebesma et al., 2023). The complete list of R packages used and their associated versions is provided in the README file of the GitHub repository.

AUTHORS' CONTRIBUTIONS

A contributor is defined as any individual who directly or indirectly supported the development of this report. To reflect the diversity and varying extent of these contributions, we applied a modified version of the Contributor Role Taxonomy (CRediT). Thirteen distinct contribution types were identified, covering the full range of activities undertaken from the report's inception to its final publication. In the table below, an 'X' indicates that a contributor engaged in the activity listed in the column header, while 'W' and 'R' denote involvement in writing or reviewing the corresponding section, respectively. 'Supervision' covers all project-management duties, including coordinating the work and organizing meetings with the editorial team. 'Facilitation' refers to support provided during specific stages of the report process, such as organising a workshop. 'Data acquisition' refers specifically to the collection of benthic cover coral reef monitoring data. Given the large number of individuals involved in the acquisition of AGRRA data, their contributions are not listed in this table but are instead detailed in the chapters of Part 2 of the report. 'Participation in workshop' denotes the in-person attendance of an author at the workshop which was held in Puerto Morelos, Mexico, from 1 to 4 October 2024. 'Communication' encompasses all activities designed to disseminate and promote the report's findings, such as participating in the webinar or drafting the policy brief.

First name	Last name	Funding acquisition	Supervision	Facilitation	Data acquisition	Data analysis	Participation in workshop	Executive summary	Synthesis for the region	Syntheses for count. and terr.	Case studies	Materials and Methods	Supplementary Materials	Communication
Andrés Felipe	**Acosta-Chaparro**				X				R	W				
Esteban	**Agudo-Adriani**				X									
Gabriela	**Aguilera Pérez**				X					W				
Andrés B.	**Alegría**									W				
Jahson B.	**Alemu I**				X				R	W				
Mary	**Allen**										W			
David Alejandro	**Alonso-Carvajal**				X									
Juan José	**Alvarado Barrientos**				X				R	W				
Lorenzo	**Alvarez-Filip**				X		X		R			R		
Robert	**Anderson**				X					W				
Louise	**Anderson**							R	R	W				
Shamwari	**Anseeuw**				X									
Maria G.	**Aponte**				X									
Eric	**Appeldoorn Sanders**				X					W				
Melissa	**Arbona Fernández**				X					W				
Jesus Ernesto	**Arias-Gonzalez**				X			R		W				
Francisco	**Arias Isaza**				X									
Alfredo	**Ascanio**				X									
Rolf P.M.	**Bak**				X									
Robert	**Balza**				X									
James	**Beets**										W			
Chantale	**Bégin**				X					W				

First name	Last name	Funding acquisition	Supervision	Facilitation	Data acquisition	Data analysis	Participation in workshop	Executive summary	Synthesis for the region	Syntheses for count. and terr.	Case studies	Materials and Methods	Supplementary Materials	Communication
Aude	**Berger**				X									
Tadzio	**Bervoets**				X									
Hadrien	**Bidenbach**									W				
Macarena	**Blanco-Pimentel**				X									
Max	**Bodmer**				X									
Mélanie	**Bon**				X				R	W				
Clément	**Bonnardel**				X									
Amandine	**Bordin**				X									
Claude	**Bouchon**				X				R	W				
Yolande	**Bouchon-Navaro**				X				R	W				
Robert	**Bourne**				X									
Leo	**Brewster**				X									
Sophie	**Brugneaux**				X									
Frances	**Budd**				X									
Eleanor	**Butler**				X					W				
Hansel	**Caballero-Aragón**									W				
Johanna	**Calle-Triviño**				X				R	W				
Ruleo	**Camacho**									W				
Shannon	**Cameron**				X									
Lisa	**Carne**									W				
Sophie	**Carolan**				X									
Juan Carlos	**Carrasco Navas-Parejo**									W				
Julien	**Chalifour**				X					W				
Pedro	**Chevalier Monteagudo**									W				
Dorka	**Cobian Rojas**				X					W				
Alistair	**Cole**				X									
Michael	**Colella**				X									
Raul Igor	**Corrada Wong**				X		X			W				
Jorge	**Cortés**									W				
Amy	**Cox**				X									
Annabel	**Cox**				X									
Jacque	**Cresswell**				X									
Aldo	**Croquer**			X	X		X	R	R	W	W			
Lizeth	**Cruz**				X									
Monique	**Curtis**				X					W				
Thomas	**Dallison**			X			X	R	R					X
Carlos	**Daza-Guerra**				X									
Didier M.	**de Bakker**				X					W				
Mareike	**De Breuyn**				X				R	W				

First name	Last name	Funding acquisition	Supervision	Facilitation	Data acquisition	Data analysis	Participation in workshop	Executive summary	Synthesis for the region	Syntheses for count. and terr.	Case studies	Materials and Methods	Supplementary Materials	Communication
Danielle	**de Kool**				X					W				
Elena	**de La Guardia Llansó**									W				
Ramón	**de León**									W	W			
John	**DeBuysser**				X				R	W				
Ethan	**Degner**				X									
Michael A.	**Del Rosario**				X									
Gilles	**Delouche**				X									
Michelle M.	**Dennis**									W				
Catherine	**Desrosiers**									W				
Charlotte R.	**Dromard**				X					W				
Damaris Elizabeth	**Dueñas**									W				
Priscilla	**Dupont**				X									
Alain	**Duran**									W				
Alicia	**Eck-Nunez**									W				
Caren	**Eckrich**				X									
Graham J.	**Edgar**				X									
Peter J.	**Edmunds**				X									
Olivia	**Eisenbach**				X									
Roxana C.	**Elorriaga**										W			
Nuria	**Estrada Saldívar**									W				
Dan	**Exton**				X									
Cindy	**Fernández-García**				X									
Miguel	**Figuerola Hernández**				X									
Graham	**Forrester**				X					W				
Patrice	**Francis**				X		X		R	W				
Jan	**Freiwald**				X									
Donavon	**French**				X									
Alan M.	**Friedlander**										W			
Victor	**Galván**				X									
Rebecca	**Garcia-Camps**				X					W	W			
Joaquin Rodrigo	**Garza**				X									
Solomon	**Gibson**						X							
Ildiko	**Gilders**									W				
David	**Gilliam**				X									
Ana	**Giro**									W				
Diana Isabel	**Gómez López**				X									
Catalina	**Gómez-Cubillos**				X									
Melina R.	**Gonzáles**				X					W	W			
Patricia	**González Díaz**				X					W				

First name	Last name	Funding acquisition	Supervision	Facilitation	Data acquisition	Data analysis	Participation in workshop	Executive summary	Synthesis for the region	Syntheses for count. and terr.	Case studies	Materials and Methods	Supplementary Materials	Communication
Juliett	**González Méndez**				X					W				
Sergio	**González-Ferrer**									W				
Gretchen	**Goodbody-Gringley**				X					W				
Renata	**Goodridge**				X									
Sébastien	**Gréaux**				X					W				
Gabriel	**Grimsditch**	X					X	R	R					
Sergio D.	**Guendulain-García**				X						W			
Geury	**Guerrero**				X									
Claudia Alejandra	**Guerrero**									W				
Tessa	**Haanskorf**				X									
Andreas	**Haas**				X									
Toby	**Haddon**				X									
Ryan	**Hannum**				X									
Julie	**Hawkins**				X									
Sarah	**Heidmann**				X						W			
Katalina	**Henao Orozco**				X									
Alex	**Henderson**				X									
James	**Henkel**				X									
Scott	**Heppell**										W			
Ana Carolina	**Hernández-Oquet**				X					W				
Joan	**Hernández Albernas**				X					W				
Pilar	**Herrón**				X									
Heidi	**Hertler**				X									
Leslie	**Hickerson**				X									
Jenny	**Hickman**				X									
Bert W.	**Hoeksema**									W				
Argel	**Horton**									W				
Alfredo	**Howard**						X							
Shoneé	**Howell**				X				R	W				
Alex	**Hunt**									W				
Alwin	**Hylkema**				X			R	R	W				
Walter	**Hynds**				X									
Iker	**Irazabal**				X						W			
Ayumi	**Izioka**				X									
Cecil	**Jennings**										W			
Jack	**Johnson V**				X					W				
Michelle A.	**Johnston**				X					W				
Amy	**Jones**								R		W			
Laurent	**Juhel**				X									

First name	Last name	Funding acquisition	Supervision	Facilitation	Data acquisition	Data analysis	Participation in workshop	Executive summary	Synthesis for the region	Syntheses for count. and terr.	Case studies	Materials and Methods	Supplementary Materials	Communication
Elizabeth	**Kadison**										W			
Mohsen	**Kayal**				X				R	W				
Thomas	**Kemenes van Uden**				X				R	W	W			
Kimani	**Kitson-Walters**									W				
Oliver	**Klokman**				X									
Patricia	**Kramer**			X	X				R					
Nicole	**Krampitz**				X									
Florian	**Labadie**				X									
Lucie	**Labbouz**		X											X
Jean Claude	**Lalanne**				X									
Damaris	**Landers**				X									
Judith	**Lang**							R	R	W				
Sophie	**Le Loc'h**				X									
Raoul	**Lebrave**				X									
Hélène	**Legrand**				X									
Marnik	**Lehwald**				X									
Sarah	**Leinbach**				X									
Franciane	**Lequellec**				X									
Sandra	**Little**				X									
Max	**Louis**				X									
Natalie	**Lubbock**				X									
Josianne	**Mahieu**				X									
Jennie	**Mallela**				X				R	W				
Jean-Philippe	**Maréchal**				X				R	W		R		
Gloria	**Marino**				X									
Tom	**Martin**				X									
Raphael	**Martinez**									W				
Juan	**Mate**									W				
Laëtitia	**Mathon**		X		X		X	W	W	W		R		X
Croy	**Mccoy**				X									
Melanie	**McField**		X	X	X		X	R	R	W				X
Alexander R.	**McLeod**				X									
Bryony	**Meakins**			X			X							
Erik H.	**Meesters**				X				R	W		R		
Daniela	**Mejia**									W				
Sandra	**Mendoza**									W				
Jenny	**Mihaly**				X									
Emy	**Miyazawa**				X						W			
Ana	**Molina-Hernández**										W			

First name	Last name	Funding acquisition	Supervision	Facilitation	Data acquisition	Data analysis	Participation in workshop	Executive summary	Synthesis for the region	Syntheses for count. and terr.	Case studies	Materials and Methods	Supplementary Materials	Communication
Courtney	**Moll**				X									
Belkis Carolina	**Montalván Torres**									W				
Luis Miguel	**Montilla**				X									
Judith	**Morales**										W			
Clare	**Morrall**				X									
Amy	**Morrison**				X									
Aaron Israel	**Muñiz-Castillo**						X		R	W		R		
Thaddeus	**Murdoch**				X									
Mylène	**Musquet**				X									
Tamarah	**Myles**				X									
Raúl	**Navas-Camacho**				X									
Sobeida	**Nazaret Nuñez**									W				
Richard S.	**Nemeth**								R	W	W			
Gerard	**Nieuwland**				X									
Stephen	**Nimrod**				X									
Ernst	**Noyons**				X									
Maggy	**Nugues**				X									
Marissa	**Nuttall**				X					W				
Alex	**O'Brien**				X				R	W				
Kelly	**O'Connell**				X				R	W				
Lizzi	**Oh**				X									
Lauren	**Olinger**				X					W				
Vincent	**Oliva**				X									
Felipe	**Ossandon**				X									
Hazel	**Oxenford**								R					
Fernando J.	**Pardo Urrutia**				X									
John Everett	**Parkinson**				X					W				
Rebecca Hamel	**Pepin**				X									
Susana	**Perera-Valderrama**									W				
Helder	**Perez**									W				
Cécile	**Pérez**				X									
Antar M.	**Pérez-Botello**										W			
Esmeralda	**Perez-Cervantes**				X					W				
Auriane	**Petit**	X	X	X			X	R	W					X
Maxime	**Philip**								R					
Valeria	**Pizarro**				X			R	R	W	W			
Pedro	**Portillo**				X									
Julian	**Prato-Valderrama**									W				
Fabio	**Quesada-Perez**				X		X							

First name	Last name	Funding acquisition	Supervision	Facilitation	Data acquisition	Data analysis	Participation in workshop	Executive summary	Synthesis for the region	Syntheses for count. and terr.	Case studies	Materials and Methods	Supplementary Materials	Communication
Karl	**Questel**				X									
Melissa	**Radley**				X									
Amanda	**Ramos Romero**				X					W				
Angela	**Randazzo Eisemann**				X					W	W			
Doug	**Rasher**										W			
Chanel	**Raynor**				X									
Bernard	**Renaudie**				X									
Mariana	**Reyna**				X				R		W			
Charlie	**Riddock**				X									
Chris	**Riley**				X									
Rodolfo	**Rioja-Nieto**									W				
Andreina	**Rivera**				X						W			
Andrea	**Rivera-Sosa**		X		X		X	R	R	W				X
Callum	**Roberts**				X									
Arlene	**Rodriguez**									W				
Claire R.	**Rosemond**										W			
Lynnette	**Roth**			X	X									
Yann	**Rousseau**				X									
Jasmin	**Ruan**				X					W				
Miguel Angel	**Ruiz Zarate**									W				
Hannah	**Rushton**				X									
Rob	**Ruzicka**				X					W				
Joe	**Ryan**									W				
Laura	**Sánchez-Valencia**				X									
Léa	**Sebesi**				X									
Rita I.	**Sellares-Blasco**				X					W	W			
Claude	**Séraline**				X									
Paula	**Sills**				X			R			W			
Nuno	**Simoes**										W			
Micaela	**Small**				X									
Tyler B.	**Smith**				X		X			W	W			
Struan Robertson	**Smith**				X					W				
Melina	**Soto**			X			X			W				
Robert	**Steneck**				X				R	W	W			
Stephanie	**Stinson**				X									
Sandin	**Stuart**				X									
Rick D.	**Stuart-Smith**				X									
Ella	**Tabares**				X									
Maureen	**Tejeda Metz**				X									

First name	Last name	Funding acquisition	Supervision	Facilitation	Data acquisition	Data analysis	Participation in workshop	Executive summary	Synthesis for the region	Syntheses for count. and terr.	Case studies	Materials and Methods	Supplementary Materials	Communication
Victor	**Tersiguel**				X									
Rubben	**Torres**				X					W	W			
Juliana	**Torres Acero**									W				
Erica K.	**Towle**			X	X			R	R	W			R	
Ewan	**Trégarot**				X									
Holly	**Trew**				X									
Sydney	**Tritschler**				X				R					
Sanne	**Tuijten**				X									
Andreina	**Valdez-Trinidad**				X						W			
Henri	**Valles**				X									
Matthijs	**van der Geest**				X					W				
Marijn	**van der Laan**				X									
Alex	**van der Last**				X									
Saskia	**van Dongen**				X									
Juliana	**Vanegas Gonzalez**				X				R	W	W			
Estefany	**Vargas-Pérez**				X									
Maria	**Vega Rodriguez**				X									
Alejandro	**Vega Zepeda**									W				
Anna	**Venema**				X									
Alejandra	**Verde**				X									
Mark	**Vermeij**				X									
Angelica	**Verschragen**				X									
Patricia	**Vicente Castro**				X					W				
Shay	**Viehman**									W				
Maria F.	**Villalpando**				X					W	W			
Estrella	**Villamizar**				X					W				
Julian	**Walcott**				X									
Joseph	**Weekes**				X									
Elizabeth	**Whiteman**										W			
Jérémy	**Wicquart**		X			X	X	W	W	W		W	W	X
Jean	**Wiener**									W				
Ross	**Williamson**				X									
Margaret	**Wilson**				X									
Anaurora	**Yranzo-Duque**									W				
Someira	**Zambrano**				X									
Yamil	**Zapata**									W				
Sven	**Zea**				X									
Calina	**Zepeda-Centeno**									W				
Ainhoa Leon	**Zubillaga**				X					W	W			

BIBLIOGRAPHY

Abello, P., Restrepo, G., & Pérez, S. (2021). A Conservation Project through Stock Restoration for the Queen Conch, Strombus gigas Linnaeus, 1758, on some of the San Andrés and Old Providence Archipelago's MPA'S.

ACF (2025). Natural Values and Anthropogenic Impacts on Reef Islands of Aruba. Aruba Conservation Foundation

Acosta, C. A. (2006). Impending trade suspensions of Caribbean queen conch under CITES: a case study on fishery impacts and potential for stock recovery. Fisheries, 31(12), 601-606.

Acosta, C., & Robertson, D. (2003). Comparative spatial ecology of fished spiny lobsters Panulirus argus and an unfished congener P. guttatus in an isolated marine reserve at Glover's Reef atoll, Belize. Coral Reefs, 22, 1-9.

Adam, T. C., Burkepile, D. E., Holbrook, S. J., Carpenter, R. C., Claudet, J., Loiseau, C., ... & Schmitt, R. J. (2021). Landscape-scale patterns of nutrient enrichment in a coral reef ecosystem: implications for coral to algae phase shifts. Ecological Applications, 31(1), e2227.

Adam, T. C., Burkepile, D. E., Ruttenberg, B. I., & Paddack, M. J. (2015). Herbivory and the resilience of Caribbean coral reefs: knowledge gaps and implications for management. Marine Ecology Progress Series, 520, 1-20.

Agostini, V. N., Margles, S. W., Knowles, J. K., Schill, S. R., Bovino, R. J., & Blyther, R. J. (2015). Marine zoning in St. Kitts and Nevis: a design for sustainable management in the Caribbean. Ocean & Coastal Management, 104, 1-10.

Agudo-Adriani, E. A., Cappelletto, J., Cavada-Blanco, F., & Cróquer, A. (2019). Structural complexity and benthic cover explain reef-scale variability of fish assemblages in Los Roques National Park, Venezuela. Frontiers in Marine Science, 6, 690.

Aguilar-Perera, A. (2006). Disappearance of a Nassau grouper spawning aggregation off the southern Mexican Caribbean coast. Marine Ecology Progress Series, 327, 289-296. 10.3354/meps327289

Aguilera-Pérez, G. C., & González-Díaz, P. (2024). Salud de los corales y su investigación en el Caribe y en Cuba. Revista de Investigaciones Marinas, 44, 18-46.

Ajagunna, I., Ilori, M. O., & McLean, E. (2022). An analysis of post-pandemic scenarios: what are the prospects for the Caribbean cruise industry?. Worldwide Hospitality and Tourism Themes, 14(2), 91-98.

Ajwang, V. (2024). Impact of Pollution on Coral Health in Coastal Reef Ecosystems. American Journal of Natural Sciences, 5(1), 33-43.

Akhtar, R., Sirwal, M.Y., Hussain, K., Dar, M.A., Shahnawaz, M., Daochen, Z. (2022). Impact of Plastic Waste on the Coral Reefs: An Overview. In: Shahnawaz, M., Sangale, M.K., Daochen, Z., Ade, A.B. (eds) Impact of Plastic Waste on the Marine Biota. Springer, Singapore.

Akita, Y., Kurihara, T., Uehara, M., Shiwa, T., & Iwai, K. (2022). Impacts of overfishing and sedimentation on the feeding behavior and ecological function of herbivorous fishes in coral reefs. Marine Ecology Progress Series, 686, 141-157.

Albins, M. A. (2013). Effects of invasive Pacific red lionfish Pterois volitans versus a native predator on Bahamian coral-reef fish communities. Biological Invasions, 15, 29-43.

Albins, M. A., & Hixon, M. A. (2008). Invasive Indo-Pacific lionfish Pterois volitans reduce recruitment of Atlantic coral-reef fishes. Marine Ecology Progress Series, 367, 233-238.

Albright, R., & Langdon, C. (2011). Ocean acidification impacts multiple early life history processes of the Caribbean coral Porites astreoides. Global change biology, 17(7), 2478-2487.

Albright, R., Mason, B., Miller, M., & Langdon, C. (2010). Ocean acidification compromises recruitment success of the threatened Caribbean coral Acropora palmata. Proceedings of the National Academy of Sciences, 107(47), 20400-20404.

Alcolado, P. M., Hernández-Muñoz, D., Caballero-Aragón, H., Busutil, L., Perera, S., & Hidalgo, G. (2009). Efectos de un inusual período de alta frecuencia de huracanes sobre el bentos de arrecifes coralinos. Revista Ciencias Marinas y Costeras, 1, 73-93.

Alcolado, P. M. (2003). Diagnosis of some key marine and coastal ecosystem issues of saint kitts, and management recommendations.

Alcolado, P. M., Claro, R., Martínez-Daranas, B., Menéndez-Macías, G., García-Parrado, P., Cantelar-Ramos, K., ... & Neff, T. (2000). Estado general de los arrecifes coralinos de Cuba y propuestas de manejo ambiental. Informe Final del Proyecto. Instituto de Oceanología, La Habana.

Alegría-Ortega, A., Sanín-Pérez, M. J., Quan-Young, L. I., & Londoño-Mesa, M. H. (2021). Genetic structure of Orbicella faveolata population reveals high connectivity among a marine protected area and Varadero Reef in the Colombian Caribbean. Aquatic Conservation: Marine and Freshwater Ecosystems, 31(4), 764-776.

Alemu, J. B., & Mallela, J. (2021). Recent dynamics on turbid-water corals reefs following the 2010 mass bleaching event in Tobago. Marine Environmental Research, 170, 105411.

Allen, M.E., Fleming, C.S., Alva, A.T., Gonyo, S.B., Regan, S.D., and E.K. Towle. (2024). National Coral Reef Monitoring Program Socioeconomic Monitoring Component: Summary Findings for Puerto Rico, 2022. U.S. Dep. Commerce, NOAA Tech. Memo., NOAA-TM-NOS-CRCP-49, 42p. + Appendices. DOI: 10.25923/893p-7f36.

Allen, M.E., Fleming, C.S., Zito, B.M., Gonyo, S.B., Regan, S.D., and E.K. Towle. (2021). National Coral Reef Monitoring Program Socioeconomic Monitoring Component: Summary Findings for South Florida, 2019. US Dep. Commerce, NOAA Tech. Memo., NOAA-TM-NOS-CRCP-39, 60p. + Appendices. DOI:10.25923/w6pa-hv50.

Alleyne, K. S., Small, M., Corbin, M., Vallès, H., & Oxenford, H. A. (2023). Free-swimming fauna associated with influxes of pelagic sargassum: Implications for management and harvesting. Frontiers in Marine Science, 10, 1090742.

Almada-Villela, P., McField, M., Kramer, P., et al. (2002). Status of coral reefs of Mesoamerica: Mexico, Belize, Guatemala, Honduras, Nicaragua and El Salvador. In C. Wilkinson (Ed.), Status of Coral Reefs of the World: 2002 (pp. 303–324). Townsville, Queensland, Australia: Global Coral Reef Monitoring Network (GCRMN) and Australian Institute of Marine Science.

Altieri, A. H., Harrison, S. B., Seemann, J., Collin, R., Diaz, R. J., & Knowlton, N. (2017). Tropical dead zones and mass mortalities on coral reefs. Proceedings of the National Academy of Sciences, 114(14), 3660-3665.

Alvarado, J.J., j. Cortes, M.F. Esquivel, E. Salas. (2012). Costa Rica's Marine Protected Areas: status and perspectives. Revista de Biología Tropical 60(1): 129-142

Alvarado, L., E.J. Alfaro. (2003). Frecuencia de los ciclones tropicales que afectaron a Costa Rica durante el siglo XX. Tópicos Meteorológicos y Oceanográficos 10: 1-11.

Alvarez-Filip, L. (2025). Arrecifes del mar Caribe: entre el estrés y la muerte. La Jornada Ecológica, 20–22.

Alvarez-Filip, L., González-Barrios, F. J., Pérez-Cervantes, E., Molina-Hernández, A., & Estrada-Saldívar, N. (2022). Stony coral tissue loss disease decimated Caribbean coral populations and reshaped reef functionality. Communications Biology, 5(1), 440.

Alvarez-Filip, L., Estrada-Saldívar, N., Pérez-Cervantes, E., Molina-Hernández, A., & González-Barrios, F. J. (2019). A rapid spread of the Stony Coral Tissue Loss Disease outbreak in the Mexican Caribbean. PeerJ, 7(e8069), 1–17.

Alvarez-Filip, L., Millet-Encalada, M., & Reyes-Bonilla, H. (2009). Impact of Hurricanes Emily and Wilma on the coral community of Cozumel Island, Mexico. Bulletin of Marine Science, 84(3), 295-306.

Alvarez-Filip, L., Dulvy, N. K., Gill, J. A., Côté, I. M., & Watkinson, A. R. (2009). Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. Proceedings of the Royal Society B: Biological Sciences, 276(1669), 3019-3025.

Alves, C., Valdivia, A., Aronson, R. B., Bood, N., Castillo, K. D., Cox, C., ... & Bruno, J. F. (2022). Twenty years of change in benthic communities across the Belizean Barrier Reef. PLoS One, 17(1), e0249155.

Amante, C. and B. W. Eakins, ETOPO1 1 Arc-Minute Global Relief Model: Procedures, Data Sources and Analysis. NOAA Technical Memorandum NESDIS NGDC-24, 19 pp, March 2009. Retrieved from https://developers.google.com/earth-engine/datasets/catalog/NOAA_NGDC_ETOPO1.

Amaya, D. J., Jacox, M. G., Fewings, M. R., Saba, V. S., Stuecker, M. F., Rykaczewski, R. R., ... & Powell, B. S. (2023). Marine heatwaves need clear definitions so coastal communities can adapt. Nature, 616(7955), 29-32.

Amelot, M., Brunel, T., Debrot, D., & Kitson-Walters, K. (2021). Update on the 2012-2020 trends in the St. Eustatius fisheries.

Anderson, R., Morrall, C., Jossart, J., Nimrod, S., Bolda, E., Musser, K., ... & Balza, R. (2014). Marine Protected Area monitoring in the nearshore waters of Grenada, Eastern Caribbean: benthic cover and fish populations. Revista de Biología Tropical, 62, 273-286.

Anderson, R., Morrall, C., Nimrod, S., Balza, R., Berg, C., & Jossart, J. (2012). Benthic and fish population monitoring associated with a marine protected area in the nearshore waters of Grenada, Eastern Caribbean. Revista de Biologia Tropical, 60, 71-87.

Andradi-Brown, D. A., Grey, R., Hendrix, A., Hitchner, D., Hunt, C. L., Gress, E., ... & Exton, D. A. (2017). Depth-dependent effects of culling—do mesophotic lionfish populations undermine current management?. Royal Society Open Science, 4(5), 170027. 10.1098/rsos.170027.

Andrello, M., Darling, E. S., Wenger, A., Suárez-Castro, A. F., Gelfand, S., & Ahmadia, G. N. (2022). A global map of human pressures on tropical coral reefs. Conservation Letters, 15(1), e12858.

Aranda, D. A., Oxenford, H. A., Medina, J., Delgado, G., Díaz, M. E., Samano, C., ... & Bouchon, C. (2022). Widespread microplastic pollution across the Caribbean Sea confirmed using queen conch. Marine Pollution Bulletin, 178, 113582.

Araya-Vargas, A., N. Nova-Bustos. (2017). Health status evaluation of shallow coral reefs in Cahuita and Manzanillo, Costa Rica. Biodiversity and Natural History 43: 48-55.

Arias-González JE, Fung T, Seymour RM, Garza-Pérez JR, Acosta-González G, Bozec Y-M, et al. (2017) A coral-algal phase shift in Mesoamerica not driven by changes in herbivorous fish abundance. PLoS ONE 12(4): e0174855.

Arnold, S. N., Steneck, R. S., & Mumby, P. J. (2010). Running the gauntlet: inhibitory effects of algal turfs on the processes of coral recruitment. Marine Ecology Progress Series, 414, 91-105.

Aronson, R. B., & Precht, W. F. (2006). Conservation, precaution, and Caribbean reefs. Coral reefs, 25(3), 441-450.

Aronson, R. B., & Precht, W. F. (2001). White-band disease and the changing face of Caribbean coral reefs. Hydrobiologia, 460(1), 25-38.

Aronson, R. B., Precht, W. F., Macintyre, I. G., & Murdoch, T. J. (2000). Coral bleach-out in Belize. Nature, 405(6782), 36-36.

Aronson, R. B., Edmunds, P. J., Precht, W. F., Swanson, D. W., & Levitan, D. R. (1994). Large-scale, long-term monitoring of Caribbean coral reefs: simple, quick, inexpensive techniques. Atoll Research Bulletin.

Augustine, S. P., Bailey-Marren, I., Charton, K. T., Kiel, N. G., & Peyton, M. S. (2024). Improper data practices erode the quality of global ecological databases and impede the progress of ecological research. Global Change Biology, 30(1), e17116.

Autoridad Nacional de Asuntos Marítimos (ANAMAR). (2025). Datos de calidad de agua – Zonas costeras de la República Dominicana, 2021–2024. Santo Domingo, R.D.: ANAMAR. [Conjunto de datos abiertos sobre monitoreo costero-marino.]

Babcock, E. A., Tewfik, A., & Burns-Perez, V. (2018). Fish community and single-species indicators provide evidence of unsustainable practices in a multi-gear reef fishery. Fisheries research, 208, 70-85.

Baine, M. (2001). Artificial reefs: a review of their design, application, management and performance. Ocean & Coastal Management, 44(3-4), 241-259.

Baisre, J. A. (2018). An overview of Cuban commercial marine fisheries: the last 80 years. Bulletin of Marine Science, 94(2), 359-375.

Bak, R. P., Nieuwland, G., & Meesters, E. H. (2005). Coral reef crisis in deep and shallow reefs: 30 years of constancy and change in reefs of Curacao and Bonaire. Coral Reefs, 24(3), 475–479.

Bak, R. P. M., & Nieuwland, G. (1995). Long-term change in coral communities along depth gradients over leeward reefs in the Netherlands-Antilles. Bulletin of Marine Science, 56(2), 609–619.

Bak, R. (1987). Effects of chronic oil pollution on a Caribbean coral reef. Marine Pollution Bulletin 18:534-539.

Bak, R. P. M., Carpay, M. J. E., & de Ruyter van Steveninck, E. D. (1984). Densities of the sea urchin Diadema antillarum before and after mass mortalities on the coral reefs of Curacao. Marine Ecology Progress Series, 17, 105–108.

Bak, R. P. M., & Criens, S. R. (1981). Survival after fragmentation of colonies of Madracis mirabilis, Acropora palmata, and A. cervicornis (Scleractinia) and the subsequent impact of a coral disease. Proc. 4th Int. Coral Reef Symp. (Manila) 2, 2(Stoddart 1969), 221–228.

Bak, R.P.M. (1977). Coral Reefs and Their Zonation in Netherlands Antilles. Studies in Geology No. 4 (pp. 96-109). The American Association of Petroleum Geologists.

Bakker, A. C., Gleason, A. C. R., Mantero, A., Dempsey, A. C., Andréfouët, S., Harborne, A. R., & Purkis, S. J. (2023). Heat, human, hydrodynamic, and habitat drivers measured from space correlate with metrics of reef health across the South Pacific. Coral Reefs, 42(1), 219-238.

Baldini, L. M., Baldini, J. U., McElwaine, J. N., Frappier, A. B., Asmerom, Y., Liu, K. B., ... & Breitenbach, S. F. (2016). Persistent northward North Atlantic tropical cyclone track migration over the past five centuries. Scientific reports, 6(1), 37522.

Baldwin, K., Punnett, S., & Smith, C. (2021). Involving communities in the implementation of Reef Check: Strategies for co-management of marine resources in St. Vincent and the Grenadines.

Ballantine, D. L., Appeldoorn, R. S., Yoshioka, P. M., Weil, E., Armstrong, R. A., García, J. R., ... & Bruckner, A. W. (2008). Biology and ecology of Puerto Rican coral reefs. In B. M. Riegl & R. E. Dodge (Eds.), Coral reefs of the USA (pp. 375–406). Springer.

Barrios-Garrido, H., Petit-Rodríguez, M. J., Vera, F., Montiel-Villalobos, M. G., Morán, L., & Wildermann, N. (2016). Riqueza y distribución de corales pétreos en la costa noroccidental del Golfo de Venezuela/Richness and distribution of stony corals in the north-western coast of the Gulf of Venezuela. Ciencia, 24, 27-40.

Bastidas, C., Bone, D., Croquer, A., Debrot, D., Garcia, E., Humanes, A., ... & Rodríguez, S. (2012). Massive hard coral loss after a severe bleaching event in 2010 at Los Roques, Venezuela. Revista de Biología Tropical, 60, 29–37.

Bastidas, C., & Garcia, E. M. (2004). Sublethal effects of mercury and its distribution in the coral Porites astreoides. Marine Ecology Progress Series, 267, 133-143.

Bastidas, C., and E. Garcıa (1999b). Metal content on the reef coral Porites astreoides: an evaluation of river influence and 35 years of chronology. Marine Pollution Bulletin 38, no. 10, 899-907.

Bastidas, C., Bone, D., & Garcia, E. M. (1999a). Sedimentation rates and metal content of sediments in a Venezuelan coral reef. Marine Pollution Bulletin, 38(1), 16-24.

Baumann, J. H., Zhao, L. Z., Stier, A. C., and Bruno, J. F. (2022). Remoteness does not enhance coral reef resilience. Global Change Biology, 28(2), 417-428.

Beeden, R., Maynard, J., Puotinen, M., Marshall, P., Dryden, J., Goldberg, J., & Williams, G. (2015). Impacts and recovery from severe tropical cyclone Yasi on the Great Barrier Reef. PloS one, 10(4), e0121272.

Beets, J. and C. Rogers. (2002). Changes in fishery resources and reef fish assemblages in a Marine Protected Area in the US Virgin Islands: the need for a no take marine reserve. In: M.K. Moosa, ed. Proceedings of the 9th International Coral Reef Symposium, Bali, Indonesia, 23—27 October 2000. Volume 1, p. 449—454.

Beets, J. and Friedlander A. (1999) Evaluation of a conservation strategy: a spawning aggregation closure for red hind, Epinephelus guttatus, in the US Virgin Islands. Environmental Biology of Fishes 55:91–98.

Beets, J. and Friedlander A. (1992) Stock analysis and management strategies for red hind (Epinephelus guttatus) in the US Virgin Islands. Proceedings of the Gulf Caribbean Fisheries Institute 42:66-79.

Begin, C. (2012). Land use and sedimentation impacts on coral reefs in the eastern Caribbean.

Belize Fisheries Department (2023). Stony Coral Tissue Loss Disease (SCTLD). Government of Belize. https://fisheries.gov.bz/tag/sctld/

Belize National Coral Reef Monitoring Network (2024). [Data set]. In Global Coral Reef Monitoring Network (GCRMN), Status of Coral Reefs of the World: 2025. Contributing organizations: Belize Audubon Society, Belize Fisheries Department, Blue Ventures, Coastal Zone Management Authority & Institute, ECOMAR, Fragments of Hope, Healthy Reefs for Healthy People, Hol Chan Marine Reserve, Southern Environmental Association, The Nature Conservancy, Toledo Institute for Development and Environment, Turneffe Atoll Sustainability Association, University of Belize Environmental Research Institute, Wildlife Conservation Society, and World Wildlife Fund [Dataset].

Bell, P.R.F., Tomascik, T. (1994). The demise of the fringing coral reefs of Barbados and in regions in the Great Barrier Reef (GBR) lagoon – impacts of eutrophication. In: Ginsburg RN (ed) Proc Colloquium on Global Aspects of Coral Reefs: Health, Hazards and History. Rosenstiel School of Marine and Atmospheric Science, University of Miami, pp 319–325.

Bennett, C. T., Robertson, A., & Patterson, W. F., III. (2019). First record of the non-indigenous Indo-Pacific damselfish, Neopomacentrus cyanomos (Bleeker, 1856) in the northern Gulf of Mexico. BioInvasions Records, 8(1), 154–166.

Berger, M., Canty, S. W., Tuholske, C., & Halpern, B. S. (2022). Sources and discharge of nitrogen pollution from agriculture and wastewater in the Mesoamerican Reef region. Ocean & Coastal Management, 227, 106269.

Bergseth, B. J., Arias, A., Barnes, M. L., Caldwell, I., Datta, A., Gelcich, S., ... & Cinner, J. E. (2023). Closing the compliance gap in marine protected areas with human behavioural sciences. Fish and Fisheries, 24(4), 695-704.

Berke, P., & Wenger, D. (1991). Montserrat: Emergency planning, response and recovery related to hurricane Hugo. Texas: Hazard Reduction and Recovery Centre and the Texas A&M University.

Beroske, E., and F. Timpen. (2018). The impact of illegal dumpsites on the environment. UAUCU Student Research Exchange Collected Papers 2018:142-168.

Bertelsen, R. D., & Cox, C. (2001). Sanctuary roles in population and reproductive dynamics of Caribbean spiny lobster. Spatial processes and management of marine populations, 591-605.

Bertoncelj, V., Mienis, F., Stocchi, P., & van Sebille, E. (2025). Flow patterns, hotspots, and connectivity of land-derived substances at the sea surface of Curaçao in the southern Caribbean. Ocean Science, 21(3), 945-964.

Bervoets, T. (2010). Working paper on the economic valuation of country St. Maarten's coral reef resources. Nature Foundation St. Maarten.

Besancon, C., & Mudford, T. (2023). Evaluation Framework for Marine Protected AreaManager Networks, OCEAN GOVERNANCE Protecting and Restoring MarineEcosystems, Catalysts for Building Peace and Security and Fostering SustainableEconomies. South Asia and Atlantic Ocean Basin.

Betancourt Fernández, L., & Herrera-Moreno, A. (2019). Identification of ecosystem services in selected pilot marine areas: Punta Cana, Bayahíbe and Samaná (Technical report). GIZ / BVE-ARMB.

Betancur-R, R., Hines, A., Acero P, A., Ortí, G., Wilbur, A. E., & Freshwater, D. W. (2011). Reconstructing the lionfish invasion: insights into Greater Caribbean biogeography. Journal of Biogeography, 38(7), 1281-1293.

Bhagaloo, K., Ali, R., Baboolal, A., & Ward, K. (2023). Powering Clean and Affordable Energy through Geothermal Integration: A Case Study on Dominica. In Computer Aided Chemical Engineering (Vol. 52, pp. 2879-2884). Elsevier.

Bhuiyan, M. M. U., Rahman, M., Naher, S., Shahed, Z. H., Ali, M. M., & Islam, A. R. M. T. (2024). Oxygen declination in the coastal ocean over the twenty-first century: Driving forces, trends, and impacts. Case Studies in Chemical and Environmental Engineering, 9, 100621.

Biecek, P., & Burzykowski, T. (2021). Explanatory model analysis: explore, explain, and examine predictive models. Chapman and Hall/CRC.

Birkart, L. V., & Álvarez-Filip, L. (2025). 2023 global heatwave causes mass mortality of a keystone coral on shallow Western Atlantic reefs. iScience, 28(11).

Black, J. T., Whitmire, S. L., Smith, E. M., Scaroni, A. E., & Van Bloem, S. J. (2025). Sargassum brown tide impacts nearshore water quality and seagrasses in Jobos Bay, Puerto Rico. Biogeochemistry, 168(2), 1-20.

Blakeway, R. D., Fogg, A. Q., Johnston, M. A., Rooker, J. R., & Jones, G. A. (2022). Key life history attributes and removal efforts of invasive lionfish (Pterois volitans) in the Flower Garden Banks National Marine Sanctuary, Northwestern Gulf of Mexico. Frontiers in Marine Science, 9, 774407.

Blanchard, J. L., & Novaglio, C. (2024). Climate change risks to marine ecosystems and fisheries: Projections to 2100 from the Fisheries and Marine Ecosystem Model Intercomparison Project. Food & Agriculture Org.

Blanco-Pimentel, M., Calle-Triviño, J., Barshis, D. J., van der Meij, S. E. T., & Morikawa, M. K. (2025). Building heat-resilient Caribbean reefs: Integrating thermal thresholds and coral colonies selection in restoration. PeerJ, 13, e19987.

Blanco-Pimentel, M., Evensen, N. R., Cortés-Useche, C., Calle-Triviño, J., Barshis, D. J., Galván, V., ... & Morikawa, M. K. (2022). All-inclusive coral reef restoration: How the tourism sector can boost restoration efforts in the caribbean. Frontiers in Marine Science, 9, 931302.

Boaden, A. E., & Kingsford, M. J. (2015). Predators drive community structure in coral reef fish assemblages. Ecosphere, 6(4), 1-33.

Boden, T. A., Marland, G., & Andres, R. J. (2010). Global, Regional, and National Fossil-Fuel CO2 Emissions, 1751-2007 (Version 2010). Environmental System Science Data Infrastructure for a Virtual Ecosystem (ESS-DIVE)(United States); Carbon Dioxide Information Analysis Center (CDIAC), Oak Ridge National Laboratory (ORNL), Oak Ridge, TN (United States).

Bodmer, M. D., Rogers, A. D., Speight, M. R., Lubbock, N., & Exton, D. A. (2015). Using an isolated population boom to explore barriers to recovery in the keystone Caribbean coral reef herbivore Diadema antillarum. Coral Reefs, 34(4), 1011-1021.

Boehmke, B., & Greenwell, B. M. (2019). Hands-on machine learning with R. Chapman and Hall/CRC.

Bohorquez, J. J., Santora, C., Valladares, A. G., van Pul, J., Verde, J., Gamboa, M., ... & Gonzalez, M. J. (2025). A roundtable on marine protected area finance: Lessons from Latin America and the Caribbean on four keys to success for improving financial sustainability. Marine Policy, 182, 106891.

Bon, M., Kayal, M., Dromard, C. R., & Desrosiers, C. (2025). Impact of the 2023 coral bleaching event in Martinique, Eastern Caribbean. Coral Reefs, 1-14.

Borger, J. L., & Steiner, S. C. (2005). The spatial and temporal dynamics of coral diseases in Dominica, West Indies. Bulletin of Marine Science, 77(1), 137-154.

Bosker, T., Guaita, L., & Behrens, P. (2018). Microplastic pollution on Caribbean beaches in the Lesser Antilles. Marine Pollution Bulletin, 133, 442-447.

Boström-Einarsson, L., Babcock, R. C., Bayraktarov, E., Ceccarelli, D., Cook, N., Ferse, S. C., ... & McLeod, I. M. (2020). Coral restoration–A systematic review of current methods, successes, failures and future directions. PloS one, 15(1), e0226631.

Bouchon C, de Lavigne S., Cordonnier S., Bouchon-Navaro Y. (2025). Le phénomène de blanchissement corallien en Guadeloupe de décembre à août 2025. Rapport ÉcoRécif Environnement et Caraïbe Aqua Conseil : 50 pp.

Bouchon C., Bouchon-Navaro Y. (2024). Les communautés récifales de la Baleine du Pain-de-Sucre en 2024. Développement d'un épisode de blanchissement corallien. Rapport EcoRécif Environnement, 28 pp.

Bouchon C., De Lavigne S., Cordonnier S., Bouchon-Navaro Y. (2024b). Le phénomène de blanchissement corallien en Guadeloupe de janvier à août 2024. Rapport ÉcoRécif Environnement et Caraïbe Aqua Conseil : 44 pp.

Bouchon C., De Lavigne S., Cordonnier S., Bouchon-Navaro Y. (2024a). Le phénomène de blanchissement corallien en Guadeloupe en 2023. Rapport ÉcoRécif Environnement et Caraïbe Aqua Conseil : 43 pp.

Bouchon C., Bouchon-Navaro Y. (2023). Evolution des communautés récifales de Saint-Barthélemy. Années 2018 à 2023. Influence de la maladie corallienne SCTLD. Rapport EcoRécif Environnement, 38 pp.

Bouchon C., Bouchon-Navaro Y. (2019). Évolution des communautés récifales de Saint-Barthélemy : années 2002 à 2018. Rapport Université des Antilles, 53 pp.

Bouchon C., Bouchon-Navaro Y. (2017). Maladies et agressions affectant les coraux, gorgones et éponges des Antilles: Impacts sur les communautés récifales. Rapport Université des Antilles – Agence française pour la biodiversité. 71 pp.

Bouchon C., Bouchon-Navaro Y., Louis M., Mazeas F., Marechal J-P., Portillo P., Tregarot E. (2014). French Antilles. Pp 233-237. In: Status and trends of Caribbean Coral Reefs: 1970-2012. Jackson J.B.C., Donovan M.K., Cramer K.L., Lam V. (eds). IUCN publications, 304 pp.

Bouchon, C., Portillo, P., Louis, M., Mazeas, F., and Bouchon-Navaro, Y. (2008). Évolution récente des récifs coralliens des îles de la Guadeloupe et de Saint-Barthélemy. Revue d'écologie, 63(1-2), 45-65.

Bouchon, C., Portillo, P., Bouchon-Navaro, Y., Louis, M., Hoetjes, P., De Meyer, K., ... & Wynne, S. (2008). Status of coral reefs of the Lesser Antilles: the French West Indies, the Netherlands Antilles, Anguilla, Antigua, Grenada, Trinidad and Tobago. In Status of Coral Reefs of the World: 2008

Bouchon C., Portillo P., Bouchon-Navaro Y., Louis M., Hoetjes P., Brathwaite A., Roach R., Oxenford H., O'farrell S., Day O. (2008). Status of coral reefs of the Lesser Antilles after the 2005 coral bleaching event. Pp 85-103. In: Wilkinson C., Souter D., (eds). Status of Caribbean Coral reefs after bleaching and hurricanes in 2005. Global Coral Reef Monitoring Network, and Reef and Rainforest Research Centre, Townsville, 152 pp.

Bouchon C., Bouchon-Navaro Y., Louis M., Portillo P. (2006). Bilan du suivi des communautés récifales de Saint-Barthélemy : années 2002 - 2006. Rapport UAG, 26 pp.

Bouchon C, Miller A, Bouchon-Navaro Y, Portillo P, Louis M. (2004). Status of coral reefs in the French Caribbean islands and other islands of the eastern Antilles. Pp. 493-508 in: Wilkinson C, editor. Status of coral reefs of the world: 2004 Volume 2. Townsville, Queensland, Australia: Global Coral Reef Monitoring Network (GCRMN), 557 pp.

Bouchon C., Bouchon-Navaro Y., Louis M., Portillo P. (2002). Mise en place d'une station de suivi des communautés récifales dans la réserve marine de Saint-Barthélemy : premiers résultats. Rapport UAG, 12 pp.

Bouchon, C., Bouchon-Navaro, Y., Brugneaux, S., & Mazeas, F. (2002). L'état des récifs coralliens dans les Antilles françaises. Rapport IFRECOR, Université des Antilles et de la Guyane.

Bouchon C., Bouchon-Navaro Y. (1998). Rapport préliminaire sur un phénomène de mortalité de poissons et de « blanchissement » de coraux en Martinique. Rapport Université Antilles-Guyane : 12 pp.

Bouchon, C., Bouchon-Navaro Y., Imbert D., Louis M. (1991). Effets de l'ouragan Hugo sur les communautés côtières de Guadeloupe (Antilles françaises). Ann. Inst. océanogr. 67 (1) : 5 - 33.

Bouchon C., Laborel J. (1990). Les peuplements coralliens du Grand Cul-de-Sac Marin de Guadeloupe (Antilles françaises). Annales de l'Institut Océanographique, Paris, 66(1-2) : 19-36.

Bouchon C., Bouchon-Navaro Y., Louis M. (1990). Les biocénoses marines côtières de l'île de Saint Barthélemy. Dossier scientifique pour la création d'une réserve marine : 21 pp.

Bouchon C., Laborel J. (1986). Les peuplements coralliens des côtes de la Martinique. Ann. Inst. Oceanogr., Paris, 62(2) : 199-237.

Bouchon C.,Bouchon-Navaro Y., Laborel J., Laborel-Deguen F., Lamy D., Louis M., Philippot V., Vacelet J. (1986). Étude des biocénoses marines côtières des îles de Saint-Barthélemy, Saint-Martin et Anguilla. Rapport préliminaire de la mission "Ecorécif". Rapport Université des Antilles et de la Guyane, 21 pp.

Bove, C. B., Mudge, L., & Bruno, J. F. (2022). A century of warming on Caribbean reefs. PloS Climate, 1(3), e0000002.

Bove, C. B., Ries, J. B., Davies, S. W., Westfield, I. T., Umbanhowar, J., & Castillo, K. D. (2019). Common Caribbean corals exhibit highly variable responses to future acidification and warming. Proceedings of the Royal Society B, 286(1900), 20182840.

Box, S., Canty, S. (2010). The long and short term economic drivers of overexploitation in honduran coral reef fisheries due to their dependence on export markets. In: Proceedings of the 63rd Gulf and Caribbean Fisheries Institute 63. pp. 43–51.

Brandt, M. E., Ennis, R. S., Meiling, S. S., Townsend, J., Cobleigh, K., Glahn, A., ... & Smith, T. B. (2021). The emergence and initial impact of stony coral tissue loss disease (SCTLD) in the United States Virgin Islands. Frontiers in Marine Science, 8, 715329.

Brandt, M. E., & McManus, J. W. (2009). Disease incidence is related to bleaching extent in reef-building corals. Ecology, 90(10), 2859-2867.

Bravo-Calderon, A., Saenz-Arroyo, A., Fulton, S., Espinoza-Tenorio, A., & Sosa-Cordero, E. (2021). Goliath grouper Epinephelus itajara oral history, use, and conservation status in the Mexican Caribbean and Campeche Bank. Endangered Species Research, 45, 283-300.

Bries, J. M., Debrot, A. O., & Meyer, D. L. (2004). Damage to the leeward reefs of Curacao and Bonaire, Netherlands Antilles from a rare storm event: Hurricane Lenny, November 1999. Coral Reefs, 23(2), 297-307.

Briones-Fourzán, P., Álvarez-Filip, L., Barradas-Ortíz, C., Morillo-Velarde, P. S., Negrete-Soto, F., Segura-García, I., ... & Lozano-Álvarez, E. (2019). Coral reef degradation differentially alters feeding ecology of co-occurring congeneric spiny lobsters. Frontiers in Marine Science, 5, 516.

Bruckner, A. W. (2012). Factors contributing to the regional decline of Montastraea annularis (complex). In Proceedings of the 12th International Coral Reef Symposium, Cairns, Australia, July (pp. 9-13).

Bruckner, A., & Williams, A. (2012). Assessment of the community structure, status, health and resilience of coral reefs off St. Kitts and Nevis, Landover MD.

Bruckner, A. (2010). Cayman Islands coral reef health and resilience assessments. Khaled bin Sultan Living Oceans Foundation, Landover, MD, 58.

Brunel, M. A. T., Debrot, D., & Kitson-Walters, K. (2021). Update on the 2012-2020 trends in the St. Eustatius fisheries.

Bruno, J. F., & Selig, E. R. (2007). Regional decline of coral cover in the Indo-Pacific: timing, extent, and subregional comparisons. PLoS one, 2(8), e711.

Budd, A. F., Fukami, H., Smith, N. D., & Knowlton, N. (2012). Taxonomic classification of the reef coral family Mussidae (Cnidaria: Anthozoa: Scleractinia). Zoological Journal of the Linnean Society, 166(3), 465-529.

Buglass, S., Donner, S. D., & Alemu, J. B. (2016). A study on the recovery of Tobago's coral reefs following the 2010 mass bleaching event. Marine Pollution Bulletin, 104(1-2), 198-206.

Burke, L., Greenhalgh, S., Prager, D., & Cooper, E. (2008). Coastal capital: economic valuation of coral reefs in Tobago and St. Lucia.

Burke, L., & Maidens, J. (2004). Reefs at risk in the Caribbean. World Resources Institute. https://files.wri.org/d8/s3fs-public/pdf/reefs_caribbean_full.pdf

Bustamante, G., Vanzella, A., Glazer, R., & Collado-Vides, L. (2018). The evolution of the Caribbean marine protected area management network and forum (CaMPAM): 20 years of the regional multidimensional program for strengthening MPA practitioners. Gulf and Caribbean Research, 29(1), GCFI1-GCF9.

Bustamante, G., Vanzella-Khouri, A., & Glazer, R. (2014). Marine Protected Areas Management in the Caribbean and Mediterranean Seas: Making Them More than Paper Parks. UNEP-CAR/RCU, GEF. https://gefcrew.org/carrcu/SPAWSTAC9/Info-Docs/WG.42-INF.8-Rev.1-en.pdf

Butler IV, M. J., Duran, A., Feehan, C. J., Harborne, A. R., Hykema, A., Patterson, J. T., ... & Williams, S. M. (2024). Restoration of herbivory on Caribbean coral reefs: are fishes, urchins, or crabs the solution?. Frontiers in Marine Science, 11, 1329028.

Butler, C. B., Butler, J., Ross, E. P., & Matthews, T. R. (2022). Seasonal and fishery impacts on the nutritional condition of the Caribbean spiny lobster Panulirus argus in Florida, USA. Marine Ecology Progress Series, 696, 43-56.

Bégin, C., Schelten, C. K., Nugues, M. M., Hawkins, J., Roberts, C., & Côté, I. M. (2016). Effects of protection and sediment stress on coral reefs in Saint Lucia. PLoS One, 11(2), e0146855.

Bégin, C., Brooks, G., Larson, R. A., Dragićević, S., Scharrón, C. E. R., & Côté, I. M. (2014). Increased sediment loads over coral reefs in Saint Lucia in relation to land use change in contributing watersheds. Ocean & Coastal Management, 95, 35-45.

Caballero Aragón, H., Armenteros, M., Perera Valderrama, S., Rey Villiers, N., Cobián Rojas, D., Campos Verdecia, K., & Alcolado Menéndez, P. M. (2019). Ecological condition of coral reef assemblages in the Cuban Archipelago. Marine Biology Research, 15(1), 61-73.

Cabanillas-Terán, N., Pérez-Posada, I., Rosas-Luis, R., Rodríguez-Martínez, R.E., Lizcano, D.J., Aragón-Moreno, A.A. (2025). Overlapping niches among sea urchin species reveal Sargassum-induced degradation at Caribbean shallow coral reefs. Science of The Total Environment, 999, 180297.

Caldeira, K., & Wickett, M. E. (2003). Anthropogenic carbon and ocean pH. Nature, 425(6956), 365-365.

California State University Northridge, & Edmunds, P. (2022). Virgin Islands National Park: Coral Reef: Population dynamics: Landscape-scale variation in scleractinian corals (ver. 1) [Dataset]. Environmental Data Initiative. https://doi.org/10.6073/pasta/c69935f8d4909969082381bc7c45a759 (Accessed 2023-02-08).

Calle-Triviño, J., Rojas-Cano, D., Niño-Torres, L. A., Colín-García, N., Hernández-Landa, R. C., Blanco-Pimentel, M., ... & Rioja-Nieto, R. (2025). Rebuilding Coral Reefs: How Tourism Can Be a Driver Behind Solutions in a Changing Ocean. Diversity, 17(4), 268.

Callwood, K. A. (2021). Examining the development of a parrotfish fishery in The Bahamas: Social considerations & management implications. Global Ecology and Conservation, 28, e01677.

Camacho, R., Steele, S., Challenger, S., & Archibald, M. (2020). Status of coral reefs in Antigua & Barbuda: using data to inform management. PeerJ, 8, e9236.

Camacho, R. A., & Steneck, R. S. (2017). Creating a TURF from the bottom-up: Antigua's community-based coral reef no-take reserve. Bulletin of Marine Science, 93(1), 217-232.

Cambers, G. (2018). Caribbean Islands, coastal ecology and geomorphology. In Encyclopedia of coastal science (pp. 1-8). Springer, Cham.

Cambers, G., & Wynne, S. P. (2019). Beach changes and associated ecosystem services in Anguilla, West Indies, 1992–2014. Aquatic Ecosystem Health & Management, 22(2), 193-204.

Cameron, C., & Roopnarine, P. D. (2022). The impacts of the invasive lionfish on Caribbean food webs. ARPHA Preprints, 3, e97321.

Camp, J., Scaife, A. A., & Heming, J. (2018). Predictability of the 2017 North Atlantic hurricane season. Atmospheric Science Letters, 19(5), e813.

Cano, I., Sellares-Blasco, R. I., Lefcheck, J. S., Villalpando, M. F., & Croquer, A. (2021). Effects of herbivory by the urchin Diadema antillarum on early restoration success of the coral Acropora cervicornis in the central Caribbean. Journal of Experimental Marine Biology and Ecology, 539, 151541.

Cardoso, G. O., Kersting, D. K., Brachert, T. C., Heiss, G. A., Leinfelder, R., Maréchal, J. P., & D'Olivo, J. P. (2025). Emerging skeletal growth responses of Siderastrea siderea corals to multidecadal anthropogenic impacts in Martinique, Caribbean Sea. Scientific Reports, 15(1), 23127.

Carlot, J., Vousdoukas, M., Rovere, A., Karambas, T., Lenihan, H. S., Kayal, M., ... & Parravicini, V. (2023). Coral reef structural complexity loss exposes coastlines to waves. Scientific Reports, 13(1), 1683.

Carlson, R. R., Evans, L. J., Foo, S. A., Grady, B. W., Li, J., Seeley, M., ... & Asner, G. P. (2021). Synergistic benefits of conserving land-sea ecosystems. Global Ecology and Conservation, 28, e01684.

Carlson, R. R., Foo, S. A., & Asner, G. P. (2019). Land use impacts on coral reef health: A ridge-to-reef perspective. Frontiers in Marine Science, 6, 562.

Carpenter, R. C. (1990b). Mass mortality of Diadema antillarum: II. Effects on population densities and grazing intensity of parrotfishes and surgeonfishes. Marine Biology, 104(1), 79-86.

Carpenter, R. C. (1990a). Mass mortality of Diadema antillarum: I. Long-term effects on sea urchin population-dynamics and coral reef algal communities. Marine Biology, 104(1), 67-77.

Carpenter, R. C. (1986). Partitioning herbivory and its effects on coral reef algal communities. Ecological monographs, 56(4), 345-364.

Carr, L. M., & Heyman, W. D. (2009). Jamaica bound? Marine resources and management at a crossroads in Antigua and Barbuda. Geographical Journal, 175(1), 17-38.

Carrasco Navas-Parejo, J. C., Montalván, B. C., & Guity, J. (2024). Contaminación por microplásticos en arrecifes de alto valor ecológico y en estado crítico de extinción (Acropora palmata) en el Parque Nacional Jeannette Kawas. [Technical report UICN]. 28 p

Carrasco Navas-Parejo, J., Paspapyrou, S., Haro, S., De Frutos, I., & Corzo, A. (2023). Trophic status of a coastal lagoon - marine harbor system: Potential outwelling rates to the Mesoamerican Barrier Reef southern region. The Science of the total Environment, 163202.

Carrigan, A. D., & Puotinen, M. (2014). Tropical cyclone cooling combats region-wide coral bleaching. Global Change Biology, 20(5), 1604-1613.

Castillo Jáquez, J. del C., Gómez Santana, R. L., & Rodríguez-Núñez, V. A. (2022). Evaluation of the quality of the coastal waters of Playa Dorada (Dominican Republic). IOSR Journal of Environmental Science, Toxicology and Food Technology, 16(2), 23–30.

Cavada-Blanco, F., Cappelletto, J., Agudo-Adriani, E., Martinez, S., Rodriguez, J. P., & Cróquer, A. (2020). Status of the pillar coral Dendrogyra cylindrus in Los Roques National Park, Southern Caribbean. bioRxiv, 2020-09.

Caviedes V, Arenas-Granados P, Carrasco JCNP (2014). Una contribución a la política pública para el manejo costero integrado de Honduras: análisis diagnóstico. Rev Gestão Costeira Integr 14:645–662. doi: 10.5894/rgci461

Cayemitte, K., Aoki, N., Ferguson, S. R., Mooney, T. A., & Apprill, A. (2023). Ramicrusta invasive alga causes mortality in Caribbean coral larvae. Frontiers in Marine Science, 10, 1158947.

CCO (2024). Boletín Expedición Científica Seaflower 2021-II Isla Cayos de Bajo Nuevo. Bogotá D.C.: Puntoaparte Editores.

Center for International Earth Science Information Network (CIESIN) - Columbia University (2018). Gridded Population of the World, Version 4 (GPWv4): Population Count, Revision 11. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). https://doi.org/10.7927/H4JW8BX5. Retrieved from https://developers.google.com/earth-engine/datasets/catalog/CIESIN_GPWv411_GPW_Population_Count.

Central Intelligence Agency (CIA) (2025). The World Factbook – Belize. Accessed on 08/06/2025 at https://www.cia.gov/the-world-factbook/countries/belize/

Cetina-Heredia, P., & Allende-Arandía, M. E. (2023). Caribbean Marine Heatwaves, Marine Cold Spells, and Co-Occurrence of Bleaching Events. Journal of Geophysical Research: Oceans, 128(10), e2023JC020147.

Chalifour J. (2018). Reef Check Saint-Martin : Réseau de suivi de l'état de santé des stations coralliennes : année 2017 ; Campagne post-Irma, 20 pages.

Chalifour J. (2017). Reef Check Saint-Martin : Réseau de suivi de l'état de santé des stations coralliennes : année 2017, 20 pages.

Chamberlain, S. A., & Szöcs, E. (2013). taxize: taxonomic search and retrieval in R. F1000Research, 2, 191.

Charles, D. (2022). Addressing Climate Change and Waste Management Challenges Through the Development of the Waste-to-Energy Value Chain for Trinidad and Tobago. In: Walker, T., Wendt, S., Goubran, S., Schwartz, T. (eds) Business and Policy Solutions to Climate Change. Palgrave Studies in Sustainable Business In Association with Future Earth. Palgrave Macmillan, Cham.

Chaves-Fonnegra A, Panassiti B, Smith TB, Brown E, Clemens E, Sevier M, Brandt ME (2021). Seawater turbidity and temperature as drivers of white plague disease on Caribbean coral reefs. Ecography 44:1071-1085

Chaves-Fonnegra, A., Panassiti, B., Smith, T. B., Brown, E., Clemens, E., Sevier, M., & Brandt, M. E. (2021). Environmental and biological drivers of white plague disease on shallow and mesophotic coral reefs. Ecography, 44(7), 1071-1085.

Cheal, A. J., MacNeil, M. A., Emslie, M. J., & Sweatman, H. (2017). The threat to coral reefs from more intense cyclones under climate change. Global change biology, 23(4), 1511-1524.

Chen, T., & Guestrin, C. (2016). Xgboost: A scalable tree boosting system. In Proceedings of the 22nd acm sigkdd international conference on knowledge discovery and data mining (pp. 785-794).

Chevalier Monteagudo, P. P. (2017). Efecto de las poblaciones de Pterois volitans (Pisces: Scorpaenidae) sobre sus principales presas y competidores entre los peces de arrecifes en varias localidades en Cuba. Ph.D. thesis, Acuario Nacional De Cuba, La Habana,Cuba, 117.

Chollett, I., & Mumby, P. J. (2013). Reefs of last resort: Locating and assessing thermal refugia in the wider Caribbean. Biological conservation, 167, 179-186.

Chollett, I., Mumby, P. J., Müller-Karger, F. E., & Hu, C. (2012). Physical environments of the Caribbean Sea. Limnology and Oceanography, 57(4), 1233-1244.

Chollett, I., Müller-Karger, F. E., Heron, S. F., Skirving, W., & Mumby, P. J. (2012). Seasonal and spatial heterogeneity of recent sea surface temperature trends in the Caribbean Sea and southeast Gulf of Mexico. Marine pollution bulletin, 64(5), 956-965.

Christie, M., Remoundou, K., Siwicka, E., & Wainwright, W. (2015). Valuing marine and coastal ecosystem service benefits: Case study of St Vincent and the Grenadines' proposed marine protected areas. Ecosystem services, 11, 115-127.

Christie, P., White, A., & Deguit, E. (2003). Starting point or solution? Community-based marine protected areas in the Philippines. Marine Policy, 27(2), 117–130.

Chávez, E. A., Tunnell, J. W., Jr., & Withers, K. (2010). Zonación y ecología de los arrecifes: Plataforma Veracruzana y Banco de Campeche. En J. W. Tunnell Jr., E. A. Chávez, & K. Withers (Eds.), Arrecifes coralinos del sur del Golfo de México (pp. 60–100). Instituto Politécnico Nacional.

Chávez, E. A. (2009). Potential production of the Caribbean spiny lobster (Decapoda, Palinura) fisheries. Crustaceana, 1393-1412.

Chérubin, L. M., & Richardson, P. L. (2007). Caribbean current variability and the influence of the Amazon and Orinoco freshwater plumes. Deep Sea Research Part I: Oceanographic Research Papers, 54(9), 1451-1473.

Cinner, J. E., Maire, E., Huchery, C., MacNeil, M. A., Graham, N. A., Mora, C., ... & Mouillot, D. (2018). Gravity of human impacts mediates coral reef conservation gains. Proceedings of the National Academy of Sciences, 115(27), E6116-E6125.

Cissell, E. C., Eckrich, C. E., & McCoy, S. J. (2022). Cyanobacterial mats as benthic reservoirs and vectors for coral black band disease pathogens. Ecological Applications, 32(6), e2692.

Civallero E., Seux B , Chapron S. (2021). Étude de l'impact environnemental des sites d'épandage de sargasses en Guadeloupe. Rapport final. BRGM/RP-70520-FR, 115 p.

Clark, K. E., Bravo, V. D., Giddings, S. N., Davis, K. A., Pawlak, G., Torres, M. A., ... & Collin, R. (2022). Land use and land cover shape river water quality at a continental Caribbean land-ocean interface. Frontiers in Water, 4, 737920.

Clementi, G. M., Babcock, E. A., Valentin-Albanese, J., Bond, M. E., Flowers, K. I., Heithaus, M. R., ... & Chapman, D. D. (2021). Anthropogenic pressures on reef-associated sharks in jurisdictions with and without directed shark fishing. Marine Ecology Progress Series, 661, 175-186.

Coad, L., Leverington, F., Knights, K., Geldmann, J., Eassom, A., Zamora, C., ... Kapos, V. (2015). Measuring impact of protected area management interventions: current and future use of the Global Database of Protected Area Management Effectiveness. Philosophical Transactions of the Royal Society B: Biological Sciences, 370(1681), 20140281.

Cobián-Rojas, D., Chevalier-Monteagudo, P. P., Albertus Estévez, M., Domínguez Fong, X., Camejo Vergara, A., Márquez Govea, L., ... & Croquer, A. (2024). Massive spawning of bleached Caribbean corals in western Cuba during a severe heat stress period. Bulletin of Marine Science, 100(4), 777-778.

Cobián-Rojas, D., Schmitter-Soto, J. J., Betancourt, C. M. A., Aguilar-Perera, A., Ruiz-Zárate, M. Á., González-Sansón, G., ... & Valderrama, S. P. (2018). The community diversity of two Caribbean MPAs invaded by lionfish does not support the biotic resistance hypothesis. Journal of Sea Research, 134, 26-33.

Coelho, V. R., & Manfrino, C. (2007). Coral community decline at a remote Caribbean island: marine no-take reserves are not enough. Aquatic Conservation: Marine and Freshwater Ecosystems, 17(7), 666-685.

Coggins Jr, A. O. (2014). The globalization of the cruise industry: a tale of ships. Worldwide Hospitality and Tourism Themes, 6(2), 138-151.

Cohen-Sánchez, A., Sánchez-Mairata, A. G., Valencia, J. M., Box, A., Pinya, S., Tejada, S., & Sureda, A. (2023). Immune and oxidative stress response of the fish Xyrichthys novacula infected with the trematode ectoparasite Scaphanocephalus sp. in the Balearic Islands. Fishes, 8(12), 600.

Colella, M. A., Ruzicka, R. R., Kidney, J. A., Morrison, J. M., & Brinkhuis, V. B. (2012). Cold-water event of January 2010 results in catastrophic benthic mortality on patch reefs in the Florida Keys. Coral reefs, 31(2), 621-632.

Connell, S. D., Foster, M. S., & Airoldi, L. (2014). What are algal turfs? Towards a better description of turfs. Marine Ecology Progress Series, 495, 299-307.

Connell, J. H., Hughes, T. P., & Wallace, C. C. (1997). A 30-year study of coral abundance, recruitment, and disturbance at several scales in space and time. Ecological Monographs, 67(4), 461-488.

Consejo Dominicano de Pesca y Acuicultura. (2022). Calendario de veda de especies acuáticas de la República Dominicana. CODOPESCA.

Contreras-Silva, A.I., Tilstra, A., Migani, V. et al. (2020) A meta-analysis to assess long-term spatiotemporal changes of benthic coral and macroalgae cover in the Mexican Caribbean. Sci Rep 10, 8897.

Cook, C. A., Sikkel, P. C., Renoux, L. P., & Smit, N. J. (2015). Blood parasite biodiversity of reef-associated fishes of the eastern Caribbean. Marine Ecology Progress Series, 533, 1-13.

Cooper, E., Burke, L., & Bood, N. (2009). Belize's Coastal Capital: The Economic Contribution of Belize's Coral Reefs and Mangroves. Working Paper. World Resources Institute, Washington DC. 53 pp. Accessed on 08/06/2025 at http://www.wri.org/publications.

CORALINA-INVEMAR (2012). Gómez, López, D. I., C. Segura-Quintero, P. C. Sierra-Correa y J. Garay-Tinoco (Eds). Atlas de la Reserva de Biósfera Seaflower. Archipiélago de San Andrés, Providencia y Santa Catalina. Instituto de Investigaciones Marinas y Costeras "José Benito Vives De Andréis" -INVEMAR- y Corporación para el Desarrollo Sostenible del Archipiélago de San Andrés, Providencia y Santa Catalina -CORALINA-. Serie de Publicaciones Especiales de INVEMAR # 28. Santa Marta, Colombia 180 p.

Cornwall, C. E., Carlot, J., Branson, O., Courtney, T. A., Harvey, B. P., Perry, C. T., ... & Comeau, S. (2023). Crustose coralline algae can contribute more than corals to coral reef carbonate production. Communications Earth & Environment, 4(1), 105.

Cornwall, C. E., Comeau, S., Kornder, N. A., Perry, C. T., van Hooidonk, R., DeCarlo, T. M., ... & Lowe, R. J. (2021). Global declines in coral reef calcium carbonate production under ocean acidification and warming. Proceedings of the National Academy of Sciences, 118(21), e2015265118.

Corporación para el Desarrollo Sostenible del Archipiélago de San Andrés, Providencia y Santa Catalina (2020). Operación Cangrejo Negro arroja resultados preliminares de zonas sumergidas.

Correa, D., Villegas-Sánchez, C. A., Pérez-España, H., & Alvarez-Filip, L. (2025). Reef health status of the southwestern Gulf of Mexico and Mexican Caribbean coral reefs. Ciencias Marinas, 50(1B), e3501.

Cortés, A., Casillas-Hernández, R., Cambeses-Franco, C., Bórquez-López, R., Magallón-Barajas, F., Quadros-Seiffert, W., ... & Moreira, M. T. (2021). Eco-efficiency assessment of shrimp aquaculture production in Mexico. Aquaculture, 544, 737145.

Cortés, J., H.A. Oxenford, B.I. van Tussenbroek, E. Jordán-Dahlgren, A. Cróquer, C. Bastidas, J.C. Ogden. (2019). The CARICOMP network of Caribbean Marine Laboratories (1985–2007): History, key findings, and lessons learned. Frontiers in Marine Science, 5:519.

Cortés, J. (2016). The Caribbean coastal and marine ecosystems, pp. 591-617. In: M. Kappelle (Ed.). Costa Rican Ecosystems. University of Chicago Press, Chicago and London.

Cortés, J., A.C. Fonseca, J. Nivia, V. Nielsen-Muñoz, J. Samper-Villareal, E. Salas, S. Martínez P. Zamora-Trejos. (2010). Monitoring coral reefs, seagrasses and mangroves in Costa Rica (CARICOMP). Revista de Biología Tropical, 58 (Supplement 3): 1-22.

Cortés, J., C. Jiménez. (2003). Past, present and future of the coral reefs of the Caribbean coast of Costa Rica: p: 223-239. In: J. Cortés (Ed.), Latin American Coral Reefs. Elsevier Science B.V., Amsterdam.

Cortés, J., M.J. Risk. (1985). A reef under siltation stress: Cahuita, Costa Rica. Bulletin of Marine Science 36: 339-356.

Cortés, J., M.M. Murillo, H.M. Guzmán, J. Acuña. (1984). Pérdida de zooxantelas y muerte de corales y otros organismos arrecifales en el Caribe y Pacífico de Costa Rica. Revista de Biología Tropical 32: 227–232.

Cortés-Useche, C., Muñiz-Castillo, A. I., Calle-Triviño, J., Yathiraj, R., & Arias-González, J. E. (2019). Reef condition and protection of coral diversity and evolutionary history in the marine protected areas of Southeastern Dominican Republic. Regional Studies in Marine Science, 32, 100893.

Costello, K. E., Lynch, S. A., McAllen, R., O'Riordan, R. M., & Culloty, S. C. (2022). Assessing the potential for invasive species introductions and secondary spread using vessel movements in maritime ports. Marine Pollution Bulletin, 177, 113496.

Courtene-Jones, W., Maddalene, T., James, M. K., Smith, N. S., Youngblood, K., Jambeck, J. R., ... & Thompson, R. C. (2021). Source, sea and sink—A holistic approach to understanding plastic pollution in the Southern Caribbean. Science of the Total Environment, 797, 149098.

Coussens, M., Cassidy, M., Watt, S. F. L., Jutzeler, M., Talling, P. J., Barfod, D., ... Palmer, M. R. (2017). Long-term changes in explosive and effusive behaviour at andesitic arc volcanoes: Chronostratigraphy of the Centre Hills Volcano, Montserrat. Journal of Volcanology and Geothermal Research, 333–334, 15–35.

Crabbe, M. James C. (2010). Coral Ecosystem Resilience, Conservation and Management on the Reefs of Jamaica in the Face of Anthropogenic Activities and Climate Change. Diversity 2 (6): 881-896.

Cramer, K. L., Donovan, M. K., Jackson, J. B. C., Greenstein, B. J., Korpanty, C. A., Cook, G. M., & Pandolfi, J. M. (2021). The transformation of Caribbean coral communities since humans. Ecology and Evolution, 11, 10098–10118.

Cramer, K. L., Jackson, J. B., Donovan, M. K., Greenstein, B. J., Korpanty, C. A., Cook, G. M., & Pandolfi, J. M. (2020). Widespread loss of Caribbean acroporid corals was underway before coral bleaching and disease outbreaks. Science Advances, 6(17), eaax9395.

Cramer, K. L., O'Dea, A., Carpenter, C., & Norris, R. D. (2018). A 3000 year record of Caribbean reef urchin communities reveals causes and consequences of long-term decline in Diadema antillarum. Ecography, 41(1), 164-173.

Creary, M. (2010). Coral reef monitoring for the organisation of eastern Caribbean states and Tobago. Belize: Caribbean Community Climate Change Centre.

Creary, M., Alcolado, P., Vania Coelho, J. A., Crabbe, S. G., Geraldes, F., Henry, A., & Hibbert, M. (2008). 1 7. Status of Coral Reefs INTHE Northern Caribbean and Western Atlantic GCRMN Node IN 2008. Status of coral reefs of the world: 2008, 239.

Croquer, A., Zambrano, S., Evangelista-Perez, D. Y., King, S., Guendulain-Garcia, S., Villalpando, M. F., & Sellares-Blasco, R. I. (2024). Short-term temporal trends of a coral reef in Samana (Dominican Republic): the value of a permanent monitoring program to identify drivers of rapid change.

Croquer, A., Zambrano, S., Irazabal, I., & Torres, R. E. (2022). Factores globales y locales que inciden sobre la degradación de los arrecifes coralinos: una revisión para la República Dominicana.

Croquer, A., Zambrano, S., King, S., Reyes, A., Sellares Blasco, R. I., Valdez Trinidad, A., ... & Miyazawa, E. (2022). Stony Coral Tissue Loss Disease and other diseases affect adults and recruits of major reef builders at different spatial scales in the Dominican Republic. Gulf and Caribbean Research, 33(1), GCFI1-GCFI13.

Cróquer, A., Weil, E., & Rogers, C. S. (2021). Similarities and differences between two deadly Caribbean coral diseases: white plague and stony coral tissue loss disease. Frontiers in Marine Science, 8, 709544.

Cróquer, A., Cavada-Blanco, F., Zubillaga, A. L., Agudo-Adriani, E. A., & Sweet, M. (2016). Is Acropora palmata recovering? a case study in Los Roques National Park, Venezuela. PeerJ, 4, e1539.

Cróquer, A., Debrot, D., Klein, E., Kurten, M., Rodríguez, S., & Bastidas, C. (2010). What can two years of monitoring tell us about Venezuelan coral reefs? The Southern Tropical America node of the Global Coral Reef Monitoring Network (STA-GCRMN). Revista de biologia tropical, 58, 51-65.

Cróquer, A., & Weil, E. (2009). Spatial variability in distribution and prevalence of Caribbean scleractinian coral and octocoral diseases. II. Genera-level analysis. Diseases of aquatic organisms, 83(3), 209-222.

Cróquer, A., Bastidas, C., & Lipscomb, D. (2006a). Folliculinid ciliates: a new threat to Caribbean corals?. Diseases of Aquatic Organisms, 69(1), 75-78.

Cróquer, A., Bastidas, C., Lipscomp, D., Rodríguez-Martínez, R. E., Jordan-Dahlgren, E., & Guzman, H. M. (2006b). First report of folliculinid ciliates affecting Caribbean scleractinian corals. Coral Reefs, 25(2), 187-191.

Cróquer, A., Weil, E., Zubillaga, A. L., & Pauls, S. M. (2005). Impact of a white plague-II outbreak on a coral reef in the archipelago Los Roques National Park, Venezuela.

Cróquer, A., & Bone, D. (2003b). Las enfermedades en corales escleractínidos ¿Un nuevo problema en el arrecife de Cayo Sombrero Parque Nacional Morrocoy, Venezuela? Revista de Biología Tropical, 51(S4), 167-172.

Cróquer, A., Pauls, S. M., & Zubillaga, A. L. (2003a). White plague disease outbreak in a coral reef at Los Roques National Park, Venezuela. Revista de biologia tropical, 51(4), 39-45.

Cruz-Marrero, W., Stevens, B. G., & Lipcius, R. N. (2024). Newly discovered deep-water nursery and spawning habitats of the queen conch Aliger gigas in Puerto Rico. Marine Ecology Progress Series, 750, 65-76.

Créocéan (2022). Étude de la SCTLD en Martinique - Caractérisation de dix peuplements coralliens - janvier 2022 - T+10 mois. Rapport pour DEAL 972. 31 pp.

Cáceres, C., Kiszka, J. J., Luna-Acosta, A., Herrera, H., Zarza, E., & Heithaus, M. R. (2022). Predatory fish exploitation and relative abundance in a data-poor region from the Caribbean coast of Colombia, inferred from artisanal fishery interview surveys and baited remote underwater video systems. Aquatic Conservation: Marine and Freshwater Ecosystems, 32(9), 1401-1415.

Côté, I. M., Green, S. J., & Hixon, M. A. (2013). Predatory fish invaders: Insights from Indo-Pacific lionfish in the western Atlantic and Caribbean. Biological Conservation, 164, 50-61.

D'Croz, L., Del Rosario, J. B., & Gondola, P. (2005). The effect of fresh water runoff on the distribution of dissolved inorganic nutrients and plankton in the Bocas del Toro Archipelago, Caribbean Panama. Caribbean Journal of Science, 41(3), 414–429.

Dahlgren, C., Pizarro, V., Sherman, K., Greene, W., & Oliver, J. (2021). Spatial and temporal patterns of stony coral tissue loss disease outbreaks in the Bahamas. Frontiers in Marine Science, 8, 682114.

De Bakker, D. M., van Duyl, F. C., Perry, C. T., & Meesters, E. H. (2019). Extreme spatial heterogeneity in carbonate accretion potential on a Caribbean fringing reef linked to local human disturbance gradients. Global change biology, 25(12), 4092-4104.

De Bakker, D. M., Van Duyl, F. C., Bak, R. P., Nugues, M. M., Nieuwland, G., & Meesters, E. H. (2017). 40 Years of benthic community change on the Caribbean reefs of Curaçao and Bonaire: the rise of slimy cyanobacterial mats. Coral Reefs, 36(2), 355-367.

de Bettencourt, J., & Imminga-Berends, H. (2015). Overseas Countries and Territories: Environmental Profiles.

de Breuyn, M., van der Last, A. J., Klokman, O. J., & Hylkema, A. (2023). Diurnal predators of restocked lab-reared and wild Diadema antillarum near artificial reefs in Saba. PeerJ, 11, e16189.

de Graaf, M., Piontek, S., Miller, D. C. M., Brunel, T., & Nagelkerke, L. A. J. (2015). Status and trends of St. Eustatius coral reef ecosystem and fisheries: 2015 report card (No. C167/15). IMARES Wageningen UR.

De Kool, D. M., Eckrich, C. E., Francisca, R. L. (2025). Addendum: Coral Bleaching in the Bonaire National Marine Park in 2024. STINAPA Bonaire. https://www.dcbd.nl/resource/4045

de la Guardia, E., Giménez-Hurtado, E., Defeo, O., Angulo-Valdes, J., Hernández-González, Z., Espinosa–Pantoja, L., ... & Arias-González, J. E. (2018). Indicators of overfishing of snapper (Lutjanidae) populations on the southwest shelf of Cuba. Ocean & Coastal Management, 153, 116-123.

de la Guardia, E. (2000). Evento de blanqueamiento en un arrecife de coral en la costa de La Habana, Cuba. Rev. Invest. Mar., 21(1-3): 108-109.

de Leijer, D. C. A., Dogruer, G., & Meesters, H. W. G. (2023). Assessment of anthropogenic chemical pollutants in Bonaire's coastal ecosystems by sediment and PDMS sampling.

de León, R., Vane, K., Bertuol, P., Chamberland, V. C., Simal, F., Imms, E., & Vermeij, M. J. (2013). Effectiveness of lionfish removal efforts in the southern Caribbean. Endangered Species Research, 22(2), 175-182.

de Wit, C. D., & Johnson, P. T. (2024). Black Spot Syndrome in ocean surgeonfish: using video-based surveillance to quantify disease severity and test environmental drivers. Marine Biology, 171(5), 110.

De'Ath, G., Fabricius, K. E., Sweatman, H., & Puotinen, M. (2012). The 27–year decline of coral cover on the Great Barrier Reef and its causes. Proceedings of the National Academy of Sciences, 109(44), 17995-17999.

Debrot, A. O., Houtepen, E., Meesters, H. W. G., van Beek, I. J. M., Timmer, T., Boman, E., ... & Ballantine, D. L. (2014). Habitat diversity and biodiversity of the benthic seascapes of St. Eustatius (No. C078/14). IMARES.

Del Río, L., Navarro-Martínez, Z. M., Cobián-Rojas, D., Chevalier-Monteagudo, P. P., Angulo-Valdes, J. A., & Rodriguez-Viera, L. (2023). Biology and ecology of the lionfish Pterois volitans/Pterois miles as invasive alien species: a review. PeerJ, 11, e15728.

Dell, C. L., Longo, G. O., Burkepile, D. E., & Manfrino, C. (2020). Few herbivore species consume dominant macroalgae on a Caribbean coral reef. Frontiers in Marine Science, 7, 676.

Deschamps, A., Desrochers, A., & Klomp, K. D. (2003). A rapid assessment of the Horseshoe Reef, Tobago Cays Marine Park, St. Vincent, West Indies(stony corals, algae and fishes). Atoll Research Bulletin, 496(2), 438-458.

Deutsch, C., Penn, J. L., & Lucey, N. (2024). Climate, oxygen, and the future of marine biodiversity. Annual Review of Marine Science, 16(1), 217-245.

Devlin, M., & Brodie, J. (2023). Nutrients and eutrophication. In Marine pollution–monitoring, management and mitigation (pp. 75-100). Cham: Springer Nature Switzerland.

Di Cintio, A., Niccolini, F., Scipioni, S., & Bulleri, F. (2023). Avoiding "paper parks": a global literature review on socioeconomic factors underpinning the effectiveness of marine protected areas. Sustainability, 15(5), 4464.

Diez, S. M., Patil, P. G., Morton, J., Rodriguez, D. J., Vanzella, A., Robin, D. V., ... & Corbin, C. (2019). Marine pollution in the Caribbean: not a minute to waste.

Dobbelaere, T., Dekens, A., Saint-Amand, A., Alaerts, L., Holstein, D. M., & Hanert, E. (2024). Hurricanes enhance coral connectivity but also superspread coral diseases. Global Change Biology, 30(6), e17382.

Doherty, M. L., Johnson, J. V., & Goodbody-Gringley, G. (2025). Widespread coral bleaching and mass mortality during the 2023–2024 marine heatwave in Little Cayman. PLoS One, 20(5), e0322636.

Domeier, M. L., and Colin, P. L. (1997). Tropical reef fish spawning aggregations: Defined and reviewed. Bull. Mar. Sci. 60, 698–726.

Dorrestein, E. H., Conan, A., Pentzke-Lemus, L. L., Hartman, G., Sample, S. H., & Dennis, M. M. (2020). Prevalence and progression of macroscopic lesions in Orbicella annularis and O. faveolata on shallow fringing reefs of St. Kitts. Diseases of Aquatic Organisms, 140, 79-95.

Dosell, A., Edwards, D., Gregory, A., Ponteen, A., O'Garro, J., Cornick, L., & Hawkridge, J. M. (2021). Using evidence from voluntary fisheries data collection programmes to support marine spatial planning and resolve multiple-use conflicts. Frontiers in Marine Science, 8, 635890.

Drakes, C., Laing, T., Kemp-Benedict, E., & Cashman, A. (2017). Caribbean scenarios 2050: GoLoCarSce report. Bridgetown, Barbados: Centre for Resource Management and Environmental Studies (CERMES).

Dromard, C. R., Devault, D. A., Bouchon-Navaro, Y., Allénou, J. P., Budzinski, H., Cordonnier, S., ... & Bouchon, C. (2022). Environmental fate of chlordecone in coastal habitats: recent studies conducted in Guadeloupe and Martinique (Lesser Antilles). Environmental Science and Pollution Research, 29, 51-60.

Dromard, C. R., Bouchon-Navaro, Y., Cordonnier, S., Guéné, M., Harmelin-Vivien, M., & Bouchon, C. (2018). Different transfer pathways of an organochlorine pesticide across marine tropical food webs assessed with stable isotope analysis. PloS one, 13(2), e0191335.

Dromard, C. R., Guéné, M., Bouchon-Navaro, Y., Lemoine, S., Cordonnier, S., & Bouchon, C. (2018). Contamination of marine fauna by chlordecone in Guadeloupe: evidence of a seaward decreasing gradient. Environmental Science and Pollution Research, 25(15), 14294-14301.

Dromard, C. R., Mccoy, C. M., & Turner, J. R. (2010). Measuring the performance of marine protected areas: the case of Little Cayman and Cayman Brac, Cayman Islands. In Proceedings of the Gulf and Caribbean Fisheries Institute (Vol. 63, pp. 246-253). Gulf and Caribbean Fisheries Institute, c/o Harbor Branch Oceanographic Institution, Inc. Fort Pierce FL 34946 United States.

Dungen, E. V. D. (2023). Reconstructing the impact of land-use changes on coastal lagoons in the Caribbean.

Duran, A., Shantz, A. A., Burkepile, D. E., Collado-Vides, L., Ferrer, V. M., Palma, L., ... & González-Díaz, P. (2018). Fishing, pollution, climate change, and the long-term decline of coral reefs off Havana, Cuba. Bulletin of Marine Science, 94(2), 213-228.

Dustan, P., & Halas, J. C. (1987). Changes in the reef-coral community of Carysfort Reef, Key Largo, Florida: 1974 to 1982. Coral Reefs, 6, 91-106.

Díaz, J. M., Barrios, L. M., Cendales, M. H., Garzón-Ferreira, J., Geister, J., López-Victoria, M., Ospina, G. H., Parra-Velandia, F., Pinzón, J., Vargas-Ángel, B., Zapata, F., & Zea, S. (2000). Áreas coralinas de Colombia. Serie de Publicaciones Especiales No. 5. Instituto de Investigaciones Marinas y Costeras - INVEMAR.

Díaz-Ferguson, E. E., & Hunter, M. E. (2019). Life history, genetics, range expansion and new frontiers of the lionfish (Pterois volitans, Perciformes: Pteroidae) in Latin America. Regional Studies in Marine Science, 31, 100793.

Díaz-López, A.M., Hernández-Arana, H.A., Vega-Zepeda, A., Ruiz-Zárate, M.A., Victoria-Salazar, I. (2023). Changes in the community structure of stony corals in the southern Mexican Caribbean. Mar. Environ. Res., 191, 106154.

Díaz-Osorio, A. C., Schmitter-Soto, J. J., Vega-Zepeda, A., & Espinoza-Tenorio, A. (2022). How effective are marine parks in protecting their coral reef ecosystem? A study case in the Mexican Caribbean. Aquatic Conservation: Marine and Freshwater Ecosystems, 32(7), 1126-1140.

Eagleson, R. G., Lumsden, J. S., Álvarez-Filip, L., Herbinger, C. M., & Horricks, R. A. (2021). Coverage increases of Porites astreoides in Grenada determined by shifts in size-frequency distribution. Diversity, 13(7), 288.

Eakin, C. M., Sweatman, H. P., & Brainard, R. E. (2019). The 2014–2017 global-scale coral bleaching event: insights and impacts. Coral Reefs, 38(4), 539-545.

Eakin, C. M., Morgan, J. A., Heron, S. F., et. al. (2010). Caribbean corals in crisis: Record thermal stress, bleaching, and mortality in 2005. PLoS ONE, 5(11): e13969

Eakin, C. M., Feingold, J. S., & Glynn, P. W. (1993). Oil refinery impacts on coral reef communities in Aruba. In Proceedings of the Colloquium on Global Aspects of Coral Reefs, Health, Hazards and History (pp. 139-145).

Eaton, K. R., Clark, A. S., Curtis, K., Favero, M., Hanna Holloway, N., Ewen, K., & Muller, E. M. (2022). A highly effective therapeutic ointment for treating corals with black band disease. Plos one, 17(10), e0276902.

Edgar, G. J., Cooper, A., Baker, S. C., Barker, W., Barrett, N. S., Becerro, M. A., ... & Stuart-Smith, R. D. (2020). Establishing the ecological basis for conservation of shallow marine life using Reef Life Survey. Biological Conservation, 252, 108855.

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., ... & Thomson, R. J. (2014). Global conservation outcomes depend on marine protected areas with five key features. Nature, 506(7487), 216-220.

Edmunds, P. J., Schils, T., & Wilson, B. (2023). The rising threat of peyssonnelioid algal crusts on coral reefs. Current Biology, 33(21), R1140-R1141.

Edmunds, P. J. (2022). Persistence of a sessile benthic organism promoted by a morphological strategy combining sheets and trees. Proceedings of the Royal Society B, 289(1978), 20220952.

Edmunds, P. J., Didden, C., & Frank, K. (2021). Over three decades, a classic winner starts to lose in a Caribbean coral community. Ecosphere, 12(5), e03517.

Edmunds, P. J., Zimmermann, S. A., & Bramanti, L. (2019b). A spatially aggressive peyssonnelid algal crust (PAC) threatens shallow coral reefs in St. John, US Virgin Islands. Coral Reefs, 38(6), 1329-1341.

Edmunds, P. J. (2019a). Three decades of degradation lead to diminished impacts of severe hurricanes on Caribbean reefs. Ecology, 100(3), e02587.

Edmunds, P. J. (2015). A quarter-century demographic analysis of the Caribbean coral, Orbicella annularis, and projections of population size over the next century. Limnology and Oceanography, 60(3), 840-855.

Edwards, P., Pena, M., Medeiros, R. P., & McConney, P. (2018). Socioeconomic Monitoring for Sustainable Small-Scale Fisheries: Lessons from Brazil, Jamaica, and St. Vincent and the Grenadines. In Viability and Sustainability of Small-Scale Fisheries in Latin America and The Caribbean (pp. 267-293). Cham: Springer International Publishing.

Edwards, C. B., Friedlander, A. M., Green, A. G., Hardt, M. J., Sala, E., Sweatman, H. P., ... & Smith, J. E. (2014). Global assessment of the status of coral reef herbivorous fishes: evidence for fishing effects. Proceedings of the Royal Society B: Biological Sciences, 281(1774), 20131835.

Eftec (2019). A National Ecosystem Assessment of the UK Overseas Territory of Montserrat: Natural Capital Assessments, Mapping and Monitoring Methods – Phase II Contract Number: C17-0304-1159 Final Report For the Joint Nature Conservation Committee. Peterborough: JNCC.

Eicher, H. (2024). Evaluating coral ecosystem health in Aruba–the development and future of Aruba's coral reefs (Master's thesis).

Eisenbach, O. J., Johnston, M. A., O'Connell, K., Hannum, R., French, D. R., Nuttall, M. F., & Cresswell, J. (2025). Long-term monitoring at East and West Flower Garden Banks: 2023 annual report. National Marine Sanctuaries Conservation Series ONMS-25-01. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of National Marine Sanctuaries.

Ellis, M. (2021). Implications for Resilience in the Cruise Tourism-Marine Protected Area Nexus in the Caribbean: The Case of St. Lucia. In Managing Crises in Tourism: Resilience Strategies from the Caribbean (pp. 67-87). Cham: Springer International Publishing.

Elmer, F., Kohl, Z. F., Johnson, P. T., & Peachey, R. B. (2019). Black spot syndrome in reef fishes: using archival imagery and field surveys to characterize spatial and temporal distribution in the Caribbean. Coral Reefs, 38(6), 1303-1315.

Elsner, J. B., Kossin, J. P., & Jagger, T. H. (2008). The increasing intensity of the strongest tropical cyclones. Nature, 455(7209), 92-95.

Emslie, M. J., Ceccarelli, D. M., Logan, M., Blandford, M. I., Bray, P., Campili, A., ... and Sinclair-Taylor, T. H. (2024). Changing dynamics of Great Barrier Reef hard coral cover in the Anthropocene. Coral Reefs, 43(3), 747-762.

Environmental Law Institute (2015). Legal Frameworks for MPA Enforcement in the Caribbean: Challenges and Opportunities. Prepared for the Gulf and Caribbean Fisheries Institute.

Erdoğan, E. (2021). The terrestrial consequences of poor wastewater management in Curaçao: how exploring public health risks and ground water quality levels can provide insights into establishing its sustainable management for small island development states (Master's thesis).

Escovar-Fadul, X., Hein, M. Y., Garrison, K., McLeod, E., Eggers, M., & Comito, F. (2022). A Guide to Coral Reef Restoration for the Tourism Sector: Partnering with Caribbean Tourism Leaders to Accelerate Coral Restoration. The Nature Conservancy.

Espinosa, J., Estrada, R., & Ruiz-Allais, J. (2023). Presencia en Cuba de la especie marina invasora" Unomia stolonifera"(Gohar, 1938)(Octocorallia, Alcyonacea): Acciones para su control y eliminación. Revista de Investigaciones Marinas, 43(1), 140-146.

Espinosa-Sáez, J. E., Estrada, R., & Ruiz-Allais, J. (2023). Presencia en Cuba de la especie marina invasora" Unomia stolonifera"(Gohar, 1938) (Octocorallia, Alcyonacea): Acciones para su control y eliminación. Revista de Investigaciones Marinas, 43(1), 140-146.

Estep, A., Croquer, A., Flower, J., Brandtneris, V., Mandoske, J., Bond, M., & Miller, K. (2018). Marine scientific assessment: A description of Montserrat's nearshore marine environment. Washington, DC: Waitt Institute.

Estrada, R. E., Morales, G. M., Hernández-Albernas, J., Borrego Acevedo, R., Acosta, J. O., Betancourt, Y. C., ... & de la Fuente, L. C. (2023). Physical-Geographic Characteristics of Cuban Reefs. In Coral Reefs of Cuba (pp. 51-73). Cham: Springer International Publishing.

Estrada-Saldívar, N., Quiroga-García, B. A., Pérez-Cervantes, E., Rivera-Garibay, O. O., & Alvarez-Filip, L. (2021). Effects of the stony coral tissue loss disease outbreak on coral communities and the benthic composition of cozumel reefs. Frontiers in Marine Science, 8, 632777.

Faricha, A., Hadi, T. A., Putra, R. D., & Handayani, T. (2023). The ability of surgeonfish for supporting coral reef resilience: a case study in Sabang Islands, Indonesia. Aquaculture, Aquarium, Conservation & Legislation, 16(3), 1430-1440.

Farr, T.G., Rosen, P.A., Caro, E., Crippen, R., Duren, R., Hensley, S., Kobrick, M., Paller, M., Rodriguez, E., Roth, L., Seal, D., Shaffer, S., Shimada, J., Umland, J., Werner, M., Oskin, M., Burbank, D., and Alsdorf, D.E., 2007, The shuttle radar topography mission: Reviews of Geophysics, v. 45, no. 2, RG2004, at https://doi.org/10.1029/2005RG000183. Retrieved from https://developers.google.com/earth-engine/datasets/catalog/USGS_SRTMGL1_003.

Feehan, C. J., Brown, M. S., Sharp, W. C., Lauzon-Guay, J. S., & Adams, D. K. (2016). Fertilization limitation of Diadema antillarum on coral reefs in the Florida Keys. Ecology, 97(8), 1897-1904.

Feely, R. A., Orr, J., Fabry, V. J., Kleypas, J. A., Sabine, C. L., & Langdon, C. (2009). Present and future changes in seawater chemistry due to ocean acidification. Geophysical Monograph Series, 183, 175-188.

Ferguson, A., & Daniel, S. (2015). Montserrat: Coral cay conservation annual marine survey report, year 2013–2014. Coral Cay Conservation Expeditions.

Fernández, C., J.J. Alvarado. (2004). El arrecife coralino de Punta Cocles, costa Caribe de Costa Rica. Revista Biología Tropical 52(Supplement 2): 121-129.

Ferrario, F., Beck, M. W., Storlazzi, C. D., Micheli, F., Shepard, C. C., & Airoldi, L. (2014). The effectiveness of coral reefs for coastal hazard risk reduction and adaptation. Nature communications, 5(1), 1-9. 10.1038/ncomms4794

Findlay, H. S., & Turley, C. (2021). Chapter 13—Ocean acidification and climate change, Editor (s): Trevor M. Letcher, Climate Change.

Fisch, J., Drury, C., Towle, E. K., Winter, R. N., & Miller, M. W. (2019). Physiological and reproductive repercussions of consecutive summer bleaching events of the threatened Caribbean coral Orbicella faveolata. Coral Reefs, 38(4), 863-876.

Flanders Marine Institute (2024). Marine and land zones: the union of world country boundaries and EEZ's, version 4. Retrieved from https://www.marineregions.org/downloads.php. Accessed on 2024-10-17.

Flanders Marine Institute (2023). Maritime Boundaries Geodatabase: Maritime Boundaries and Exclusive Economic Zones (200NM), version 12. Retrieved from https://www.marineregions.org/downloads.php. Accessed on 2024-06-16.

Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute. (2024). Dry Tortugas Coral Reef Evaluation and Monitoring (CREMP) [Dataset]. https://geodata.myfwc.com/documents/797abdd95d4146e1b7546d7df6a1ecf5/about (Accessed 2024-11-05).

Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute. (2024). Florida Keys Coral Reef Evaluation and Monitoring (CREMP) [Dataset]. https://geodata.myfwc.com/documents/2ab2e706c83d4247855f8e4e689c7cba/about (Accessed 2024-11-05).

Florida Keys Boundary Polygon. National Marine Sanctuary GIS. National Oceanic and Atmospheric Administration. https://sanctuaries.noaa.gov/library/imast_gis.html. Accessed on 2024-08-01.

Flower Garden Banks Boundary Polygon. National Marine Sanctuary GIS. National Oceanic and Atmospheric Administration. https://sanctuaries.noaa.gov/library/imast_gis.html. Accessed on 2024-08-01.

Flower, J., Ortiz, J. C., Chollett, I., Abdullah, S., Castro-Sanguino, C., Hock, K., ... & Mumby, P. J. (2017). Interpreting coral reef monitoring data: A guide for improved management decisions. Ecological indicators, 72, 848-869.

Flowers, K. I., Babcock, E. A., Papastamatiou, Y. P., Bond, M. E., Lamb, N., Miranda, A., ... & Chapman, D. D. (2022). Varying reef shark abundance trends inside a marine reserve: evidence of a Caribbean reef shark decline. Marine Ecology Progress Series, 683, 97-107.

Fonseca, A. C. (2008). Coral Reefs of Miskitus Cays, Nicaragua. Gulf and Caribbean Research, 20(1), 1–10. DOI:10.18785/gcr.2001.02

Fonseca, A.C. (2003). A rapid assessment at Cahuita National Park, Costa Rica, 1999 (Part 1: stony corals and algae). Atoll Research Bulletin 496: 248-257.

Forrester, G., Arton, L., Horton A., & Aeby, G. (2024). The relative effectiveness of chlorine treatment for Stony Coral Tissue Loss Disease. Frontiers in Marine Science, 11, 1465173.

Forster, P. M., Smith, C., Walsh, T., Lamb, W. F., Lamboll, R., Hall, B., ... & Zhai, P. (2024). Indicators of Global Climate Change 2023: annual update of key indicators of the state of the climate system and human influence. Earth System Science Data, 16(6), 2625-2658.

Frade, P. R., Bongaerts, P., Baldwin, C. C., Trembanis, A. C., Bak, R. P., & Vermeij, M. J. (2019). Bonaire and Curaçao. In Mesophotic coral ecosystems (pp. 149-162). Cham: Springer International Publishing.

Franklin, A. (2007). Determination of the socio-economic importance of the lobster fishery of the British Virgin Islands. Caribbean Natural Resources Institute (CANARI)

Friedrich, T., Timmermann, A., Abe-Ouchi, A., Bates, N. R., Chikamoto, M. O., Church, M. J., ... & Santana-Casiano, J. M. (2012). Detecting regional anthropogenic trends in ocean acidification against natural variability. Nature Climate Change, 2(3), 167-171.

Frölicher, T. L., Fischer, E. M., & Gruber, N. (2018). Marine heatwaves under global warming. Nature, 560(7718), 360-364.

Fuentes-Gandara, F., Pinedo-Hernández, J., Gutiérrez, E., Marrugo-Negrete, J., & Díez, S. (2021). Heavy metal pollution and toxicity assessment in Mallorquin swamp: A natural protected heritage in the Caribbean Sea, Colombia. Marine pollution bulletin, 167, 112271.

Funes, M., Zylich, K., Divovich, E., Zeller, D., Lindop, A., Pauly, D., & Box, S. (2015). Honduras, a fish exporting country: preliminary re-constructed marine catches in the Caribbean Sea and The Gulf of Fonseca, 1950 – 2010, (90), 17.

G.B.F. (2022). Kunming-montreal global biodiversity framework. In Convention Biol. Divers. Kunming-Montreal Glob. Biodivers. Framew.(cbd. int).

Gahtan, J., K. R. Knapp, C. J. Schreck, H. J. Diamond, J. P. Kossin, M. C. Kruk, (2024). International Best Track Archive for Climate Stewardship (IBTrACS) Project, Version 4r01. NOAA National Centers for Environmental Information. doi:10.25921/82ty-9e16. Accessed on 2024-07-23.

Ganase A and Lochan H (2022). The Status of Coral Reefs in Tobago, 2020. Institute of Marine Affairs, Hilltop Lane, Chaguaramas, Trinidad and Tobago, pp54.

Garcia Ch., M.C, (2022). The ebb and flow of the Seaflower marine biosphere reserve: Law entanglements and socio-environmental justice in the southwestern Caribbean Sea. Geogr J. 2022;00:1–20.

Garcias-Sais, J. R., Williams, S. M., Sabater-Clavell, J., & Carlo, M. (2023). Puerto Rico Coral Reef Monitoring Program: 2023 Survey. Final report. Coral Reef Conservation Program NA21NOS4820014 State and Territories Conservation Cooperative Agreement.

García, E. M., Bastidas, C., Cruz-Motta, J. J., & Farina, O. (2011). Metals in waters and sediments of the Morrocoy National Park, Venezuela: increased contamination levels of cadmium over time. Water, Air, & Soil Pollution, 214(1), 609-621.

García, A., Cróquer, A., & Pauls, S. M. (2003). Estado actual de las enfermedades y otros signos de deterioro coralino en siete arrecifes del Parque Nacional Archipiélago de Los Roques, Venezuela. Revista de biología tropical, 51(S4), 173-180.

Gardner, L. (2009). Protected Areas Management in the Caribbean: Core Themes for Education, Awareness, and Communication Programmes. The Trust For Sustainable Livelihoods and WCPA Caribbean.

Gardner, T. A., Côté, I. M., Gill, J. A., Grant, A., & Watkinson, A. R. (2005). Hurricanes and Caribbean coral reefs: impacts, recovery patterns, and role in long-term decline. Ecology, 86(1), 174-184.

Gardner, T. A., Côté, I. M, Gill, J. A., Grant, A., & Watkinson, A. R. (2003). Long-term region-wide declines in Caribbean corals. Science, 301, 958–960.

Gelfand, S. (2025). Mermaid: Interface to the 'MERMAID' API. R package version 1.2.5. https://github.com/data-mermaid/mermaidr.

Gibson, J. (2011). The Belize barrier reef: a world heritage site. Fisheries Centre Research Reports, 19(6), 8.

Gil-Agudelo, D. L., Cintra-Buenrostro, C. E., Brenner, J., González-Díaz, P., Kiene, W., Lustic, C., & Pérez-España, H. (2020). Coral Reefs in the Gulf of Mexico Large Marine Ecosystem: Conservation Status, Challenges, and Opportunities. Frontiers in Marine Science, 6, 807.

Gil-Agudelo, D.L., Garzón-Ferreira, J., Rodríguez-Ramírez, A., Reyes-Nivia, M. C., Navas-Camacho, R., Venera-Pontón, D. E., Díaz–Pulido, G., Sánchez, J. A. (2005). Estado de los arrecifes coralinos en Colombia. En: Informe del Estado de los Ambientes Marinos y Costeros de Colombia: 2005. Serie de publicaciones periódicas del INVEMAR, No. 8, Santa Marta. 360 pp.

Gil-Agudelo, D. L., Smith, G. W., Garzón-Ferreira, J., Weil, E., & Petersen, D. (2004). Dark spots disease and yellow band disease, two poorly known coral diseases with high incidence in Caribbean reefs. In Coral health and disease (pp. 337-349). Berlin, Heidelberg: Springer Berlin Heidelberg.

Gill, J. A., Watkinson, A. R., McWilliams, J. P., & Côté, I. M. (2006). Opposing forces of aerosol cooling and El Nino drive coral bleaching on Caribbean reefs. Proceedings of the National Academy of Sciences, 103(49), 18870-18873.

Giró Petersen, A. (2020). Evaluación del estado de salud de los arrecifes del Caribe de Guatemala: Una década de datos y los cambios en el tiempo (Master's thesis). Universidad de San Carlos de Guatemala, Centro de Estudios del Mar y Acuicultura. 56 pp.

Giró, A., & Mojica, A. (2022). Cayman Crown reef characterization document (26 pp.). Mesoamerican Reef Fund; Healthy Reefs for Healthy People; Pixan'Ja.

Gladfelter, W. B. (1982). White-band disease in Acropora palmata: implications for the structure and growth of shallow reefs. Bulletin of marine Science, 32(2), 639-643.

Gledhill, D. K., Wanninkhof, R., Millero, F. J., & Eakin, M. (2008). Ocean acidification of the greater Caribbean region 1996–2006. Journal of Geophysical Research: Oceans, 113(C10).

Global Coral Reef Monitoring Network (2024). Polygons of GCRMN regions and subregions. https://github.com/GCRMN/gcrmn_regions.

Glynn, P. W. (1993). Coral reef bleaching: ecological perspectives. Coral reefs, 12(1), 1-17.

Gochfeld, D. J., Olson, J. B., Chaves-Fonnegra, A., Smith, T. B., Ennis, R. S., & Brandt, M. E. (2020). Impacts of hurricanes Irma and Maria on coral reef sponge communities in St. Thomas, US Virgin Islands. Estuaries and Coasts, 43(5), 1235-1247.

Goddijn-Murphy, L., Woolf, D., & James, N. A. (2025). Patterns and trends in marine microplastics density distributions using a long-term, global, field database. Marine Pollution Bulletin, 212, 117528.

Gombos, M., Arrivillage, A., Wusinich-Mendez, D., Glazer, R., Frew, S., Bustamante, G., ... & Causey, B. (2021). A management capacity assessment of selected coral reef marine protected areas in the Caribbean.

González-Díaz, P., González-Sansón, G., Aguilar Betancourt, C., Alvarez Fernandez, S., Perera Perez, O., Hernandez Fernandez, L., ... & de la Guardia Llanso, E. (2018). Status of Cuban coral reefs. Bulletin of Marine Science, 94(2), 229-247.

González Díaz, S. P. (2010). Efecto acumulativo de agentes estresantes múltiples sobre los corales hermatípicos de la región noroccidental de Cuba.

González-Gándara, C., & de la Cruz-Francisco, V. (2014). Unusual record of the Indo-Pacific pomacentrid Neopomacentrus cyanomos (Bleeker, 1856) on coral reefs of the Gulf of Mexico. BioInvasion Records, 3(1), 49–52.

González-Barrios, F. J., Estrada-Saldívar, N., Pérez-Cervantes, E., Secaira-Fajardo, F., & Álvarez-Filip, L. (2023). Legacy effects of anthropogenic disturbances modulate dynamics in the world's coral reefs. Global Change Biology, 29(12), 3285-3303.

Good, A. M., and Bahr, K. D. (2021). The coral conservation crisis: interacting local and global stressors reduce reef resiliency and create challenges for conservation solutions. SN Applied Sciences, 3(3), 312.

Goodbody-Gringley, G., & Chequer, A. D. (2025). Mesophotic reefs offer thermal refuge to the 2023 Caribbean mass bleaching event in the Cayman Islands. Scientific Reports, 15(1), 16496.

Gore, S., Wynne, S. P., & Myers, A. (2019). UK Overseas Territories in the Northeast Caribbean: Anguilla, British Virgin Islands, Montserrat. In World Seas: an Environmental Evaluation (pp. 549-565). Academic Press.

Goreau, T.J.F., R.L. Hayes. (2024) Record marine heat waves: coral reef bleaching HotSpot maps reveal global sea surface temperature extremes, coral mortality, and ocean circulation changes, Oxford Open Climate Change, 4(1).

Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D., & Moore, R. (2017). Google Earth Engine: Planetary-scale geospatial analysis for everyone. Remote sensing of Environment, 202, 18-27.

Gorstein, M., J. Loerzel, P. Edwards, A. Levine. (2019). National Coral Reef Monitoring Program Socioeconomic Monitoring Component: Summary Findings for USVI, 2017. U.S. Dep. Commerce, NOAA Tech. Memo., NOAA-TM-NOS-CRCP-35, 72p. + Appendices. DOI:10.25923/40mw-kw93.

Gottlieb, B., Pruckner, S., & Anthony, B. P. (2024). An ecological coherence assessment of the Wider Caribbean Region marine protected area network. Ocean & Coastal Management, 255, 107249.

Gove, J. M., Williams, G. J., Lecky, J., Brown, E., Conklin, E., Counsell, C., ... & Asner, G. P. (2023). Coral reefs benefit from reduced land–sea impacts under ocean warming. Nature, 621(7979), 536-542.

Graham, R. E. (2024). A study of illegal dumping near coastal zones and waterways in Grenada: Advancing waste management solutions and reducing marine litter. PLoS Water, 3(4), e0000175.

Graham, R. T., Carcamo, R., Rhodes, K. L., Roberts, C. M., & Requena, N. (2008). Historical and contemporary evidence of a mutton snapper (Lutjanus analis Cuvier, 1828) spawning aggregation fishery in decline. Coral reefs, 27, 311-319.

Grant Thornton and Vrije Universiteit Amsterdam (2025). An assessment of the coastal resilience of Sint Maarten. 82p

Grant, R., Rhode-Barbarigos, L., Roy, S. S., Britton, L., Li, C., Rowe, A., ... & Bello, M. (2024). No port stands alone: PortMiami and the resilience of its Caribbean and Mesoamerican maritime network. Maritime Economics & Logistics, 1-24.

Gratwicke, B., & Speight, M. R. (2005). The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. Journal of Fish Biology, 66(3), 650–667.

Green, R. H., Lowe, R. J., Buckley, M. L., Foster, T., & Gilmour, J. P. (2019). Physical mechanisms influencing localized patterns of temperature variability and coral bleaching within a system of reef atolls. Coral Reefs, 38(4), 759-771.

Green, SJ, Côté IM (2014) Trait-based diet selection: prey behaviour and morphology predict vulnerability to predation in reef fish communities. Journal of Animal Ecology 83: 1451– 667 1460.

Green, S. J., Akins, J. L., Maljković, A., & Côté, I. M. (2012). Invasive lionfish drive Atlantic coral reef fish declines. PloS one, 7(3), e32596.

Green, SJ, Akins J, Côté I (2011) Foraging behaviour and prey consumption in the Indo-Pacific lionfish on Bahamian coral reefs. Marine Ecology Progress Series 433: 159–167.

Green, S., & Cote, I. (2010). Consumption potential of invasive lionfish (Pterois volitans) on Caribbean coral reefs. In Proceedings of the Gulf and Caribbean Fisheries Institute (Vol. 62, pp. 358-359). Gulf and Caribbean Fisheries Institute, c/o Harbor Branch Oceanographic Institution, Inc. Fort Pierce FL 34946 United States.

Green, D. H., Edmunds, P. J., & Carpenter, R. C. (2008). Increasing relative abundance of Porites astreoides on Caribbean reefs mediated by an overall decline in coral cover. Marine Ecology Progress Series, 359, 1-10.

Green, E. P., & Bruckner, A. W. (2000). The significance of coral disease epizootiology for coral reef conservation. Biological conservation, 96(3), 347-361.

Grieve, B. D., Curchitser, E. N., & Rykaczewski, R. R. (2016). Range expansion of the invasive lionfish in the Northwest Atlantic with climate change. Marine Ecology Progress Series, 546, 225-237.

Griffith, A., Buglisi, M., Lee, S., & Castillo, K. (2025). Tracking over 30 years of coral reef infrastructure degradation in Barbados. Scientific Reports, 15(1), 3907.

Grossman, G. D., Jones, G. P., Seaman Jr, W. J., Can, M. H., Hixon, M. A., Steimle, F. W., ... & Coutant, C. C. (1997). Special Issue on Artificial Reef Management. Fisheries.

Grupo UKT (2005). Informe anti-Xenia umbellata en Boca de Calderas del 16 de Julio del 2025. in www.agrra.org/wp-content/uploads/2025/07/Informe-anti-Xenia-Caldera_16-julio-2025-.pdf

Guarderas, A. P., Hacker, S. D., & Lubchenco, J. (2008). Current status of marine protected areas in Latin America and the Caribbean. Conservation Biology, 22(6), 1630-1640.

Guinotte, J. M., & Fabry, V. J. (2008). Ocean acidification and its potential effects on marine ecosystems. Annals of the New York Academy of Sciences, 1134(1), 320-342.

Gumbs, K., Wynne, S., & Johnson, R. (2015). Anguilla fisheries development plan. Government of Anguilla.

Guzman, H. M., Kaiser, S., & Weil, E. (2020). Assessing the long-term effects of a catastrophic oil spill on subtidal coral reef communities off the Caribbean coast of Panama (1985–2017). Marine Biodiversity, 50(3), 28.

Guzmán, H. M., Benfield, S., Breedy, O., & Mair, J. M. (2003). Broad patterns of coral reef community structure in the Caribbean of Panama. Bulletin of Marine Science, 73(2), 313–331.

Guzmán, H. M., & Jiménez, C. E. (1992). Contamination of coral reefs by heavy metals along the Caribbean coast of Central America (Costa Rica and Panama). Marine Pollution Bulletin, 24(11), 554–561.

Gómez, C., Millán, S., Acosta-Chaparro, A., Abril-Howard, A., & Navas-Camacho, R. (2022). Evaluación preliminar del impacto del huracán Iota en arrecifes coralinos de las islas de Providencia y Santa Catalina. Boletín De Investigaciones Marinas Y Costeras, 51(1), 193–200.

Harborne, A. R., Renaud, P. G., Tyler, E. H. M., & Mumby, P. J. (2009). Reduced density of the herbivorous urchin Diadema antillarum inside a Caribbean marine reserve linked to increased predation pressure by fishes. Coral Reefs, 28(3), 783-791.

Hardt, M. J. (2009). Lessons from the past: the collapse of Jamaican coral reefs. Fish and fisheries, 10(2), 143-158.

Harmelin-Vivien, M. L. (1994). The effects of storms and cyclones on coral reefs: a review. Journal of Coastal Research, 211-231.

Harms-Tuohy, C. A. (2021). Parrotfishes in the Caribbean: a regional review with recommendations for management (Vol. 1240). Food & Agriculture Org.

Harvell, C. D., Jordán-Dahlgren, E., Merkel, S., Rosenberg, E., Raymundo, L., Smith, G., Weil, E., & Willis, B. L. (2007). Coral diseases, Environmental drivers and the balance between corals and microbial associates. Oceanography, 20(1), 172–195.

Hawkins, J. P., Roberts, C. M., Dytham, C., Schelten, C., & Nugues, M. M. (2006). Effects of habitat characteristics and sedimentation on performance of marine reserves in St. Lucia. Biological Conservation, 127(4), 487-499.

Hawkins, J. P., & Roberts, C. M. (2004). Effects of artisanal fishing on Caribbean coral reefs. Conservation Biology, 18(1), 215-226.

Hayes, N. K., Walton, C. J., & Gilliam, D. S. (2022). Tissue loss disease outbreak significantly alters the Southeast Florida stony coral assemblage. Frontiers in Marine Science, 9, 975894.

Heidmann, S. L., R. S. Nemeth, C. R. Biggs, E. Kadison, B. L. Kojis. (2024b). Diel movement, space use and efficacy of a seasonal closed area for management of a Lutjanus analis spawning aggregation site. Marine Ecology Progress Series. doi.org/10.3354/meps14620

Heidmann, S. L., L. K. Olinger, V. W. Brandtneris, R. S. Ennis, J. Blondeau, L, J. W. Grove and T. B. Smith. (2024a). Depth and structure as environmental drivers of fish communities across a shallow to mesophotic gradient in the northern US Virgin Islands. Bull. Mar. Sci.

Hein, M. Y., Vardi, T., Shaver, E. C., Pioch, S., Boström-Einarsson, L., Ahmed, M., ... & McLeod, I. M. (2021). Perspectives on the use of coral reef restoration as a strategy to support and improve reef ecosystem services. Frontiers in Marine Science, 8, 618303.

Hein MY, McLeod IM, Shaver EC, Vardi T, Pioch S, Boström-Einarsson L, Ahmed M, Grimsditch G (2020). Coral Reef Restoration as a strategy to improve ecosystem services – A guide to coral restoration methods. United Nations Environment Program, Nairobi, Kenya.

Hein, M. Y., Beeden, R., Birtles, A., Gardiner, N. M., Le Berre, T., Levy, J., ... & Willis, B. L. (2020). Coral restoration effectiveness: multiregional snapshots of the long-term responses of coral assemblages to restoration. Diversity, 12(4), 153.

Hein, M. Y., Birtles, A., Willis, B. L., Gardiner, N., Beeden, R., & Marshall, N. A. (2019). Coral restoration: Socio-ecological perspectives of benefits and limitations. Biological conservation, 229, 14-25.

Henwood, W. D., Aucoin, S., & Turner, M. (2017). The Protected Area of Managed Natural Resources of the Three Bays (PA3B): Management Plan 2017-2027.

Heres, M. M., Farmer, B. H., Elmer, F., & Hertler, H. (2021). Ecological consequences of stony coral tissue loss disease in the Turks and Caicos Islands. Coral Reefs, 40(2), 609-624.

Hernández-Delgado, E. A., Alejandro-Camis, P., Cabrera-Beauchamp, G., Fonseca-Miranda, J. S., Gómez-Andújar, N. X., Gómez, P., ... & Suleimán-Ramos, S. E. (2024). Stronger hurricanes and climate change in the Caribbean Sea: Threats to the sustainability of endangered coral species. Sustainability, 16(4), 1506.

Hernández-Delgado, E. A., Toledo-Hernández, C., Mercado-Molina, A. E., Suleimán-Ramos, S. E., Candelas-Sánchez, F., Fonseca, J., ... & Maldonado, M. (2014). Community-based coral reef rehabilitation in a changing climate: Lessons learned from Puerto Rico. Marine Ecology Progress Series, 506, 163–178.

Hernández-Delgado, E. A. (2000). Effects of anthropogenic stresses on Puerto Rican coral reef communities. Proceedings of the 9th International Coral Reef Symposium, 1, 493–498.

Hernández-Solís, P. A. (2024). Análisis de conectividad entre plataformas petroleras y sistemas arrecifales utilizando modelos de dispersión larval en el sur del Golfo de México. Tesis de Maestría. Unversidad Nacional Autónoma de México.

Heron, S. F., Maynard, J. A., Van Hooidonk, R., & Eakin, C. M. (2016). Warming trends and bleaching stress of the world's coral reefs 1985–2012. Scientific reports, 6(1), 38402.

Herrera-Silveira, J. A., Pech-Cardenas, M. A., Morales-Ojeda, S. M., Cinco-Castro, S., Camacho-Rico, A., Sosa, J. P. C., ... & Teutli-Hernandez, C. (2020). Blue carbon of Mexico, carbon stocks and fluxes: a systematic review. PeerJ, 8, e8790.

Hewson, I., Ritchie, I. T., Evans, J. S., Altera, A., Behringer, D., Bowman, E., ... & Breitbart, M. (2023). A scuticociliate causes mass mortality of Diadema antillarum in the Caribbean Sea. Science Advances, 9(16), eadg3200.

Higgins, E., Duppel, K., Connell, M., Brenna, G., & Sobolev, K. (2024). Coral reef rehabilitation following Hurricane Irma using nano-engineered artificial reefs in Sint Maarten. PeerJ, 12, e18487.

Hijmans, Robert J. (2015). University of California, Berkeley Museum of Vertebrate Zoology. International Rice Research Institute. University of California, Davis. Global administrative areas version 2.8. University of California, Berkeley. Museum of Vertebrate Zoology. Retrieved from https://maps.princeton.edu/catalog/stanford-jv457hb9421. Accessed on 2024-06-24.

Hill, T. S., & Hoogenboom, M. O. (2022). The indirect effects of ocean acidification on corals and coral communities. Coral Reefs, 41(6), 1557-1583.

Hill, J., & Wilkinson, C. L. I. V. E. (2004). Methods for ecological monitoring of coral reefs. Australian Institute of Marine Science, Townsville, 117.

Hilmi, N., Crisóstomo, M., Bermudo, A., Chami, R., Imbert, N., & Duarte, C. M. (2025). Resilience of Small Islands: Unveiling Nature-Based Solutions for Sustainable Futures. Climate-Resilient Cities, 257-280.

Hixon, M. A., Green, S. J., Albins, M. A., Akins, J. L., & Morris, J. A., Jr. (2016). Lionfish: A major marine invasion. Marine Ecology Progress Series, 558, 161–165.

Hixon, M. A., & Beets, J. P. (1993). Predation, prey refuges, and the structure of coral-reef fish assemblages. Ecological Monographs, 63(1), 77–101.

Hodgson, G., & Torres, R. E. (2022). Santé des récifs coralliens en République d'Haïti: rapport 2020 et 2021= Coral reef health in the Republic of Haiti: report 2020 and 2021.

Hoegh-Guldberg, O., Skirving, W., Dove, S. G., Spady, B. L., Norrie, A., Geiger, E. F., ... & Manzello, D. P. (2023). Coral reefs in peril in a record-breaking year. Science, 382(6676), 1238-1240.

Hoegh-Guldberg, O. (1999). Climate change, coral bleaching and the future of the world's coral reefs. Marine and freshwater research, 50(8), 839-866.

Hoeksema, B. W., van der Loos, L. M., van Moorsel, G. W. N. M. (2022). Coral diversity matches marine park zonation but not economic value of coral reef sites at St. Eustatius, eastern Caribbean. Journal of Environmental Management 320, 115829.

Hoeksema, B. W., Hassell, D., Meesters, E. H., & van Duyl, F. C. (2018). Wave-swept coralliths of Saba bank, Dutch Caribbean. Marine Biodiversity, 48(4), 2003-2016.

Hoeksema, B.W. (2016). Marine biodiversity survey of St. Eustatius, Dutch Caribbean, 2015. Naturalis Biodiversity Center, Leiden, and ANEMOON Foundation, Bennebroek, 157 pp.

Hollister, K. J., Ennis, R. S., Spalding, H. L., Gabrielson, P. W., & Smith, T. B. (2021). Caribbean corals exhibit species-specific differences in competitive abilities with an aggressive encrusting alga, Ramicrusta textilis. Coral Reefs, 40(6), 1729-1740.

Honey, M. (2019). Cruise Tourism in the Caribbean. Routledge.

Horn, C., Karnauskas, M., Doerr, J. C., Miller, M. H., Neuman, M., Hill, R., & McCarthy, K. J. (2022). Endangered species act status review report: Queen conch (Aliger gigas).

Horsford, I. (2017). Hurricane Irma, Preliminary damage Assessment for Antigua and Barbuda's Fisheries Sector. Fisheries Division, Ministry of Agriculture, Lands, Fisheries and Barbuda Affairs: St. John's, Antigua.

Horsford, I. (2008). The Grouper Fishery of Antigua and Barbuda. In Report on Regional Workshop of Nassau Grouper, Cartagena, Colombia (pp. 20-21).

Horta, P.A. et al. (2021). Marine Eutrophication: Overview from Now to the Future. In: Häder, DP., Helbling, E.W., Villafañe, V.E. (eds) Anthropogenic Pollution of Aquatic Ecosystems. Springer, Cham.

Hughes, D. J., Alderdice, R., Cooney, C., Kühl, M., Pernice, M., Voolstra, C. R., & Suggett, D. J. (2020). Coral reef survival under accelerating ocean deoxygenation. Nature Climate Change, 10(4), 296-307.

Hughes, T. P., Kerry, J. T., Baird, A. H., Connolly, S. R., Chase, T. J., Dietzel, A., ... & Woods, R. M. (2019). Global warming impairs stock–recruitment dynamics of corals. Nature, 568(7752), 387-390.

Hughes, T. P., Barnes, M. L., Bellwood, D. R., Cinner, J. E., Cumming, G. S., Jackson, J. B., ... and Scheffer, M. (2017). Coral reefs in the Anthropocene. Nature, 546(7656), 82-90.

Hughes, T. P., Graham, N. A., Jackson, J. B., Mumby, P. J., & Steneck, R. S. (2010). Rising to the challenge of sustaining coral reef resilience. Trends in ecology & evolution, 25(11), 633-642.

Hughes, T. P. (1994). Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. Science, 265(5178), 1547–1551.

Hughes, T. P., Reed, D. C., & Boyle, M. J. (1987). Herbivory on coral reefs: community structure following mass mortalities of sea urchins. Journal of Experimental Marine Biology and Ecology, 113(1), 39-59.

Hunt, A. (2003). A rapid assessment of the coral reef community structure and health of the Corn Islands using the AGRRA protocol. ReefMap Project Report for the Corn Islands and Pearl Cays.

Hutchinson, E., Matthews, T. R., & Renchen, G. F. (2024). Relationships between postlarval settlement and commercial landings of Caribbean spiny lobster (Panulirus argus) in Florida (USA). Fisheries Research, 279, 107137.

Hylkema, A., Debrot, A., Osinga, R., Nagelkerken, I., & van Kuijk, T, (2023). The 2022 Diadema antillarum die off event: Comparison with the 1983-1984 mass mortality. Frontiers in Marine Science, 10, 1119539.

Hylkema, A., Debrot, A. O., Van De Pas, E. E., Osinga, R., & Murk, A. J. (2022b). Assisted natural recovery: a novel approach to enhance Diadema antillarum recruitment. Frontiers in Marine Science, 9, 929355.

Hylkema, A., Debrot, A. O., Pistor, M., Postma, E., Williams, S. M., & Kitson-Walters, K. (2022a). High peak settlement of Diadema antillarum on different artificial collectors in the Eastern Caribbean. Journal of Experimental Marine Biology and Ecology, 549, 151693.

Hylkema, A., Hakkaart, Q. C., Reid, C. B., Osinga, R., Murk, A. J., & Debrot, A. O. (2021). Artificial reefs in the Caribbean: a need for comprehensive monitoring and integration into marine management plans. Ocean & Coastal Management, 209, 105672.

Idjadi, J. A., Haring, R. N., & Precht, W. F. (2010). Recovery of the sea urchin Diadema antillarum promotes scleractinian coral growth and survivorship on shallow Jamaican reefs. Marine Ecology Progress Series, 403, 91-100.

IFRECOR (2021). Initiative Française pour les Récifs Coralliens. État de santé des récifs coralliens, herbiers marins et mangroves des outre-mer français. Bilan 2020. 335 pp.

Impact Mer (2023). Suivi du blanchissement corallien de la Martinique, Octobre 2023. Directive Cadre Européenne sur l'Eau. Rapport de la première campagne. 16p + Annexes.

INEGI (2023). https://www.inegi.org.mx/temas/turismosat/#informacion_general

INSEE (2022). Institut national de la statistique et des études économiques. Séries historiques du recensement de la population, 2022.

Institute for Marine Remote Sensing, University of South Florida (IMaRS/USF), Institut de Recherche pour le Développement (IRD), UNEP-WCMC, The WorldFish Center, and WRI, 2011. Global Coral Reefs composite dataset compiled from multiple sources for use in the Reefs at Risk Revisited project incorporating products from the Millennium Coral Reef Mapping Project prepared by IMaRS/USF and IRD. Retrieved from https://datasets.wri.org/dataset/tropical-coral-reefs-of-the-world-500-m-resolution-grid. Accessed on 2021-08-10.

International Coral Reef Initiative (2024). Guidance Document on Integrating Coral Reefs and Associated Ecosystems into National Biodiversity Strategies and Action Plans. London, United Kingdom.

International Maritime Organisation (IMO), 2012. International Shipping Facts and Figures

INVEMAR-MinAmbiente (2020). Atlas de Áreas coralinas de Colombia. Disponible en: https://areas-coralinas-de-colombia-invemar.hub.arcgis.com

INVEMAR (2025). Informe del estado de los ambientes y recursos marinos y costeros en Colombia. Serie de Publicaciones Periódicas No. 3. Santa Marta. 210 p.

IPCC, 2021: Technical Summary. In Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 33-144.

IPCC, 2023: Summary for Policymakers. (2023) In: Climate Change 2023: Synthesis Report. Contribution of Working Groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, H. Lee and J. Romero (eds.)]. IPCC, Geneva, Switzerland, pp. 1-34.

Irazabal, I. and M. Rodríguez (2019). First report of Stony Coral Tissue Loss Disease in the Dominican Republic. Atlantic and Gulf Rapid Reef Assessment, Dominican Republic.

Irvine, J., Cox, A. and Howell, S. (2025). Barbados 2024 Status of Coral Reefs: Bleaching and Disease Report. Coastal Zone Management Unit.

Irvine, J., Oxenford, H.A., Cumberbatch, J., McConney, P., Hinds, C., Speede, R., Desrochers, A., Degia, K. et van Tussenbroek, B. (2023). Managing sargassum shoreline inundations: a toolbox. SargAdapt Good Practice Guide Series 5. University of the West Indies, Centre for Resource Management and Environmental Studies (UWI-CERMES), Barbados. 29 p.

Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (Eds.) (2014). Status and trends of Caribbean Coral Reefs: 1970–2012. Global Coral Reef Monitoring Network.

Jackson, J. B., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., ... & Warner, R. R. (2001). Historical overfishing and the recent collapse of coastal ecosystems. Science, 293(5530), 629-637.

Jaramillo, C., Pellón, E., Freitas, L. D., & González, M. (2025). Assessment of coastal morphodynamics in the reef-sheltered littoral of Sosúa, Dominican Republic.

Jeffrey, C., Hile, S., Christensen, J., Monaco, M., Caldow, C., Roberson, K., Clark, R., Bauer, L., Pittman, S., Kendall, M., Friedlander, A., & Morgan, J. (2019). NCCOS assessment: Biogeographic surveys of benthic and fish communities in the Caribbean and Gulf of Mexico from 2001-01-09 to 2012-11-02 (NCEI Accession 0206670) [Dataset]. NOAA National Centers for Environmental Information. https://doi.org/10.25921/rt0s-ty25 (Accessed 2025-01-30).

Jimenez, N. & Vega Rodríguez, M. (2025). Emergency Response Strategy to the invasive octocoral of the Xeniidae family. In www.agrra.org.

Jiménez, C. (2001). Bleaching and mortality of reef organisms during a warming event in 1995 on the Caribbean coast of Costa Rica. Revista de Biología Tropical, 49 (Supplement 2): 233–238.

Johnson, P. T., Malawauw, R. J., Piaskowy, J., Calhoun, D. M., Kohl, Z., ter Horst, L. J., & Zelmer, D. A. (2024). Emergence of black spot syndrome in Caribbean reefs: a century of fish collections reveal long-term increases in Scaphanocephalus infection. Proceedings B, 291(2034), 20242065.

Johnson, J. V., Chequer, A. D., & Goodbody-Gringley, G. (2023). Insights from the 2-year-long human confinement experiment in Grand Cayman reveal the resilience of coral reef fish communities. Scientific Reports, 13(1), 21806.

Johnson, M. D., Comeau, S., Lantz, C. A., & Smith, J. E. (2021). Coral reef deoxygenation: Drivers, consequences, and responses. Nature Reviews Earth & Environment, 2(9), 645–659

Johnson, M. D., Scott, J. J., Leray, M., Lucey, N., Bravo, L. M. R., Wied, W. L., & Altieri, A. H. (2021). Rapid ecosystem-scale consequences of acute deoxygenation on a Caribbean coral reef. Nature communications, 12(1), 4522.

Johnson, M. D., Swaminathan, S. D., Nixon, E. N., Paul, V. J., & Altieri, A. H. (2021). Differential susceptibility of reef-building corals to deoxygenation reveals remarkable hypoxia tolerance. Scientific Reports, 11(1), 23168.

Johnson, C. C., & Kauffman, E. G. (2001). Cretaceous evolution of reef ecosystems: a regional synthesis of the Caribbean tropics. The history and sedimentology of ancient reef systems, 311-349.

Johnston, M. A., French, D. R., Eisenbach, O. J., O'Connell, K., Hannum, R., Nuttall, M. F., & Emmert, J. (2024). Long-term monitoring at East and West Flower Garden Banks: 2022 annual report. National Marine Sanctuaries Conservation Series ONMS-24-06. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of National Marine Sanctuaries.

Johnston, M. A ., Studivan, M. S., Enochs, I. C., Correa, A., Besemer, N., Eckert, R. J., Edwards, K., Hannum, R., Hu, X., Nuttall, M., & O'Connell, K. (2023). Coral disease outbreak at the remote Flower Garden Banks, Gulf of Mexico. Frontiers in Marine Science, 10, 1111749.

Johnston, M. A., Hickerson, E. L., Nuttall, M. F., Blakeway, R. D., Sterne, T. K., Eckert, R. J., & Schmahl, G. P. (2019). Coral bleaching and recovery from 2016 to 2017 at East and West Flower Garden Banks, Gulf of Mexico. Coral Reefs, 38, 787–799.

Johnston, M. A., Nuttall, M. F., Eckert, R. J., Blakeway, R. D., Sterne, T. K., Hickerson, E. L., Schmahl, G. P., Lee, M. T., MacMillan, J., & Embesi, J. A. (2018). Localized coral reef mortality event at East Flower Garden Bank, Gulf of Mexico. Bulletin of Marine Science, 95, 239–250.

Johnston, M. W., & Purkis, S. J. (2015). Hurricanes accelerated the Florida–Bahamas lionfish invasion. Global change biology, 21(6), 2249-2260.

Johnston, M. W., & Purkis, S. J. (2011). Spatial analysis of the invasion of lionfish in the western Atlantic and Caribbean. Marine Pollution Bulletin, 62(6), 1218-1226.

Jokiel, P. L., Rodgers, K. S., Brown, E. K., Kenyon, J. C., Aeby, G., Smith, W. R., & Farrell, F. (2015). Comparison of methods used to estimate coral cover in the Hawaiian Islands. PeerJ, 3, e954.

Jonas, H. D., Bingham, H. C., Bennett, N. J., Woodley, S., Zlatanova, R., Howland, E., ... & Ruiz, L. (2024). Global status and emerging contribution of other effective area-based conservation measures (OECMs) towards the '30x30'biodiversity Target 3. Frontiers in Conservation Science, 5, 1447434.

Jones, L., Warner, G., Linton, D., Alcolado, P., Claro-Madruga, R., Clerveaux, W., ... & Wild, R. (2004). Status of coral reefs in the Northern Caribbean and Western Atlantic node of the GCRMN. Australian Institute of Marine Science.

Jordán-Dahlgren, E., Maldonado, M. A., & Rodríguez-Martínez, R. E. (2005). Diseases and partial mortality in Montastraea annularis species complex in reefs with differing environmental conditions (NW Caribbean and Gulf of México). Diseases of Aquatic Organisms, 63(1), 3–12.

Jordán-Dahlgren, E., & Rodríguez-Martínez, R. E. (1998). Post-hurricane initial recovery of Acropora palmata in two reefs of the Yucatán Peninsula, Mexico. Bulletin of Marine Science, 63(1), 213–228.

Joseph, C., Dolique, F., & Saffache, P. (2019). Impacts des activités anthropiques sur les écosystèmes littoraux et marins: les mangroves du Parc National des Trois Baies (Haïti) étudiées à partir d'images LANDSAT. Les Cahiers d'Outre-Mer. Revue de géographie de Bordeaux, 72(279), 69-95.

Jouanno, J., Berthet, S., Muller-Karger, F. et al. (2025b) An extreme North Atlantic Oscillation event drove the pelagic Sargassum tipping point. Commun Earth Environ 6, 95.

Jouanno, J., Almar, R., Muller-Karger, F., Morvan, G., van Tussenbroek, B., Benshila, R., ... & Addo, K. A. (2025a). Socio-ecological vulnerability assessment to Sargassum arrivals. Scientific Reports, 15(1), 9998.

Juhl, R. (1973). Fishery resources of the Caribbean and their potential. Microbial Safety of Fishery Products, 25-40.

Kadison, E., Brandt, M., Nemeth, R., Martens, J., Blondeau, J., & Smith, T. (2017). Abundance of commercially important reef fish indicates different levels of over-exploitation across shelves of the US Virgin Islands. PLoS One, 12(7), e0180063.

Kadison, E., Nemeth, R. S., Blondeau, J., Smith, T, & Calnan. J., (2017). Abundance of commercially important reef fish indicates different levels of ovre-exploitation across shelves of the U.S. Virgin Islands. PLOS ONE, 12 (6), e0180063.

Kaplan, K. A., Ahmadia, G. N., Fox, H., Glew, L., Pomeranz, E. F., & Sullivan, P. (2015). Linking ecological condition to enforcement of marine protected area regulations in the greater Caribbean region. Marine Policy, 62, 186-195.

Kapoor, S., Cantrell, E. M., Peng, K., Pham, T. H., Bail, C. A., Gundersen, O. E., ... & Narayanan, A. (2024). REFORMS: Consensus-based Recommendations for Machine-learning-based Science. Science Advances, 10(18), eadk3452.

Kayal, M., Vercelloni, J., Wand, M. P., & Adjeroud, M. (2015). Searching for the best bet in life-strategy: A quantitative approach to individual performance and population dynamics in reef-building corals. Ecological Complexity, 23, 73-84.

Kemenes van Uden, T., Meesters, E. H., Arango, G., & Meneses, J. (2024). Eastward Bound: charting Bonaire's undescribed underwater ecosystems. Wageningen Marine Research.

Kikaki, A., Karantzalos, K., Power, C. A., & Raitsos, D. E. (2020). Remotely Sensing the Source and Transport of Marine Plastic Debris in Bay Islands of Honduras (Caribbean Sea). Remote Sensing, 12(11), 1727.

Kimura, T., Chou, L. M., Huang, D., Tun, K., and Goh, E. (2022). Status and Trends of East Asian Coral Reefs: 1983–2019. Global Coral Reef Monitoring Network. East Asia Region.

Kitson-Walters, K. (2017). St. Eustatius Global Coral Reef Monitoring Network Final Report 2017. 22p

Kjerfve, B., McField, M., Thattai, D., & Giró, A. (2021). Coral reef health in the Gulf of Honduras in relation to fluvial runoff, hurricanes, and fishing pressure. Marine Pollution Bulletin, 172, 112865.

Kleitz J., Cheng C., Batailler C., Coulon S., (2022). Suivi du réseau de surveillance des masses d'eau côtières du district de la Guadeloupe 2018-2021 au titre de la DCE : Physico-chimie, phytoplancton, communautés coralliennes et herbiers - Année 2022. Rapport OFB-ODE-CREOCEAN : 240 p.

Klokman, O. J., & Hylkema, A. (2024). Spatial distribution of settlement of Diadema antillarum around Saba, Dutch Caribbean. PeerJ, 12, e17291.

Klomp, K. D., & Kooistra, D. J. (2003). A post-hurricane, rapid assessment of reefs in the windward Netherlands Antilles(stony corals, algae and fishes). Atoll Research Bulletin, 496(3), 404-437.

Knapp, K. R., M. C. Kruk, D. H. Levinson, H. J. Diamond, and C. J. Neumann, 2010: The International Best Track Archive for Climate Stewardship (IBTrACS): Unifying tropical cyclone best track data. Bulletin of the American Meteorological Society, 91, 363-376. doi:10.1175/2009BAMS2755.1

Knipp, A. L., Pettijohn, J. C., Jadot, C., & Hertler, H. (2020). Contrasting color loss and restoration in survivors of the 2014–2017 coral bleaching event in the Turks and Caicos Islands. SN Applied Sciences, 2(3), 331.

Knutson, T., Camargo, S. J., Chan, J. C., Emanuel, K., Ho, C. H., Kossin, J., ... & Wu, L. (2020). Tropical cyclones and climate change assessment: Part II: Projected response to anthropogenic warming. Bulletin of the American Meteorological Society, 101(3), E303-E322.

Knutson, T. R., McBride, J. L., Chan, J., Emanuel, K., Holland, G., Landsea, C., ... & Sugi, M. (2010). Tropical cyclones and climate change. Nature geoscience, 3(3), 157-163.

Kobara S, Heyman WD, Pittman SJ, Nemeth RS. (2013). Biogeography of transient reef fish spawning aggregations in the Caribbean: a synthesis for future research and management. Oceanogr Mar Biol Annu Rev. 51:281–324.

Kornder, N. A., Cappelletto, J., Mueller, B., Zalm, M. J., Martinez, S. J., Vermeij, M. J., ... & de Goeij, J. M. (2021). Implications of 2D versus 3D surveys to measure the abundance and composition of benthic coral reef communities. Coral Reefs, 40(4), 1137-1153.

Kossin, J. P., Emanuel, K. A., & Vecchi, G. A. (2014). The poleward migration of the location of tropical cyclone maximum intensity. Nature, 509(7500), 349-352.

Kramer, P., & Dowksza, E. (2005). Coral reefs of Carriacou Island, Grenadines, Grenada: Assessment of condition and key threats. Proceedings of the Carriacou AGRRA training and reef assessment, Carriacou Island, Grenadines, 18-25.

Kramer, P.A. and Kramer, P.R. (ed. M. McField) (2002). Ecoregional Conservation Planning for the Mesoamerican Caribbean Reef. Washington, D.C., World Wildlife Fund.

Kramer, P., Richards Kramer, P., Arias-González, E., & McField, M. (2000). Status of Coral Reefs of Northern Central America: Mexico, Belize, Guatemala, Honduras, Nicaragua and El Salvador. Final Report to the World Bank. UNESCO / Global Coral Reef Monitoring Network.

Kramer, P.A. and Kramer, P.R. (2000). Ecological status of the Mesoamerican Barrier Reef System: impacts of Hurricane Mitch and 1998 coral bleaching. Final report to the World Bank.

Krampitz NM, Ennis RS, Heidmann SL, Hollander EH, Kadison E, Smith TB (2023). The United States Virgin Islands Territorial Coral Reef Monitoring Program. 2023 Annual Report. University of the Virgin Islands, United States Virgin Islands 291pp. University of the Virgin Islands, Saint Thomas, United States Virgin Islands 291.

Kuhn, M., & Wickham, H. (2020). Tidymodels: a collection of packages for modeling and machine learning using tidyverse principles.

Kuhn, M. Johnson, K. (2013). Applied predictive modeling.

Labisky, R. F., Gregory Jr, D. R., & Conti, J. A. (1980). Florida's spiny lobster fishery: an historical perspective. Fisheries, 5(4), 28-37.

Laboy-Nieves, E. N., Klein, E., Conde, J. E., Losada, F., Cruz, J. J., & Bone, D. (2001). Mass mortality of tropical marine communities in Morrocoy, Venezuela. Bulletin of Marine Science, 68(2), 163-179.

Laffoley, D. & Baxter, J.M. (eds.) (2019). Ocean deoxygenation: Everyone's problem - Causes, impacts, consequences and solutions. Full report. Gland, Switzerland: IUCN. 580pp

Laffoley, D. D. A., Roe, H. S., Angel, M. V., Ardron, J., Bates, N. R., Boyd, I. L., ... & Vats, V. (2011). The protection and management of the Sargasso Sea. Sargasso Sea Alliance.

Lamb, J. B., Willis, B. L., Fiorenza, E. A., Couch, C. S., Howard, R., Rader, D. N., ... & Harvell, C. D. (2018). Plastic waste associated with disease on coral reefs. Science, 359(6374), 460-462.

Landsberg, J. H. (1995). Tropical reef-fish disease outbreaks and mass mortalities in Florida, USA: what is the role of dietary biological toxins. Diseases of Aquatic Organisms, 22(2), 83-100.

Lang, J., Alcolado, P., Carricart-Ganivet, J. P., Chiappone, M., Curran, H. A., Dustan, P., ... & Wiener, J. (1998). Status of coral reefs in the northern areas of the wider Caribbean.

Lapointe, B. E., R. Langton, B. J. Bedford, A. C. Potts, O. Day, and C. Hu (2010). Land-based nutrient enrichment of the Buccoo Reef Complex and fringing coral reefs of Tobago, West Indies. Marine Pollution Bulletin 60 (3):334-343.

Lapointe, B. E., R. Langton, O. Day, and A. Potts (2003). Integrated water quality and coral reef monitoring on fringing reefs of Tobago: Chemical and ecological evidence of sewage-driven eutrophication in Buccoo Reef Complex. Proceedings of the 54th Annual Gulf and Caribbean Fisheries Institute:457-472.

Latijnhouwers, K. R. W., van Aalst, M., Huckeba, J., Miller, M. W., Vermeij, M. J. A., & Chamberland, V. F. (2024). The sea urchin Diadema antillarum facilitates recruitment of the Critically Endangered Caribbean coral species Acropora palmata. Marine Ecology Progress Series, 746, 35-48.

Laufkötter, C., Zscheischler, J., & Frölicher, T. L. (2020). High-impact marine heatwaves attributable to human-induced global warming. Science, 369(6511), 1621-1625.

Lawman, A. E., Dee, S. G., DeLong, K. L., & Correa, A. M. S. (2022). Rates of future climate change in the Gulf of Mexico and the Caribbean Sea: Implications for Coral Reef Ecosystems. Journal of Geophysical Research: Biogeosciences, 127(9), e2022JG006999.

Lawrence, J. M. (2007). Edible sea urchins: use and life-history strategies. In Developments in Aquaculture and Fisheries Science (Vol. 37, pp. 1-9). Elsevier.

Laydoo, R. S., et al. (1998). Buccoo Reef and Bon Accord Lagoon, Tobago, Republic of Trinidad and Tobago. CARICOMP coastal region and small island papers. Caribbean coral reef, seagrass and mangrove sites. B. Kjerfve, UNESCO, Paris: 171-176.

Lebreton, L., & Andrady, A. (2019). Future scenarios of global plastic waste generation and disposal. Palgrave Communications, 5(1), 1-11.

Lee Hing, C., Guifarra, Z., Damaris, D., Ochoa, G., Nunez, A., Forman,K., Craig, N., McField, M. (2022). Management responses in Belize and Honduras, as stony coral tissue loss disease expands its prevalence in the Mesoamerican Reef. V9. P.1-10. Frontiers in Marine Science.

Leichter, J. J., Helmuth, B., & Fischer, A. M. (2006). Variation beneath the surface: quantifying complex thermal environments on coral reefs in the Caribbean, Bahamas and Florida.

Leotaud, N., Laidlow-Ferdinand, A., Clauzel, S., & Vaughan, N. (2024). Towards sustainable blue tourism in the Caribbean: Policy pathways to support community-based coastal and marine tourism. Blue Tourism Initiative Diagnostic Study. CANARI: Port of Spain.

Lesser, M. P. (2021). Eutrophication on coral reefs: what is the evidence for phase shifts, nutrient limitation and coral bleaching. BioScience, 71(12), 1216-1233.

Lessios, H. A. (2016). The great Diadema antillarum die-off: 30 years later. Annual review of marine science, 8(1), 267-283.

Lessios, H. A. (1988). Mass mortality of Diadema antillarum in the Caribbean: What have we learned? Annual Review of Ecology and Systematics, 19, 371–393.

Lessios, H. A., Cubit, J. D., Robertson, D. R., Shulman, M. J., Parker, M. R., Garrity, S. D., & Levings, S. C. (1984). Mass mortality of Diadema antillarum on the Caribbean coast of Panama. Coral Reefs, 3(4), 173-182.

Lessios, H. A., Robertson, D. R., & Cubit, J. D. (1984). Spread of Diadema mass mortality through the Caribbean. Science, 226(4672), 335-337.

Leverington, F., Lemos Costa, K., Courrau, J., Pavese, H., Nolte, C., Marr, M., Coad, L., Burgess, N. D., Bomhard, B., & Hockings, M. (2010). Management effectiveness evaluation in protected areas: a global study – second edition. University of Queensland / IUCN WCPA.

Levin, L. A. (2018). Manifestation, drivers, and emergence of open ocean deoxygenation. Annual review of marine science, 10(1), 229-260.

Levitan, D. R., Best, R. M., & Edmunds, P. J. (2023). Sea urchin mass mortalities 40 y apart further threaten Caribbean coral reefs. Proceedings of the National Academy of Sciences, 120(10), e2218901120.

Levitan, D. R., Boudreau, W., Jara, J., & Knowlton, N. (2014). Long-term reduced spawning in Orbicella coral species due to temperature stress. Marine Ecology Progress Series, 515, 1-10.

Lewis, J. (2002). Evidence from aerial photography of structural loss of coral reefs at Barbados, West Indies. Coral Reefs, 21(1), 49-56.

Ley-Cooper, K., & Chavez, E. A. (2009). Bio-economic modelling applied to a spiny lobster fishery of the Northwestern Caribbean. Department of Environmental and Aquatic Sciences Curtin University.

Li, W., Bai, X., Yang, D., & Hou, Y. (2023). Maritime connectivity, transport infrastructure expansion and economic growth: A global perspective. Transportation Research Part A: Policy and Practice, 170, 103609.

Liddell, W. D., & Ohlhorst, S. L. (1986). Changes in benthic community composition following the mass mortality of Diadema at Jamaica. Journal of Experimental Marine Biology and Ecology, 95(3), 271-278.

Lirman, D. (2001). Competition between macroalgae and corals: effects of herbivore exclusion and increased algal biomass on coral survivorship and growth. Coral reefs, 19(4), 392-399.

Littler, M. M., & Littler, D. S. (2013). The nature of crustose coralline algae and their interactions on reefs.

Lizama, D., & Belize Audubon Society. (2023). Addressing the Onset of Stony Coral Tissue Loss Disease in Lighthouse Reef Atoll, Belize. MAR Fund.

Logan, A. (2013). Coral reefs of the Cayman Islands. In Coral Reefs of the United Kingdom Overseas Territories (pp. 61-68). Dordrecht: Springer Netherlands.

Logan, A., & Sealey, K. S. (2013). The reefs of the Turks and Caicos Islands. In Coral reefs of the United Kingdom overseas territories (pp. 97-114). Dordrecht: Springer Netherlands.

Loh, T. L., McMurray, S. E., Henkel, T. P., Vicente, J., & Pawlik, J. R. (2015). Indirect effects of overfishing on Caribbean reefs: sponges overgrow reef-building corals. PeerJ, 3, e901.

Lucas, M. Q., Collazo Roman, D. L., Mercado, M. A., Fain, E. J., Toledo-Rodríguez, D. A., & Weil, E. (2024). Stony coral tissue loss disease (SCTLD) induced mass mortality at Arecibo, Puerto Rico. Marine Biodiversity, 54(1), 10.

Luckhurst, B. E. and S. O'Farrell (2014). Rapid recovery of parrotfish (Scaridae) and Surgeonfish (Acanthuridae) populations following the fish pot ban in Bermuda. Proceedings of the 66th Gulf and Caribbean Fisheries Institute. 66:301-306.

Lyons, M. B., Murray, N. J., Kennedy, E. V., Kovacs, E. M., Castro-Sanguino, C., Phinn, S. R., ... & Roelfsema, C. M. (2024). New global area estimates for coral reefs from high-resolution mapping. Cell Reports Sustainability, 1(2).

López, J. C. R., & Torregroza, A. C. (2017). Suspended sediment load in northwestern South America (Colombia): A new view on variability and fluxes into the Caribbean Sea. Journal of South American Earth Sciences, 80, 340-352.

López-Londoño, T., Galindo-Martínez, C. T., Gómez-Campo, K., González-Guerrero, L. A., Roitman, S., Pollock, F. J., Pizarro, V., López-Victoria, M., Medina, M., & Iglesias-Prieto, R. (2021). Physiological and ecological consequences of the water optical properties degradation on reef corals. Coral Reefs, 40(4), 1243-1256.

MacRae, D., & De Meyer, K. (2020). A new approach to monitoring marine protected area management success in the Dutch Caribbean. In Marine Protected Areas (pp. 379-409). Elsevier.

MacRae, D. (2011) Man of war shoal marine park management plan 2011. Nature Foundation St. Maarten. 112 p.

Madibenthos (2016). Compte-rendu de la campagne d'inventaire de la faune et de la flore marine côtière de la Martinique, expédition Madibenthos 2016. Muséum National d'Histoire Naturelle. 233 pp.

Madin, J. S., Baird, A. H., Dornelas, M., & Connolly, S. R. (2014). Mechanical vulnerability explains size-dependent mortality of reef corals. Ecology letters, 17(8), 1008-1015.

Mah, A. J., & Stearn, C. W. (1986). The effect of Hurricane Allen on the Bellairs fringing reef, Barbados. Coral Reefs, 4(3), 169-176.

Maishal, S. Rising Global Ocean Deoxygenation: Delving into Drivers. Available at SSRN 4881714.

Malawauw, R. J., Piaskowy, J., ter Horst, L. J., Calhoun, D. M., & Johnson, P. T. (2024). Parasitism in reef fish communities: evaluating the roles of host traits, habitat use, and phylogeny on infection by Scaphanocephalus (Trematoda). Coral Reefs, 43(3), 509-522.

Mallela, J., Hetzinger, S., Halfar, J. (2016). "Thermal stress markers in Colpophyllia natans provide an archive of site-specific bleaching events." Coral Reefs 35(1): 181-186.

Mallela, J., Parkinson, R., Day, O. (2010). "An assessment of coral reefs in Tobago." Caribbean Journal of Science 46(1): 83-87.

Mallela, J. and M. J. C. Crabbe (2009). "Hurricanes and coral bleaching linked to changes in coral recruitment in Tobago." Marine Environmental Research 68: 158–162.

Mallela, J. and Harrod, C. (2008). "□13C and □15N reveal significant differences in the coastal foodwebs of the seas surrounding the islands of Trinidad and Tobago." Marine Ecology Progress Series 368: 41-51.

Mallela, J. and R. Parkinson (2008). "Coral disease succession in Tobago: from yellow band to black band disease." Coral Reefs 27: 787.

Mallon, J. E., Altieri, A. H., Cyronak, T., Melendez-Declet, C. V., Valerie, P. J., & Johnson, M. D. (2025). Sublethal changes to coral metabolism in response to deoxygenation. Journal of Experimental Biology, jeb-249638.

Manfrino, C., Jacoby, C. A., Camp, E., & Frazer, T. K. (2013). A positive trajectory for corals at Little Cayman Island. PLoS One, 8(10), e75432.

Mansoor, U., & Inam, S. (2025). Climate Change, Carbon Emissions, and Legal Vulnerability: The Case of Small Island States. IUP Law Review, 15(1).

Manuel, O. S., Williams, S. M., Weil, E., & Cruz-Motta, J. J. (2021). Experimental evaluation of Diadema antillarum herbivory effects on benthic community assemblages. Journal of Experimental Marine Biology and Ecology, 541, 151566.

Manzello, D. P., Cunning, R., Karp, R. F., Baker, A. C., Bartels, E., Bonhag, R., ... & Williamson, O. M. (2025). Heat-driven functional extinction of Caribbean Acropora corals from Florida's Coral Reef. Science, 390(6771), 361-366.

Manzello, D. P., Brandt, M., Smith, T. B., Lirman, D., Hendee, J. C., & Nemeth, R. S. (2007). Hurricanes benefit bleached corals. Proceedings of the National Academy of Sciences, 104(29), 12035-12039.

Mariño-Briceño, G., Cappelletto, J., Ascanio, A., Agudo-Adriani, E., & Cróquer, A. (2024). Describing the dynamics of recruits and juvenile scleractinian corals using 3D models: a case study from Cayo Mero reef, Morrocoy National Park, Venezuela. Novitates Caribaea, (23), 51-73.

Mariño-Tapia, I., Enriquez, C., Silva, R., Mendoza-Baldwin, E., Escalante-Mancera, E., & Ruiz-Renteria, F. (2014). Comparative morphodynamics between exposed and reef protected beaches under hurricane conditions. Coast. Eng. Proc, 1(55).

Marsh, R., Skliris, N., Tompkins, E. L., Dash, J., Dominguez Almela, V., Tonon, T., ... & Webber, M. (2023). Climate-sargassum interactions across scales in the tropical Atlantic. Plos Climate, 2(7), e0000253.

Marshall, S., & Koon, R. K. (2021). Barbados towards 100% Renewable Energy: Case Scenarios for 2030 National Energy Target Plans. West Indian Journal of Engineering, 11-17.

Martinez, S. J., Cavada-Blanco, F., Cappelletto, J., Agudo-Adriani, E., & Cróquer A. (2021). Distribution, abundance, and health indicators of the critically endangered coral species Acropora cervicornis in Los Roques National Park, 2014. Advances in Oceanography and Limnology, 12(2).

Martín-Arias, V., Evans, C., Griffin, R., Cherrington, E. A., Lee, C. M., Mishra, D. R., ... & Rosado, S. (2022). Modeled impacts of LULC and climate change predictions on the hydrologic regime in Belize. Frontiers in Environmental Science, 10, 848085.

Maréchal J.-P., and Trégarot, E. (2025). Suivi de l'état écologique des communautés benthiques et des peuplements de poissons de la Martinique. Campagne 2025. Rapport NBE. 123p + Annexes.

Maréchal, J.-P. and Trégarot, E. (2019). Suivi de l'état écologique des communautés benthiques et des peuplements de poissons de la Martinique. campagne 2019. Rapport NBE. 59p.

Mateos-Jasso, A., Zavala-Hidalgo, J., Romero-Centeno,R., Allende-Arandı´a, M.E., (2012). Variability of the thermohaline structure in the northern Veracruz Coral Reef System, Mexico. Continental Shelf Research 50–51, 30–40

Mathon, L. (2025). Marine and Coastal Biodiversity Assessment of Sint Maarten. Nature Foundation St. Maarten. 178 p.

Mathon, L. (2024). Coral reef monitoring report of Sint Maarten (2016 - 2024). Nature Foundation St. Maarten. 84 p.

Maynard, J., Van Hooidonk, R., Eakin, C. M., Puotinen, M., Garren, M., Williams, G., ... & Harvell, C. D. (2015). Projections of climate conditions that increase coral disease susceptibility and pathogen abundance and virulence. Nature Climate Change, 5(7), 688-694.

McClanahan, T. R., Friedlander, A. M., Chabanet, P., Bruggemann, J. H., Wickel, J., & Azali, M. K. (2024). Modeling the spatial distribution of numbers of coral reef fish species and community types in the Western Indian Ocean faunal province. Marine Ecology Progress Series, 730, 59-78.

McClanahan, T. R., Sala, E., Stickels, P. A., Cokos, B. A., Baker, A. C., Starger, C. J., & Jones Iv, S. H. (2003). Interaction between nutrients and herbivory in controlling algal communities and coral condition on Glover's Reef, Belize. Marine Ecology Progress Series, 261, 135-147.

McClenachan, L. (2009). Documenting loss of large trophy fish from the Florida Keys with historical photographs. Conservation biology, 23(3), 636-643.

McConney, P., Charles, A., & Mahon, R. (2015). Governance of Caribbean coastal fisheries. Marine Policy, 51, 63–71.

McField, M. E., Bood, N. A., Fonseca, A., Arrivillaga, A. L., Franquesa Rinos, A., Loreto Viruel, R. M. (2008). Status of the Mesoamerican Reef after the 2005 coral bleaching event. In: Status of Caribbean coral reefs after bleaching and hurricanes in 2005, eds. Wilkinson C, Souter D.

McField, M. Soto, M., Martinez, R. Giró, A. Guerrero,C., Rueda, M., Kramer,P., Roth,L., Muñiz, I. (2024). 2024 Mesoamerican Reef Report Card. Healthy Reefs for Healthy People. www.healthyreefs.org.

McField, M., Kramer, P., Giró Petersen, A., Soto, M., Drysdale, I., Craig, N., & Rueda-Flores, M. (2022). Mesoamerican reef report card. Healthy reefs initiative.

McField, M., Kramer, P., Giró Petersen, A., Soto, M., Drysdale, I., Craig, N., & Rueda-Flores, M. (2020). Mesoamerican reef report card. Healthy reefs initiative.

McField, M.D. (2002). Influence of Disturbance on coral reef community structure in Belize. Proceedings of the Ninth International Coral Reef Symposium, Bali, Oct. 2000. Vol. 1: 63-68.

McField, M.D. (1999). Coral response during and after mass bleaching in Belize. Bull. Mar. Sci. 64(1): 155-172

McManus, L. C., Forrest, D. L., Tekwa, E. W., Schindler, D. E., Colton, M. A., Webster, M. M., ... & Pinsky, M. L. (2021). Evolution and connectivity influence the persistence and recovery of coral reefs under climate change in the Caribbean, Southwest Pacific, and Coral Triangle. Global change biology, 27(18), 4307-4321.

McWilliam, M., Hoogenboom, M. O., Baird, A. H., Kuo, C. Y., Madin, J. S., & Hughes, T. P. (2018). Biogeographical disparity in the functional diversity and redundancy of corals. Proceedings of the National Academy of Sciences, 115(12), 3084-3089.

Medina-Valmaseda AE, Blanchon P, Alvarez-Filip L, Pérez-Cervantes E. (2022). Geomorphically controlled coral distribution in degraded shallow reefs of the Western Caribbean. PeerJ 10:e12590.

Meesters, E. H., van der Geest, M., van Uden, T. K., Boman, E., Butler, E., Hylkema, A., ... & Francisca, R. (2025). Conservation state of coral reefs and communities of the Caribbean Netherlands.

Meesters, E.H. & Bak, R.P.M. (1993). Effects of Coral Bleaching on Tissue Regeneration Potential and Colony Survival. Mar. Ecol. Prog. Ser., 96, 189-198.

Melendez, M., & Salisbury, J. (2017). Impacts of ocean acidification in the coastal and marine environments of Caribbean small island developing states (SIDS). Caribbean Mar. Climate Change Rep. Card Sci. Rev, 2017, 31-30.

Mendoza, J. J., & Larez, A. (2004). A biomass dynamics assessment of the southeastern Caribbean snapper–grouper fishery. Fisheries Research, 66(2-3), 129-144.

Mengo, E., Grilli, G., Luisetti, T., Watt, H. C., Jones, C. H., & Posen, P. (2022). Marine and coastal accounts for Small Island Developing States: A case study and application in Grenada. One Ecosystem, 7, e84865.

Mesquita, Y. W., Mengatto, M. F., & Nagai, R. H. (2022). Where and how? A systematic review of microplastic pollution on beaches in Latin America and the caribbean (LAC). Environmental Pollution, 314, 120231.

Mezger, S. D., Klinke, A., Tilstra, A., El-Khaled, Y. C., Thobor, B., & Wild, C. (2022). The widely distributed soft coral Xenia umbellata exhibits high resistance against phosphate enrichment and temperature increase. Scientific Reports, 12(1), 22135.

Miller, M. W., Mendoza Quiroz, S., Lachs, L., Banaszak, A. T., Chamberland, V. F., Guest, J. R., ... & Petersen, D. (2024). Assisted sexual coral recruits show high thermal tolerance to the 2023 Caribbean mass bleaching event. PloS one, 19(9), e0309719.

Miller, A. W., & Richardson, L. L. (2015). Emerging coral diseases: a temperature-driven process?. Marine Ecology, 36(3), 278-291.

Miller, J., Muller, E., Rogers, C., Waara, R., Atkinson, A., Whelan, K. R. T., ... & Witcher, B. (2009). Coral disease following massive bleaching in 2005 causes 60% decline in coral cover on reefs in the US Virgin Islands. Coral Reefs, 28(4), 925-937.

Miloslavich P, Díaz JM, Klein E, Alvarado JJ, Díaz C, Gobin J, et al. (2010). Marine Biodiversity in the Caribbean: Regional Estimates and Distribution Patterns. PLoS ONE 5(8): e11916.

Miloslavich, P., Klein, E., Yerena, E., & Martin, A. (2003). Marine biodiversity in Venezuela: status and perspectives Biodiversidad marina en Venezuela: Estado actual y perspectivas. Gayana, 67(2), 275-301.

Ministerio de Economía, Planificación y Desarrollo (MEPyD), Oficina Nacional de Estadística (ONE), & Consejo Dominicano de Pesca y Acuicultura (CODOPESCA). (2020). I Censo Nacional Pesquero 2019: Informe general. Santo Domingo, República Dominicana.

Mitchell, A. (2018). Coral reef monitoring in St Maarten 2016 and 2017 Impact of Hurricane Irma on St Maarten Reefs and on the Man of War Shoal Marine Protected Area. Nature Foundation St. Maarten. 28p.

Mittempergher, D., Raes, L., Jain, A., Mittempergher, D., Raes, L., & Jain, A. (2022). The economic impact of marine plastic pollution in Antigua and Barbuda: Impacts on the fisheries and tourism sectors, and the benefits of reducing mismanaged waste. North American Journal of Fisheries Management, 31, 880-893.

Miyazawa, E., Montilla, L. M., Agudo-Adriani, E. A., Ascanio, A., Mariño-Briceño, G., & Croquer, A. (2020). On the importance of spatial scales on beta diversity of coral assemblages: a study from Venezuelan coral reefs. PeerJ, 8, e9082.

Mohammed, E. (2021). Quantifying marine biodiversity changes in the southeastern Caribbean.

Mohammed, E., Straker, L. E., & Jardine, C. (2003). St. Vincent and the Grenadines: Reconstructed fisheries catches and fishing effort, 1942-2001. Fisheries Centre Research Reports, 11(6), 95-116.

Molina-Hernández, A., & Álvarez-Filip, L. (2024). Incorporating parrotfish bioerosion into the herbivory paradigm of coral reef resilience. Conservation Letters, 17(6), e13058.

Monaco, M. E., Friedlander, A. M., Caldow, C., Christensen, J. D., Rogers, C., Beets, J., ... & Boulon, R. (2007). Characterising reef fish populations and habitats within and outside the US Virgin Islands Coral Reef National Monument: a lesson in marine protected area design. Fisheries Management and Ecology, 14(1), 33-40.

Montilla, L. M., Miyazawa, E., Verde, A., Agudo-Adriani, E., Ascanio, A., Cappelletto, J., López-Hernández, M., Mariño-Briceño, G., Martínez, S., Rebolledo-Sánchez, Z., Rivera, A., Mancilla, D. S., & Cróquer, A. (2021). A photographic data set of reef and coral communities across Venezuela. Data in Brief, 37, 107235.

Mora, C. (2008). A clear human footprint in the coral reefs of the Caribbean. Proceedings of the Royal Society B: Biological Sciences, 275(1636), 767-773.

Moritz, C., Vii, J., Long, W. L., Tamelander, J., Thomassin, A., and Planes, S. (2018). Status and trends of coral reefs of the Pacific.

Morris, J. T., Enochs, I. C., Besemer, N., Viehman, T. S., Groves, S. H., Blondeau, J., ... and Manzello, D. P. (2022). Low net carbonate accretion characterizes Florida's coral reef. Scientific Reports, 12(1), 19582.

Morris Jr, J.A. and Akins, J.L. (2009). Feeding ecology of invasive lionfish (Pterois volitans) in the Bahamian archipelago. Environmental Biology of Fishes, 86(3), pp.389-398.

Morris, A., Chormanski, D., & Die, D. (2004). Overview of the assessment history and the current management for the fishery of spiny lobster, Panulirus argus, in the US Caribbean. SEDAR 8-DW-03, available online at http://www. sefsc. noaa. gov/sedar/download/S8DW_03. pdf.

Muller, E. M., & van Woesik, R. (2014). Genetic susceptibility, colony size, and water temperature drive white-pox disease on the coral Acropora palmata. PLoS One, 9(11), e110759.

Muller, E. M., Rogers, C. S., Spitzack, A. S., & Van Woesik, R. (2008). Bleaching increases likelihood of disease on Acropora palmata (Lamarck) in Hawksnest Bay, St John, US virgin islands. Coral Reefs, 27(1), 191-195.

Mumby, P. J., Steneck, R. S., Roff, G., & Paul, V. J. (2021). Marine reserves, fisheries ban, and 20 years of positive change in a coral reef ecosystem. Conservation Biology, 35(5), 1473-1483.

Mumby, P. J., Steneck, R. S., Edwards, A. J., Ferrari, R., Coleman, R., Harborne, A. R., & Gibson, J. P. (2012). Fishing down a Caribbean food web relaxes trophic cascades. Marine Ecology Progress Series, 445, 13-24.

Mumby, P. J., Hastings, A., & Edwards, H. J. (2007). Thresholds and the resilience of Caribbean coral reefs. Nature, 450(7166), 98-101.

Mumby, P. J. (2006). The impact of exploiting grazers (Scaridae) on the dynamics of Caribbean coral reefs. Ecological applications, 16(2), 747-769.

Mumby, P. J., Green, E. P., Clark, C. D., & Edwards, A. J. (1998). Digital analysis of multispectral airborne imagery of coral reefs. Coral Reefs, 17(1), 59–69.

Muraoka, W. T., Cramer, K. L., O'Dea, A., Zhao, J. X., Leonard, N. D., & Norris, R. D. (2022). Historical declines in parrotfish on Belizean coral reefs linked to shifts in reef exploitation following European colonization. Frontiers in Ecology and Evolution, 10, 972172.

Muñiz-Castillo, A. I., Rivera-Sosa, A., McField, M., et al. (2024). Underlying drivers of coral reef vulnerability to bleaching in the Mesoamerican Reef. Communications Biology, 7(1), Article 1452.

Muñiz-Castillo, A. I., Rivera-Sosa, A., Chollett, I., Eakin, C. M., Andrade-Gómez, L., McField, M., & Arias-González, J. E. (2019). Three decades of heat stress exposure in Caribbean coral reefs: a new regional delineation to enhance conservation. Scientific reports, 9(1), 11013.

Muñoz-Nuñez, D. (2009). The Caribbean spiny lobster fishery in Cuba: An approach to sustainable fishery management.

Myers, A. (2013). Coral reefs of Montserrat. In C. R. C. Sheppard (Ed.), Coral reefs of the United Kingdom overseas territories, coral reefs of the world (pp. 89–96). Dordrecht: Springer.

Nama, S., Shanmughan, A., Nayak, B. B., Bhushan, S., & Ramteke, K. (2023). Impacts of marine debris on coral reef ecosystem: A review for conservation and ecological monitoring of the coral reef ecosystem. Marine Pollution Bulletin, 189, 114755.

National Park Authority (NPA). (2023). Coral Bleaching Report.

Natural Earth (2024b). Cross-Blended Hypsometric Tints v. 6.0. Retrieved from https://shadedrelief.com/ne-draft/#Raster-Data. Accessed on 2024-07-30.

Natural Earth (2024a). Admin 0 - Boundary lines (land boundaries). Retrieved from https://www.naturalearthdata.com/downloads/10m-cultural-vectors/10m-admin-0-boundary-lines/. Accessed on 2024-07-30.

Navas-Camacho, Raúl, Rodríguez-Ramírez, Alberto, & Reyes-Nivia, MaríaCatalina. (2010). Agents of coral mortality on reef formations of the Colombian Pacific. Revista de Biología Tropical, 58(Suppl. 1), 133-138.

Neely, K. L., Shea, C. P., Macaulay, K. A., Hower, E. K., & Dobler, M. A. (2021). Short-and long-term effectiveness of coral disease treatments. Frontiers in Marine Science, 8, 675349.

Nemeth, R. S., Kadison E., Heidmann S. and Smith T. B. (in review). Road to recovery for the endangered Nassau grouper in the US Virgin Islands. GCRMN report: Status and Trends of Caribbean Coral Reefs: 1980-2024.

Nemeth, R. S., Kadison, E., Jossart, J., Shivji, M., Wetherbee, B. M., & Matley, J. K. (2023). Acoustic telemetry provides insights for improving conservation and management at a spawning aggregation site of the endangered Nassau grouper (Epinephelus striatus). Frontiers in Marine Science, 10, 1154689.

Nemeth, R.S., S. Herzlieb, J. Blondeau (2006). Comparison of two seasonal closures for protecting red hind spawning aggregations in the US Virgin Islands. Proc. 10th International Coral Reef Conference, Okinawa, Japan. 4:1306-1313.

Nemeth, R. S. (2005). Population characteristics of a recovering US Virgin Islands red hind spawning aggregation following protection. Marine Ecology Progress Series 286:81-97.

NFSXM (2010b). Results of Nature Foundation Baseline Coral Reef Ecosystem Monitoring: Islets Eastern Shore. Nature Foundation Sint Maarten

Nijkamp, H., Djohani, R., & Meesters, E. (1995). The Marine Environment of Sint Maarten. AIDEnvironment EcoVision

NOAA (2025). Bleaching Alert Area Annual Maximum. Coral Reef Watch https://coralreefwatch.noaa.gov/data/5km/v3.1_op/image/composite/annual/baa5-max/png/ct5km_baa5-max_v3.1_caribbean_2023.png

NOAA (2020). Coral reef condition: A status report for the Flower Garden Banks. National Oceanic and Atmospheric Administration. Coral Reef Conservation Program (CRCP), 2020.

NOAA National Centers for Environmental Information. (2024). NOAA National Centers for Coastal Ocean Science (2018). National Coral Reef Monitoring Program: Assessment of coral reef benthic communities in the U.S. Virgin Islands [Dataset]. https://doi.org/10.7289/v5ww7fqk (Accessed 2024-11-01).

NOAA National Centers for Environmental Information. (2024). NOAA National Centers for Coastal Ocean Science; NOAA Southeast Fisheries Science Center (2018). National Coral Reef Monitoring Program: Assessment of coral reef benthic communities in Puerto Rico [Dataset]. https://doi.org/10.7289/v5pg1q23 (Accessed 2024-11-01).

NOAA National Centers for Environmental Information. (2024). NOAA Southeast Fisheries Science Center; NOAA National Centers for Coastal Ocean Science (2018). National Coral Reef Monitoring Program: Assessment of coral reef benthic communities in the Florida Reef Tract [Dataset]. https://doi.org/10.7289/v5xw4h4z (Accessed 2024-11-01).

NOAA National Centers for Environmental Information (2023). NOAA National Centers for Coastal Ocean Science; NOAA Southeast Fisheries Science Center (2016). National Coral Reef Monitoring Program: Assessment of coral reef benthic communities in Flower Garden Banks National Marine Sanctuary [Dataset]. https://doi.org/10.7289/v5vd6wts (Accessed 2024-11-01).

Noble, M. M., van Laake, G., Berumen, M. L., & Fulton, C. J. (2013). Community change within a Caribbean coral reef marine protected area following two decades of local management. PloS one, 8(1), e54069.

Nowlis, J. S., Roberts, C., Smith, A., & Siirila, E. (1997). Human-enhanced impacts of a tropical storm on nearshore coral reefs. AMBIO: A Journal of the Human Environment, 26(8), 515-521.

Nugues, M. M. (2002). Impact of a coral disease outbreak on coral communities in St. Lucia: What and how much has been lost?. Marine Ecology Progress Series, 229, 61-71.

Nugues, M. M., & Roberts, C. M. (2003). Coral mortality and interaction with algae in relation to sedimentation. Coral reefs, 22(4), 507-516.

Nunes, B. Z., Zanardi-Lamardo, E., Choueri, R. B., & Castro, Í. B. (2021). Marine protected areas in Latin America and Caribbean threatened by polycyclic aromatic hydrocarbons. Environmental Pollution, 269, 116194.

O Debrot, A., Yranzo, A., & Arocha, D. (2019). Los roques and las aves archipelagos, Venezuela: a marine ecological and conservation reconnaissance of two little-known southeastern Caribbean oceanic archipelagos. Smithsonian Institution Scholarly Press.

O'Brien, C. E., Johnston, M. W., & Kerstetter, D. W. (2017). Ports and pests: assessing the threat of aquatic invasive species introduced by maritime shipping activity in Cuba. Marine pollution bulletin, 125(1-2), 92-102.

Obura, D., Gudka, M., Rabi, F. A., Gian, S. B., Bijoux, J., Freed, S., ... & Ahamada, S. (2017). Coral reef status report for the Western Indian Ocean (2017). In Nairobi convention. Global Coral Reef Monitoring Network (GCRMN)/ International Coral Reef Initiative (ICRI).

Office de l'Eau Martinique & Impact Mer (2022). Coral reefs benthic data from the Directive Cadre sur l'Eau [Dataset].

Office of National Marine Sanctuaries, National Ocean Service, National Oceanic and Atmospheric Administration, and U.S. Department of Commerce (2021). Flower Garden Banks National Marine Sanctuary Boundary (FGBNMS_py) and Florida Keys National Marine Sanctuary Boundary (FKNMS_py). Retrieved from https://sanctuaries.noaa.gov/library/imast_gis.html. Accessed on 2024-08-01.

Olguín-Maciel, E., Leal-Bautista, R.M., Alzate-Gaviria, L., Domínguez-Maldonada, J., Tapia-Tusell, R. (2022). Environmental impact of Sargassum spp. landings: an evaluation of leachate released from natural decomposition at Mexican Caribbean coast. Environ. Sci. Pollut. Control Ser. 29, 91071–91080.

Olsen, D. A. and LaPlace J. A. (1978). A study of Virgin Islands grouper fishery based on a breeding aggregation. Proceedings of the Gulf Caribbean Fisheries Institute. 31:130–144.

ONMS (2024). Office of National Marine Sanctuaries. Flower Garden Banks National Marine Sanctuary condition report: 2009–2021. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service.

Oribe-Pérez, I., Velázquez-Abunader, I., & Monroy-García, C. (2023). Factors affecting the relative abundance in an overfished stock: red grouper (Epinephelus morio) in the Southeastern Gulf of Mexico. PeerJ, 11, e16490.

Orr, M. C., Hughes, A. C., Costello, M. J., and Qiao, H. (2022). Biodiversity data synthesis is critical for realizing a functional post-2020 framework. Biological Conservation, 274, 109735.

Ortega-Ariza, D., Franseen, E. K., & Boudagher-Fadel, M. K. (2021). Effects of sea level and upwelling on development of a Miocene shallow-water tropical carbonate ramp system, Ponce, Puerto Rico. Journal of Sedimentary Research, 91(11), 1227-1256.

Ortíz, J.C. (2002). Estructura de la comunidad de corales pétreos de la Ensenada de Chichiriviche de la Costa y efecto de un período intenso de precipitación. Trabajo Especial de Grado, Universidad Central de Venezuela, Caracas, Venezuela.

Otaño-Cruz, A., Montañez-Acuña, A. A., García-Rodríguez, N. M., Díaz-Morales, D. M., Benson, E., Cuevas, E., ... & Hernández-Delgado, E. A. (2019). Caribbean near-shore coral reef benthic community response to changes on sedimentation dynamics and environmental conditions. Frontiers in Marine Science, 6, 551.

Oxenford, H. A., Roach, R., Brathwaite, A., Nurse, L., Goodridge, R., Hinds, F., ... & Finney, C. (2008). Quantitative observations of a major coral bleaching event in Barbados, Southeastern Caribbean. Climatic Change, 87(3), 435-449.

Paddack, M. J., Reynolds, J. D., Aguilar, C., Appeldoorn, R. S., Beets, J., Burkett, E. W., ... & Côté, I. M. (2009). Recent region-wide declines in Caribbean reef fish abundance. Current Biology, 19(7), 590-595.

Pallis, T. (2015). Cruise shipping and urban development: State of the art of the industry and cruise ports. International Transport Forum Discussion Paper.

Pandolfi, J. M., Bradbury, R. H., Sala, E., Hughes, T. P., Bjorndal, K. A., Cooke, R. G., ... & Jackson, J. B. (2003). Global trajectories of the long-term decline of coral reef ecosystems. Science, 301(5635), 955-958.

Papke, E., Carreiro, A., Dennison, C., Deutsch, J. M., Isma, L. M., Meiling, S. S., ... & Ushijima, B. (2024). Stony coral tissue loss disease: a review of emergence, impacts, etiology, diagnostics, and intervention. Frontiers in Marine Science, 10, 1321271.

PARETO (2012). Suivi de l'état de santé des réserves naturelles marines de Guadeloupe, de Saint-Martin et Saint- Barthélemy. Etat des lieux 2012 et évolution 2007-2012. Décembre 2012, 62 pages + annexes.

Pascal N., Leport G., Allenbach M., Marchand C., (2016). Valeur économique des services rendus par les récifs coralliens et écosystèmes associés des Outre-mer français. Rapport technique IFRECOR, 56 pages.

Pattengill-Semmens, C. V., & Semmens, B. X. (2003). Status of coral reefs of Little Cayman and Grand Cayman, British West Indies, in 1999(Part 2: Fishes). Atoll Research Bulletin, 496, 226-247.

Patterson, K. L., Porter, J. W., Ritchie, K. B., Polson, S. W., Mueller, E., Peters, E. C., ... & Smith, G. W. (2002). The etiology of white pox, a lethal disease of the Caribbean elkhorn coral, Acropora palmata. Proceedings of the National Academy of Sciences, 99(13), 8725-8730.

Pebesma, E., & Bivand, R. (2023). Spatial data science: With applications in R. Chapman and Hall/CRC.

Peiffer, F., Bejarano, S., de Witte, G. P., & Wild, C. (2017). Ongoing removals of invasive lionfish in Honduras and their effect on native Caribbean prey fishes. PeerJ, 5, e3818.

Pellerin, F., Espel, D., Ballard, J., Salaün, J., Kayal, E., Zubia, M., ... & Kayal, M. (2025). Snapshots of coral reef biodiversity. In The Future of Coral Reefs: Evidence from Research (pp. 187-241). Cham: Springer Nature Switzerland.

Pelose, G., Blanco-Pimentel, M., Calle-Triviño, J., León, A., Galván, V., Schopmeyer, S., Foster, D., Harms, E., Burdett, C., & Morikawa, M. K. (2024). Novel ex-situ Stony Coral Tissue Loss Disease treatment protocol. protocols.io.

Pepe, B. A., Van der Roest, V., Vlam, O., Nouse, S., & Meesters, E. (2024). Assessing the impact of stony coral tissue loss disease on coral cover on Bonaire's Leeward side. Frontiers in Marine Science, 11, 1512371.

Perera-Valderrama, S., Hernández-Arana, H., Ruiz-Zárate, M. Á., Alcolado, P. M., Caballero-Aragón, H., González-Cano, J., ... & Cobián-Rojas, D. (2016). Condition assessment of coral reefs of two marine protected areas under different regimes of use in the north-western Caribbean. Ocean & Coastal Management, 127, 16-25.

Perez, V. B., & Tewfik, A. (2016). Brief history of management and conservation of Nassau grouper and their spawning aggregations in Belize: a collaborative approach. In Proc. Gulf Caribb. Fish. Inst (Vol. 68, pp. 118-122).

Perry, C. T., de Bakker, D. M., Webb, A. E., Comeau, S., Harvey, B. P., Cornwall, C. E., ... & Precht, W. F. (2025). Reduced Atlantic reef growth past 2° C warming amplifies sea-level impacts. Nature, 1-8.

Peterson, R. R., DiPietro, R. B., & Harrill, R. (2020). In search of inclusive tourism in the Caribbean: insights from Aruba. Worldwide Hospitality and Tourism Themes, 12(3), 225-243.

Petit, R., & Allsopp, M. (2023). The relative influence of sea surface temperature anomalies on the benthic composition of an Indo-Pacific and Caribbean coral reef over the last decade. Ecology & Evolution.

Pezner, A. K., Courtney, T. A., Barkley, H. C., Chou, W. C., Chu, H. C., Clements, S. M., ... & Andersson, A. J. (2023). Increasing hypoxia on global coral reefs under ocean warming. Nature Climate Change, 13(4), 403-409.

Phang, S., March, A., Touron-Gardic, G., Deane, K., & Failler, P. (2023). A review of the blue economy, potential, and opportunities in seven Caribbean nations pre-COVID-19. ICES Journal of Marine Science, 80(8), 2233-2243.

Phillips, M., Doyle, E., & Kramer, P. (2015). Transboundary Coral Reef Monitoring for the Grenadines Network of Marine Protected Areas.

Pike, E. P., MacCarthy, J. M., Hameed, S. O., Harasta, N., Grorud-Colvert, K., Sullivan-Stack, J., ... & Morgan, L. (2024). Ocean protection quality is lagging behind quantity: Applying a scientific framework to assess real marine protected area progress against the 30 by 30 target. Conservation Letters, 17(3), e13020.

Pilnick, A. R., O'Neil, K. L., Moe, M., & Patterson, J. T. (2021). A novel system for intensive Diadema antillarum propagation as a step towards population enhancement. Scientific Reports, 11(1), 11244.

Pina-Amargós, F., González-Díaz, P., González-Sansón, G., Aguilar-Betancourt, C., Rodríguez-Cueto, Y., Olivera-Espinosa, Y., ... & Alcolado Prieto, P. (2023). Status of Cuban Coral Reefs. In Coral Reefs of Cuba (pp. 283-307). Cham: Springer International Publishing.

Pina-Amargós, F. P., Fernández, L. H., Alonso, L. C., & Sansón, G. G. (2008). Características de los hábitats coralinos en Jardines de la Reina, Cuba. Revista de Investigaciones Marinas, 29(3), 225-237.

Pinheiro, H. T., MacDonald, C., Santos, R. G., Ali, R., Bobat, A., Cresswell, B. J., ... & Rocha, L. A. (2023). Plastic pollution on the world's coral reefs. Nature, 619(7969), 311-316.

Pittman SJ, Monaco ME, Friedlander AM, Legare B, Nemeth RS, et al. (2014). Fish with Chips: Tracking Reef Fish Movements to Evaluate Size and Connectivity of Caribbean Marine Protected Areas. PLOS ONE 9(5): e96028.

Pizarro, V., Rodríguez, S. C., López-Victoria, M., Zapata, F. A., Zea, S., Galindo-Martínez, C. T., Iglesias-Prieto, R., Pollock, J. F., & Medina, M. (2017). Unraveling the structure and composition of Varadero Reef, an improbable and imperiled coral reef in the Colombian Caribbean. PeerJ, 5, e4119.

Polaszek, T., F. Lacle, P. van Beukering, and E. Wolfs. (2018). The Economics of Ecosystems and Biodiversity, Aruba (Updated version). Amsterdam.

Pollnac, R., Pomeroy, R., & Harkes, I. (2010). Fishery policy and community-based management in the Caribbean. Marine Policy, 34(1), 1–9.

Pontes, E., Langdon, C., & Al-Horani, F. A. (2023). Caribbean scleractinian corals exhibit highly variable tolerances to acute hypoxia. Frontiers in Marine Science, 10, 1120262.

Porto-Hannes, I., Zubillaga, A. L., Shearer, T. L., Bastidas, C., Salazar, C., Coffroth, M. A., & Szmant, A. M. (2015). Population structure of the corals Orbicella faveolata and Acropora palmata in the Mesoamerican Barrier Reef System with comparisons over Caribbean basin-wide spatial scale. Marine Biology, 162(1), 81-98.

Potter, A. E., Chenoweth, S., & Day, M. (2017). Antigua and Barbuda. In Landscapes and landforms of the Lesser Antilles (pp. 99-116). Cham: Springer International Publishing.

Prato J., Mejía-Rentería J., Forbes M., Santos-Martínez A., Castaño D., Shuchmann P.W. (2024b). Extreme high-stress temperatures in Fall 2023 generate mass coral reef bleaching in the western Caribbean, Seaflower Biosphere Reserve. Bulletin of Marine Science.

Prato J., Santos-Martínez A, Cupul-Magaña A.L., Castaño D., Mancera-Pineda J.E., Medina J., Hudson A., Mejía-Rentería J.C., Velásquez C., Márquez G., Morales-de-Anda D., Wolff M and Schuhmann P.W. (2024a). Marine Ecosystem Services for Climate Change Adaptation and Mitigation Strategies in the Seaflower Biosphere Reserve: Coastal protection and fish biodiversity refuge at Caribbean insular territories. Springer Nature. P 149-182. In the Book: Climate Change Adaptation and Mitigation in the Seaflower Biosphere Reserve. Disaster Risk Reduction. Springer, Singapore.

Precht, W. F., Aronson, R. B., Gardner, T. A., Gill, J. A., Hawkins, J. P., Hernández-Delgado, E. A., ... & Côté, I. M. (2020). The timing and causality of ecological shifts on Caribbean reefs. In Advances in marine biology (Vol. 87, No. 1, pp. 331-360). Academic Press.

Precht, W. F., Vollmer, S. V., Modys, A. B., & Kaufman, L. (2019). Fossil Acropora prolifera (Lamarck, 1816) reveals coral hybridization is not only a recent phenomenon. Proceedings of the Biological Society of Washington, 132(1), 40-55.

Puerto Rico Department of Natural and Environmental Resources (2019). Puerto Rico Long-Term Coral Reef Monitoring Program database compilation: Substrate cover percent, octocoral colony counts, macroinvertebrate densities, fish densities, and fish biomass from 1999 to 2023 (NCEI Accession 0204647). PRCRMP_Benthic-sessile_data_1999–2023_(updated_11-30-2023) [Dataset]. NOAA National Centers for Environmental Information. https://www.ncei.noaa.gov/archive/accession/0204647 (Accessed 2024-10-18).

Puga, R., Valle, S., Kritzer, J. P., Delgado, G., de León, M. E., Giménez, E., ... & Karr, K. A. (2018). Vulnerability of nearshore tropical finfish in Cuba: implications for scientific and management planning. Bulletin of Marine Science, 94(2), 377-392.

Puotinen, M., Drost, E., Lowe, R., Depczynski, M., Radford, B., Heyward, A., & Gilmour, J. (2020). Towards modelling the future risk of cyclone wave damage to the world's coral reefs. Global Change Biology, 26(8), 4302-4315.

Purkis, C. (2021). Report for the benthic habitat mapping of the Southern Belize and Guatemalan Mesoamerican Reef, April 2021. Sea from Space, Smithsonian Institution, and Healthy Reefs Initiative.

Purkis, L. (2016). Summary report of satellite mapping of morphological and benthic habitats for the Honduran North Shore. 21

Purkis, L. (2015). Summary report of satellite mapping of morphological and benthic habitats for the Eastern Honduras Cays and Banks. 24

Pérez-Cervantes, E., Navarro-Espinoza,E., Estrada-Saldívar, N., Espinosa-Andrade, N., Melo-Merino, S., Rivas-Soto, M. Alvarez-Filip, L. (2016). Estado de conservación de los arrecifes de coral de la

Quezada-Perez, F., S. Mena, C. Fernández-García, J.J. Alvarado. (2023). Status of coral reef communities on the Caribbean coast of Costa Rica: Are we talking about corals or macroalgae reefs? Oceans 4(3), 315-330.

Quod, J. P., Séré, M., Hewson, I., Roth, L., & Bronstein, O. (2024). Spread of a sea urchin disease to the Indian Ocean causes widespread mortalities—Evidence from Réunion Island. Ecology, 106(1), e4476.

Quéré, G., Steneck, R. S., & Nugues, M. M. (2015). Spatiotemporal and species-specific patterns of diseases affecting crustose coralline algae in Curaçao. Coral Reefs, 34(1), 259-273.

R Core Team (2025). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Ramos, A., González-Díaz, P., Banaszak, A. T., Perera, O., Delgado, F. H., de León, S. D., ... & Duran, A. (2024). Seventeen-year study reveals fluctuations in key ecological indicators on two reef crests in Cuba. PeerJ, 12, e16705.

Ramos, R., Cipriani, R., Guzman, H. M., & García, E. (2009). Chronology of mercury enrichment factors in reef corals from western Venezuela. Marine Pollution Bulletin, 58(2), 222-229.

Ramos-Flores, T. (1983). Lower marine fungus associated with black line disease in star corals (Montastrea annularis, E. & S.). The Biological Bulletin, 165(2), 429-435.

Ramsey, V., Cooper, J. A. G., & Yates, K. L. (2015). Integrated Coastal Zone Management and its potential application to Antigua and Barbuda. Ocean & Coastal Management, 118, 259-274.

Randazzo-Eisemann, Á., & Garza-Pérez, J. R. (2023). Alacranes reef: A refuge for structurally complex coral species from increasing stressors. Ocean and Coastal Management, 244, 106817.

Randazzo-Eisemann, Á., Garza-Pérez, J. R., & Figueroa-Zavala, B. (2022). The role of coral diseases in the flattening of a Caribbean Coral Reef over 23 years. Marine Pollution Bulletin, 181, 113855.

Randazzo-Eisemann, Á., Arias-González, J. E., Velez, L., McField, M., & Mouillot, D. (2021). The last hotspots of structural complexity as conservation targets in the Mesoamerican Coral Reef. Biological Conservation, 256.

Rawlins, B. G., Ferguson, A. J., Chilton, P. J., Arthurton, R. S., Rees, J. G., & Baldock, J. W. (1998). Review of agricultural pollution in the Caribbean with particular emphasis on small island developing states. Marine Pollution Bulletin, 36(9), 658-668.

Razak, T. B., Budaya, R. R., Hukom, F. D., Subhan, B., Assakina, F. K., Fauziah, S., ... & Tebbett, S. B. (2024). Long-term dynamics of hard coral cover across Indonesia. Coral Reefs, 1-17.

Reef Check (2025). https://www.reefcheck.org/global-reef-tracker/

Reef Check Foundation (2023). Reef Check Global Reef Dataset [Dataset]. https://www.reefcheck.org/.

Reguero, B. G., Storlazzi, C. D., Gibbs, A. E., Shope, J. B., Cole, A. D., Cumming, K. A., & Beck, M. W. (2021). The value of US coral reefs for flood risk reduction. Nature Sustainability, 4(8), 688-698.

Reimer, J. D., Peixoto, R. S., Davies, S. W., Traylor-Knowles, N., Short, M. L., Cabral-Tena, R. A., ... & Voolstra, C. R. (2024). The fourth global coral bleaching event: where do we go from here?. Coral Reefs, 43(4), 1121-1125.

Resilient Islands, (2019). Valuing benefits of mangroves and coral reefs in the Caribbean. 20p

Restrepo, J. D., Park, E., Aquino, S., & Latrubesse, E. M. (2016). Coral reefs chronically exposed to river sediment plumes in the southwestern Caribbean: Rosario Islands, Colombia. Science of the Total Environment, 553, 316-329.

Reyes-Bonilla, H., & Jordán-Dahlgren, E. (2017). Caribbean coral reefs: Past, present, and insights into the future. In Marine animal forests (pp. 31-72). Springer, Cham.

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). Atoll Research Bulletin, 496(25), 460–479.

Rife, A.N.; Erisman, B.; Sanchez, A.; Aburto-Oropeza, O. (2013) When good intentions are not enough ... Insights on networks of "paper park" marine protected areas: Concerns regarding marine "paper parks". Cons. Let., 6, 200–212.

Rioja-Nieto, R., & Álvarez-Filip, L. (2019). Coral reef systems of the Mexican Caribbean: Status, recent trends and conservation. Marine pollution bulletin, 140, 616-625.

Rivas, N., Acero, A., & Tavera, J. (2022). Spatial variation of parrotfish assemblages at oceanic islands in the western Caribbean: evidence of indirect effects of fishing?. PeerJ, 10, e14178.

Rivas, E. J., Pérez, G. R., Tundisi, J. G., Vammen, K., Örmeci, B., & Forde, M. (2020). Eutrophication: A growing problem in the Americas and the Caribbean. Brazilian Journal of Biology, 80, 688-689.

Rivera, A., Bovey, M., Torres, C. et al. (2024).Community-based natural resource management in Roatan: Strengths and challenges. Ambio 53, 1864–1878

Roberts, C. M., Bohnsack, J. A., Gell, F., Hawkins, J. P., & Goodridge, R. (2001). Effects of marine reserves on adjacent fisheries. science, 294(5548), 1920-1923.

Roberts, C. M. (1995). Rapid build-up of fish biomass in a Caribbean marine reserve. Conservation Biology, 9(4), 815-826.

Roberts, H. H., Wilson, P. A., & Lugo-Fernández, A. (1992). Biologic and geologic responses to physical processes: examples from modern reef systems of the Caribbean-Atlantic region. Continental Shelf Research, 12(7-8), 809-834.

Roberts, L. (1987). Coral Bleaching Threatens Atlantic Reefs: unexplained changes are occurring in some of the most productive ecosystems on the planet, the Caribbean coral reefs. Science, 238(4831), 1228-1229.

Robertson, D. R., Simões, N., Gutiérrez-Rodríguez, C., Piñeros, V. J., & Pérez-España, H. (2016). An Indo-Pacific damselfish well established in the southern Gulf of Mexico: Prospects for a wider, adverse invasion. Journal of the Ocean Science Foundation, 19, 1–17.

Rocha, L. A., Pinheiro, H. T., Shepherd, B., Papastamatiou, Y. P., Luiz, O. J., Pyle, R. L., & Bongaerts, P. (2018). Mesophotic coral ecosystems are threatened and ecologically distinct from shallow water reefs. Science, 361(6399), 281-284.

Rodriguez, A. G., & Fanning, L. M. (2018). Queen conch (Lobatus gigas) in the Grenadine islands: A preliminary assessment on its abundance and current management needs. Ocean & Coastal Management, 165, 109-116.

Rodríguez, S., Cróquer, A., Bone, D., & Bastidas, C. (2010). Severity of the 1998 and 2005 bleaching events in Venezuela, southern Caribbean. Revista de biología tropical, 58, 189-196.

Rodriguez, S., & Cróquer, A. (2008). Dynamics of black band disease in a Diploria strigosa population subjected to annual upwelling on the northeastern coast of Venezuela. Coral Reefs, 27(2), 381-388.

Rodríguez-Martínez, R. E., Torres-Conde, E. G., Rosellón-Druker, J., Cabanillas-Terán, N., & Jáuregui-Haza, U. (2025). The Great Atlantic Sargassum Belt: Impacts on the Central and Western Caribbean-A Review. Harmful Algae, 102838.

Rodríguez-Martínez, R. E., Banaszak, A. T., McField, M. D., Beltran-Torres, A. U., & Alvarez-Filip, L. (2014). Assessment of Acropora palmata in the mesoamerican reef system. Plos one, 9(4), e96140.

Rodríguez-Martínez, R. E., Ruíz-Rentería, F., van Tussenbroek, B., Barba-Santos, G., Escalante-Mancera, E., Jordán-Garza, G., & Jordán-Dahlgren, E. (2010). Environmental state and tendencies of the Puerto Morelos CARICOMP site, Mexico. Revista de Biología Tropical, 58 (Suppl. 3), 23-43.

Rodríguez-Muñoz, R., Muñiz-Castillo, A.I., Euán-Avila, J.I., Hernández-Núñez, H., Valdés-Lozano, D.S., Collí-Dulá, R.C., Arias-González, J.E. (2021). Assessing temporal dynamics on pelagic Sargassum influx and its relationship with water quality parameters in the Mexican Caribbean. Reg. Stud. Mar. Sci. 48, 102005

Roff, G., Joseph, J., & Mumby, P. J. (2020). Multi-decadal changes in structural complexity following mass coral mortality on a Caribbean reef. Biogeosciences Discussions, 2020, 1-16.

Rogers, C. S., & Ramos-Scharrón, C. E. (2022). Assessing effects of sediment delivery to coral reefs: A Caribbean watershed perspective. Frontiers in Marine Science, 8, 773968.

Rogers, A., & Lorenzen, K. (2016). Does slow and variable recovery of Diadema antillarum on Caribbean fore-reefs reflect density-dependent habitat selection?. Frontiers in Marine Science, 3, 63.

Rogers, A., & Lorenzen, K. (2008). Recovery of Diadema antillarum and the potential for active rebuilding measures: modeling population dynamics. In Proceedings of the 11th international coral reef symposium (pp. 956-960).

Rogers, C. S., & Garrison, V. H. (2001). Ten years after the crime: lasting effects of damage from a cruise ship anchor on a coral reef in St. John, US Virgin Islands. Bulletin of Marine Science, 69(2), 793-803.

Rogers C. & Beets J. (2001). Degradation of marine ecosystems and decline of fishery resources in marine protected areas in the U.S. Virgin Islands. Environmental Conservation 28, 312–322.

Rogers, C. S. (1990). Responses of coral reefs and reef organisms to sedimentation. Marine Ecology Progress Series, 62, 185–202.

Rogers, C. S. (1983). Sublethal and lethal effects of sediments applied to common Caribbean reef corals in the field. Marine Pollution Bulletin, 14(10), 378-382.

Romero-Rodríguez, J., Acosta, A., & Zapata, F. A. (2014). Variables ambientales durante el blanqueamiento coralino en el Caribe colombiano. Revista de Biología Tropical, 62(Suppl. 1),

Rosales-Melgar, M. de los A., & Santana-Morales, O. (2024). Elasmobranchs small-scale fishery in Guatemala: Socioeconomic aspects and value chain structure. Social Sciences & Humanities Open, 10(Suppl C), Article 100506.

Rosemond R. C., Nemeth R. S. and Heppell S. A. (2022). Demographic recovery of a reef fish population over 30 years of spawning aggregation site protection. Frontiers in Marine Science. 9:931409.

Roth, L., Eviatar, G., Schmidt, L. M., Bonomo, M., Feldstein-Farkash, T., Schubert, P., ... & Bronstein, O. (2024). Mass mortality of diadematoid sea urchins in the Red Sea and Western Indian Ocean. Current Biology, 34(12), 2693-2701.

Rowlands, G. P., Purkis, S. J., Riegl, B. M. (2008). The 2005 coral-bleaching event, Roatán (Honduras): use of pseudoinvariant features (PIFs) in satellite assessments. Journal of Spatial Science, 53(1), 99-112.

Rubbens, P., Brodie, S., Cordier, T., Destro Barcellos, D., Devos, P., Fernandes-Salvador, J. A., ... & Irisson, J. O. (2023). Machine learning in marine ecology: an overview of techniques and applications. ICES Journal of Marine Science, 80(7), 1829-1853.

Rueda-Roa, D. T., & Muller-Karger, F. E. (2013). The southern Caribbean upwelling system: Sea surface temperature, wind forcing and chlorophyll concentration patterns. Deep Sea Research Part I: Oceanographic Research Papers, 78, 102-114.

Ruiz-Allais, J. P., Benayahu, Y., & Lasso-Alcalá, O. M. (2021). The invasive octocoral Unomia stolonifera (Alcyonacea, Xeniidae) is dominating the benthos in the Southeastern Caribbean Sea. Memoria de la Fundación La Salle de Ciencias Naturales, 79(187), 63-80.

Ruiz Allais, J. P., Amaro, M. E., McFadden, C. S., Halász, A., & Benayahu, Y. (2014). The first incidence of an alien soft coral of the family Xeniidae in the Caribbean, an invasion in eastern Venezuelan coral communities. Coral Reefs, 33(2), 287-287.

Rummer, J. L., and Illing, B. (2022). Coral reef fishes in a multi-stressor world. In Fish Physiology (Vol. 39, pp. 325-391). Academic Press.

Runnalls, L. A., & Coleman, M. L. (2003). Record of natural and anthropogenic changes in reef environments (Barbados West Indies) using laser ablation ICP-MS and sclerochronology on coral cores. Coral Reefs, 22(4), 416-426.

Rushmore, M. E., Ross, C., & Fogarty, N. D. (2021). Physiological responses to short-term sediment exposure in adults of the Caribbean coral Montastraea cavernosa and adults and recruits of Porites astreoides. Coral Reefs, 40, 1579-1591.

Ruttenberg, B., Caselle, J. E., Estep, A. J., Johnson, A. E., Marhaver, K. L., Richter, L. J., ... & Cannon, A. (2018). Ecological assessment of the marine ecosystems of Barbuda, West Indies: Using rapid scientific assessment to inform ocean zoning and fisheries management. PLoS One, 13(1), e0189355.

Ruzicka, R. R., Colella, M. A., Porter, J. W., Morrison, J. M., Kidney, J. A., Brinkhuis, V., ... & Colee, J. (2013). Temporal changes in benthic assemblages on Florida Keys reefs 11 years after the 1997/1998 El Niño. Marine Ecology Progress Series, 489, 125-141.

Ryan, J. (2012). Ecological and Biological Baseline Characterizations on the Nicaragua Shelf. Chapter 5, ESIA for Seismic Exploration on the Nicaragua Shelf: Report for Permit Approval submitted to the Nicaraguan Ministry of Environment and Natural Resources. 37 99.

Ryan, J., & Zapata, Y. (2003). Nicaragua's coral reefs. In K. R. Latin American Coral Reefs (pp. 203–222). Amsterdam: Elsevier.

Ryan, J. D., L. J. Miller, Y. Zapata, O. Downs and R. Chang. (1998). Nicaragua's coral reefs. In: UNESCO, 1998. CARICOMP - Caribbean coral reef, seagrass and mangrove sites. Coastal region and small island papers 3, UNESCO, Paris, xiv + 347 pp . B. Kjerfve Editor.

Ryan, J. (1994). The Corn Island Reef Surveys: Summary of Findings and Recommended Actions. Final Report Submitted to the Norwegian International

Ryan, J. (1992). Medioambientes Marinos de la Costa Caribe de Nicaragua (Primera Parte). WANI 12. pp35-47.

Sabido-Itzá, M. M., Villegas-Sánchez, C. A., Hernández-Millán, M. B., Medina-Quej, A., del Carmen García-Rivas, M., & Castro-Pérez, J. M. (2021). Characterization of queen conch (Aliger gigas) artisanal fishing prior to and after the five-year ban at Chinchorro Bank Biosphere Reserve, Mexico. Regional Studies in Marine Science, 48, 102074.

Sadovy de Mitcheson, Y., Aguilar-Perera, A., and Sosa-Cordero, E. (2018). Epinephelus striatus. IUCN Red List Threatened Species 2018, e.T7862A46909843. doi: 10.2305/IUCN.UK.2018- 2.RLTS.T7862A46909843.en

Sadovy de Mitcheson, Y., Craig, M. T., Bertoncini, A. A., Carpenter, K. E., Cheung, W. W., Choat, J. H., et al. (2013). Fishing groupers towards extinction: a global assessment of threats and extinction risks in a billion dollar fishery. Fish Fisheries 14 (2), 119–136. doi: 10.1111/j.1467-2979.2011.00455.x

Sadovy de Mitcheson, Y., Domeier M. (2005). Are aggregation-fisheries sustainable? Reef fish fisheries as a case study. Coral Reefs. 24:254–262.

Salas S, Chuenpagdee R, Seijo JC, Charles A (2007). Challenges in the assessment and management of small-scale fisheries in Latin America and the Caribbean. Fish Res 87:5–16.

Salazar, S.K., Larez, C. Arcia, M. & Montes, A. (2024, in press). Protocolo para el control y mitigación del coral blando invasor Unomia stolonoifera (Alcyonacea, Xeniidae). Revista FONTUS 18-19.

Samhouri, J. F., & Stier, A. C. (2021). Ecological impacts of an invasive mesopredator do not differ from those of a native mesopredator: lionfish in Caribbean Panama. Coral Reefs, 40(5), 1593-1600.

Sammarco, P. W., Porter, S. A., Genazzio, M., & Sinclair, J. (2015). Success in competition for space in two invasive coral species in the western Atlantic–Tubastraea micranthus and T. coccinea. PLoS ONE, 10(12), e0144581.

Sammonds, P., Alam, A., Day, S., Stavrianaki, K., & Kelman, I. (2023). Hurricane risk assessment in a multi-hazard context for Dominica in the Caribbean. Scientific reports, 13(1), 20565.

Sandin, S. A., Alcantar, E., Clark, R., De León, R., Dilrosun, F., Edwards, C. B., ... & Vermeij, M. J. (2022). Benthic assemblages are more predictable than fish assemblages at an island scale. Coral reefs, 41(4), 1031-1043.

Sandin, S. A., French, B. J., & Zgliczynski, B. J. (2022). Emerging insights on effects of sharks and other top predators on coral reefs. Emerging topics in life sciences, 6(1), 57-65.

Santander-Monsalvo, J., López-Huerta, I., Aguilar-Perera, A., & Tuz-Sulub, A. (2012). First record of the red lionfish (Pterois volitans [Linnaeus, 1758]) off the coast of Veracruz, Mexico. BioInvasions Records, 1(2), 121-124.

Scanlon, K. M., Ackerman, S. D., & Rozycki, J. E. (2005). Sedimentary environments of East and West Flower Garden banks area. Gulf of Mexico Science, 23(1), 9.

Schill, S. R., Raber, G. T., Roberts, J. J., Treml, E. A., Brenner, J., & Halpin, P. N. (2015). No reef is an island: integrating coral reef connectivity data into the design of regional-scale marine protected area networks. PLoS One, 10(12), e0144199.

Schofield, P. J. (2009). Geographic extent and chronology of the invasion of non-native lionfish (Pterois volitans [Linnaeus 1758] and P. miles [Bennett 1828]) in the Western North Atlantic and Caribbean Sea.

Schutte, V. G., Selig, E. R., & Bruno, J. F. (2010). Regional spatio-temporal trends in Caribbean coral reef benthic communities. Marine Ecology Progress Series, 402, 115-122.

Schweizer, D., Armstrong, R. A., & Posada, J. (2005). Remote sensing characterization of benthic habitats and submerged vegetation biomass in Los Roques Archipelago National Park, Venezuela. International Journal of Remote Sensing, 26(12), 2657-2667.

Scoffin, T. P. (1994). History of a Fringing Reef on the West Coast of Barbados 1974-1992.

Segaran, T. C., Soffa, F. B., Lananan, F., Sarà, G., Feng, Z., Azra, M. N., & Safuan, C. D. (2024). Coral restoration: a mapping review through a scientometric analysis. Restoration Ecology, 32(7), e14229.

Selig, E. R., & Bruno, J. F. (2010). A global analysis of the effectiveness of marine protected areas in preventing coral loss. PLos one, 5(2), e9278.

Sellares-Blasco, R., & Croquer, A. (2024). MONITOREA: Integrando la evaluación de ecosistemas marino-costeros para su conservación y manejo. Manual= de monitoreo y guía práctica de protocolos. Fundacion Dominicana de Estudios Marinos (FUNDEMAR). Santo Domingo, República Dominicana.

Seraphim, M. J., Sloman, K. A., Alexander, M. E., Janetski, N., Jompa, J., Ambo-Rappe, R., ... & Harborne, A. R. (2020). Interactions between coral restoration and fish assemblages: implications for reef management. Journal of Fish Biology, 97(3), 633-655.

Shaff, J. F., Santiago, I. A. M., Ilosvay, X. E., Tovar-Ávila, J., Ojea, E., Beaudreau, A. H., ... & Bueno, E. A. (2023). Documenting historical changes in shark fisheries near Islas Marías, Mexico, using fishers' local ecological knowledge. Fisheries Research, 265, 106748.

Shan, K., Lin, Y., Chu, P. S., Yu, X., & Song, F. (2023). Seasonal advance of intense tropical cyclones in a warming climate. Nature, 623(7985), 83-89.

Shantz, A. A., López-Londoño, T., Gomez-Campo, K., Iglesias-Prieto, R., López-Victoria, M., & Medina, M. (2024). Reassessing the role of herbivores on urban coral reefs: A case study from a heavily impacted reef near Cartagena Bay, Colombia. Urban Ecosystems, 27(3), 689-697.

Shantz, A. A., Ladd, M. C., & Burkepile, D. E. (2020). Overfishing and the ecological impacts of extirpating large parrotfish from Caribbean coral reefs. Ecological Monographs, 90(2), e01403.

Shaw, K. M. M., Standish, C. D., Fowell, S. E., Stewart, J. A., Castillo, K. D., Ries, J. B., & Foster, G. L. (2024). Century-long records of sedimentary input on a Caribbean reef from coral Ba/Ca ratios. Paleoceanography and Paleoclimatology, 39(5), e2023PA004746.

Sherman, K. D., Gomez, M. I., Kemenes, T., & Dahlgren, C. P. (2022). Spatial and Temporal Variability in Parrotfish Assemblages on Bahamian Coral Reefs. Diversity, s14(8), 625.

Sherman, K. D., Shultz, A. D., Dahlgren, C. P., Thomas, C., Brooks, E., Brooks, A., ... & Murchie, K. J. (2018). Contemporary and emerging fisheries in The Bahamas—Conservation and management challenges, achievements and future directions. Fisheries Management and Ecology, 25(5), 319-331.

Shulman, M. J., & Robertson, D. R. (1996). Changes in the coral reefs of San Blas, Caribbean Panama: 1983 to 1990.Coral Reefs, 15, 231–236..

Simancas-Giraldo, S. M., Xiang, N., Kennedy, M. M., Nafeh, R., Zelli, E., & Wild, C. (2021). Photosynthesis and respiration of the soft coral Xenia umbellata respond to warming but not to organic carbon eutrophication. PeerJ, 9, e11663.

Simpson, M. C., Mercer Clarke, C. S., Clarke, J. D., Scott, D., & Clarke, A. J. (2012). Coastal setbacks in Latin America and the Caribbean: A study of emerging issues and trends that inform guidelines for coastal planning and development.

Sing Wong, A., Vrontos, S., & Taylor, M. L. (2022). An assessment of people living by coral reefs over space and time. Global Change Biology, 28(23), 7139-7153.

Singh, A., Asmath, H., Chee, C. L., & Darsan, J. (2015). Potential oil spill risk from shipping and the implications for management in the Caribbean Sea. Marine pollution bulletin, 93(1-2), 217-227.

Singh, A., & Mee, L. (2008). Examination of policies and MEAs commitment by SIDS for sustainable management of the Caribbean Sea. Marine Policy, 32(3), 274-282.

Skirving, W., Marsh, B., De La Cour, J., Liu, G., Harris, A., Maturi, E., ... & Eakin, C. M. (2020). CoralTemp and the coral reef watch coral bleaching heat stress product suite version 3.1. Remote Sensing, 12(23), 3856.

Slijkerman, D. M., de León, R., & de Vries, P. (2014). A baseline water quality assessment of the coastal reefs of Bonaire, Southern Caribbean. Marine pollution bulletin, 86(1-2), 523-529.

Smale, D. A., Wernberg, T., Oliver, E. C., Thomsen, M., Harvey, B. P., Straub, S. C., ... & Moore, P. J. (2019). Marine heatwaves threaten global biodiversity and the provision of ecosystem services. Nature Climate Change, 9(4), 306-312.

Small, M., & Oxenford, H. A. (2022). Impacts of cruise ship anchoring during COVID-19: Management failures and lessons learnt. Ocean & Coastal Management, 229, 106332.

Smith TB, Brandt ME, Brandtneris VW, Ennis RS, Groves SH, Habtes S, Holstein DM, Kadison E, Nemeth RS (2019). The United States Virgin Islands. In: Loya Y, Puglise KA, Bridge T (eds) Coral Reefs of the World: Mesophotic Coral Ecosystems. Springer, pp131-147

Smith TB, Ennis RS, Kadison E, Nemeth RS, Henderson LM (2018). The United States Virgin Islands Territorial Coral Reef Monitoring Program. 2018 Annual Report. University of the Virgin Islands, United States Virgin Islands 289

Smith TB, Brandt ME, Calnan JM, Nemeth RS, Blondeau J, Kadison E, Taylor M, Rothenberger JP (2013). Convergent mortality responses of Caribbean coral species to seawater warming. Ecosphere 4:art87

Smith, A. H., Rogers, C. S., and Bouchon, C. (1997). Status of western Atlantic coral reefs in the Lesser Antilles. Proc. 8th Int. Coral Reef Symp. 1, 351–356.

Sobel, A. H., Wing, A. A., Camargo, S. J., Patricola, C. M., Vecchi, G. A., Lee, C. Y., & Tippett, M. K. (2021). Tropical cyclone frequency. Earth's Future, 9(12), e2021EF002275.

Souter, D., Planes, S., Wicquart, J., Logan, M., Obura, D., Staub, F. (eds.) (2021). Status of coral reefs of the world: 2020. Global Coral Reef Monitoring Network (GCRMN) and International Coral Reef Initiative (ICRI).

Spadaro, A. J., & Butler, M. J. (2021). Herbivorous crabs reverse the seaweed dilemma on coral reefs. Current Biology, 31(4), 853-859.

Spake, R., Mori, A. S., Beckmann, M., Martin, P. A., Christie, A. P., Duguid, M. C., & Doncaster, C. P. (2021). Implications of scale dependence for cross-study syntheses of biodiversity differences. Ecology Letters, 24(2), 374-390.

Spalding, M., Longley-Wood, K., Acosta-Morel, M., Cole, A., Wood, S., Haberland, C., & FerdaÒa, Z. (2018). Estimating Reef-Adjacent Tourism Value in the Caribbean. The Nature Conservancy.

Spalding, M. D., Fox, H. E., Allen, G. R., Davidson, N., Ferdaña, Z. A., Finlayson, M. A. X., ... & Robertson, J. (2007). Marine ecoregions of the world: a bioregionalization of coastal and shelf areas. BioScience, 57(7), 573-583.

Spanier, E. (2024). Marine protected areas–a review of their potential effects on lobster population size & structure and fisheries management. Fisheries Research, 275, 107025.

Speare, K. E., Duran, A., Miller, M. W., & Burkepile, D. E. (2019). Sediment associated with algal turfs inhibits the settlement of two endangered coral species. Marine Pollution Bulletin, 144, 189-195.

Stein J, Huebner LK, Colella M, Harrell C, and Ruzicka R. (2024). Florida's Coral Reef 2023-2024 Post-Bleaching Assessment Quick Look Report. Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute, St. Petersburg, FL. 23 pp. https://ocean.floridamarine.org/FRRP/Home/About

Steiner, S. C. C. (2015). Coral Reefs of Dominica (Lesser Antilles). Annalen des Naturhistorischen Museums in Wien. Serie B für Botanik und Zoologie, 47-119.

Steiner, S. C. C. (1999). Species presence and distribution of Scleractinia (Cnidaria: Anthozoa) from South Caicos, Turks and Caicos Islands. Bulletin of Marine Science, 65(3), 861–871.

Steingrover, R., Illidge, S., Martina, C., Virginia-Cova, L., & Vermeij, M. (2016). Detection and Analysis of Antibiotic Resistance in Coliform Bacteria in Caribbean Coastal Water. In Open Forum Infectious Diseases (Vol. 3, No. suppl_1, p. 704). Oxford University Press.

Steneck, R. S., & Torres, R. (2023). Trends in Dominican Republic coral reef biodiversity 2015–2022. Diversity, 15(3), 389.

Steneck, R. S., Mumby, P. J., Arnold, S., Rasher, D. B, Wilson, M., de Leon, R., Boenish, R. (2019). Managing recovery resilience in coral reefs against climate-induced bleaching and hurricanes: a 15 year case study from Bonaire, Dutch Caribbean. Frontiers in Marine Sciences. doi: 10.3389/fmars.2020.579060.

Steneck, R. S., Mumby, P. J., MacDonald, C., Rasher, D. B., & Stoyle, G. (2018). Attenuating effects of ecosystem management on coral reefs. Science advances, 4(5), eaao5493. 10.1126/sciadv.aao5493

Steneck, R. S., Arnold, S. (2009). Status and trends of Bonaire's reefs & need for immediate action. Coral reef monitoring report to STINAPA Bonaire. 193 pp.

Stock, B. C., Heppell, S. A., Waterhouse, L., Dove, I. C., Pattengill-Semmens, C. V., McCoy, C. M., ... & Semmens, B. X. (2021). Pulse recruitment and recovery of Cayman Islands Nassau Grouper (Epinephelus striatus) spawning aggregations revealed by in situ length-frequency data. ICES Journal of Marine Science, 78(1), 277-292.

Stockton, L., & Edmunds, P. J. (2021). Spatially aggressive peyssonnelid algal crusts (PAC) constrain coral recruitment to Diadema grazing halos on a shallow Caribbean reef. Journal of Experimental Marine Biology and Ecology, 541, 151569.

Stoddart, D. R. (1969). Ecology and morphology of recent coral reefs. Biological Reviews, 44(4), 433-498.

Stoner, A. W., Davis, M. H., & Kough, A. S. (2019). Relationships between fishing pressure and stock structure in queen conch (Lobatus gigas) populations: Synthesis of long-term surveys and evidence for overfishing in The Bahamas. Reviews in Fisheries Science & Aquaculture, 27(1), 51-71.

Stoner, A. W., Davis, M. H., & Booker, C. J. (2012). Abundance and population structure of queen conch inside and outside a marine protected area: repeat surveys show significant declines. Marine Ecology Progress Series, 460, 101-114.

Strategy for Reef Resilience in Belize. (2023). Coral ecosystems in Belize face many threats that inhibit their capacity to rebound from shocks: Stressors such as rising sea temperatures. Coastal Zone Belize.

Studholme, J., Fedorov, A. V., Gulev, S. K., Emanuel, K., & Hodges, K. (2022). Poleward expansion of tropical cyclone latitudes in warming climates. Nature Geoscience, 15(1), 14-28.

Swain, T. D., DuBois, E., Gomes, A., Stoyneva, V. P., Radosevich, A. J., Henss, J., ... & Marcelino, L. A. (2016). Skeletal light-scattering accelerates bleaching response in reef-building corals. BMC ecology, 16, 1-18.

Sweet, M., Burn, D., Cróquer, A., & Leary, P. (2013). Characterisation of the bacterial and fungal communities associated with different lesion sizes of dark spot syndrome occurring in the coral Stephanocoenia intersepta. PLoS One, 8(4), e62580.

Szmant, A. M. (2002). Nutrient enrichment on coral reefs: is it a major cause of coral reef decline?. Estuaries, 25(4), 743-766.

Sánchez, J. A., Gómez-Corrales, M., Gutierrez-Cala, L., Vergara, D. C., Roa, P., González-Zapata, F. L., ... & Sarmiento, A. (2019). Steady decline of corals and other benthic organisms in the SeaFlower Biosphere Reserve (Southwestern Caribbean). Frontiers in Marine Science, 6, 73.

Sánchez, J. A., Herrera, S., Navas-Camacho, R., Rodríguez-Ramírez, A., Herron, P., Pizarro, V., ... & Orozco, C. (2010). White plague-like coral disease in remote reefs of the Western Caribbean. Revista de Biología Tropical, 58, 145-154.

Sánchez-Godínez, C. (2014). Capacidad de recuperación de los arrecifes coralinos del Caribe con base en la estructura bentónica actual y la densidad de reclutas. M.Sc. Thesis, Universidad de Costa Rica. San Pedro, Costa Rica. 116 p.

Tavares, R., Carreon-Zapiain, M. T., & Perez-Jimenez, J. C. (2024). Vulnerability of elasmobranchs caught by artisanal fishery in the Southeastern Caribbean. Regional Studies in Marine Science, 79, 103851.

Taylor, G. T., Muller-Karger, F. E., Thunell, R. C., Scranton, M. I., Astor, Y., Varela, R., ... & Doherty, O. (2012). Ecosystem responses in the southern Caribbean Sea to global climate change. Proceedings of the National Academy of Sciences, 109(47), 19315-19320.

Tebbett, S. B., Connolly, S. R., & Bellwood, D. R. (2023b). Benthic composition changes on coral reefs at global scales. Nature ecology & evolution, 7(1), 71-81.

Tebbett, S. B., Crisp, S. K., Evans, R. D., Fulton, C. J., Pessarrodona, A., Wernberg, T., ... & Bellwood, D. R. (2023a). On the challenges of identifying benthic dominance on anthropocene coral reefs. Bioscience, 73(3), 220-228.

Tebbett, S. B., Siqueira, A. C., & Bellwood, D. R. (2022). The functional roles of surgeonfishes on coral reefs: past, present and future. Reviews in Fish Biology and Fisheries, 1-53.

Tebbett, S. B., Hoey, A. S., Depczynski, M., Wismer, S., & Bellwood, D. R. (2020). Macroalgae removal on coral reefs: realised ecosystem functions transcend biogeographic locations. Coral Reefs, 39(1), 203-214.

Teneva, L., Free, C. M., Hume, A., Agostini, V. N., Klein, C. J., Watson, R. A., & Gaines, S. D. (2023). Small island nations can achieve food security benefits through climate-adaptive blue food governance by 2050. Marine Policy, 151, 105577.

Tewfik, A. (2023). Caribbean spiny lobster (Panulirus argus)"... Belize's premier commercial fishery overexploited".

Tewfik, A., Babcock, E. A., & Phillips, M. (2020). Spiny lobster fisheries status across time and a mosaic of spatial management regimes. ICES Journal of Marine Science, 77(3), 1002-1016.

Theile, S. (2001): Queen Conch fisheries and their management in the Caribbean. TRAFFIC Europe.

Toledo-Rodriguez, D. A., McFadden, C. S., Marrero, N. M. J., Muñoz-Maravilla, J. D., Veglia, A. J., Weil, E., & Schizas, N. V. (2025b). First record of a second invasive soft coral species, Latissimia ningalooensis, in southern Puerto Rico. bioRxiv, 2025-04.

Toledo-Rodriguez, D. A., Veglia, A., Marrero, N. M. J., Gomez-Samot, J. M., McFadden, C. S., Weil, E., & Schizas, N. V. (2025a). Shadows over Caribbean reefs: Identification of a new invasive soft coral species, Xenia umbellata, in southwest Puerto Rico. Coral Reefs, 1-6.

Tomascik, T., & Sander, F. J. M. B. (1985). Effects of eutrophication on reef-building corals: I. Growth rate of the reef-building coral Montastrea annularis. Marine biology, 87(2), 143-155.

Topor, Z. M., Rasher, D. B., Duffy, J. E., & Brandl, S. J. (2019). Marine protected areas enhance coral reef functioning by promoting fish biodiversity. Conservation Letters, 12(4), e12638.

Toral-Granda, V. (2008). Population status, fisheries and trade of sea cucumbers in Latin America and the Caribbean. Sea cucumbers. A global review of fisheries and trade. FAO Fisheries and Aquaculture Technical Paper, 516, 213-229.

Tosic, M., Restrepo, J. D., Lonin, S., Izquierdo, A., & Martins, F. (2019). Water and sediment quality in Cartagena Bay, Colombia: Seasonal variability and potential impacts of pollution. Estuarine, Coastal and Shelf Science, 216, 187-203.

Toth, L. T., Courtney, T. A., Colella, M. A., Kupfner Johnson, S. A., & Ruzicka, R. R. (2022). The past, present, and future of coral reef growth in the Florida Keys. Global Change Biology, 28(17), 5294-5309.

Townhill, B. L., Birchenough, S. N., Engelhard, G. H., Harrod, O., McHarg, E., Monnereau, I., & Buckley, P. J. (2021). Responding to climate change in Caribbean fisheries and aquaculture through adaptation. The CLME+ Hub, 1-57.

Truc M, Rivera A, Ochoa GM, Dueñas D, Guifarro Z, Brady G, Zuniga Z, Gutierrez B, Chock C and Zaldivar L (2023). Evaluating the spread of stony coral tissue loss disease in the Bay Islands, Honduras. Front. Mar. Sci. 10:1197318.

Tuohy, E., Wade, C., & Weil, E. (2020). Lack of recovery of the long-spined sea urchin Diadema antillarum Philippi in Puerto Rico 30 years after the Caribbean-wide mass mortality. PeerJ, 8, e8428.

Turner, J. R., McCoy, C., Cottam, M., Olynik, J., Timothy, A., Blumenthal, J., ... & Ebanks-Petrie, G. (2013). Biology and ecology of the coral reefs of the Cayman Islands. In Coral Reefs of the United Kingdom Overseas Territories (pp. 69-88). Dordrecht: Springer Netherlands.

UNESCO World Heritage Centre. Belize Barrier Reef Reserve System. Accessed on 08/06/2025 at: https://whc.unesco.org/en/list/764

UNFCCC (2016). The Paris Agreement United Nations Framework Convention on Climate Change.

United Nations Environment Programme- Caribbean Environment Programme (UNEP-CEP) (2021). Sargassum white Paper – Turning the crisis into an opportunity. Ninth Meeting of the Scientific and Technical Advisory Committee (STAC) To the Protocol Concerning Specially Protected Areas and Wildlife (SPAW) in the Wider Caribbean Region, Kingston, Jamaica.

Uribe, E. S., Etter, A., Luna-Acosta, A., Diazgranados, M. C., Alonso, D., Chasqui, L., ... & Giraldo, A. (2022). A comprehensive threat analysis to support the red list of marine and coastal ecosystems of Colombia. Frontiers in Marine Science, 9, 962044.

Uribe-Castañeda, N., Morales, N., Loizzo, J., Patterson, J., & Main, M. (2024). Community engagement in coral reef restoration in the Seaflower MPA, Colombia. Restoration Ecology, 32(8), e14268.

Vadziutsina, M., & Riera, R. (2021). Artisanal and small-scale fish trap fisheries from tropical and subtropical reefs: targeted species and conservation of fish stocks. Journal of Fisheries and Environment, 45(2), 69-83.

Valdivia, A., Cox, C. E., & Bruno, J. F. (2017). Predatory fish depletion and recovery potential on Caribbean reefs. Sci Adv. 3: e1601303. 10.1126/sciadv.1601303

Valdivia, A., Bruno, J. F., Cox, C. E., Hackerott, S., & Green, S. J. (2014). Re-examining the relationship between invasive lionfish and native grouper in the Caribbean. PeerJ, 2, e348.

Valentine, J. F., & Heck, K. L. (2005). Perspective review of the impacts of overfishing on coral reef food web linkages. Coral Reefs, 24, 209-213.

Vallès, H., Walcott, J., & Oxenford, H. A. (2025). No change in key reef fish herbivores or reef fisher yields in Barbados a decade after the lionfish invasion. Scientific Reports, 15(1), 6253.

Vallès, H., J. Walcott and H.A. Oxenford. (2023). Assessment and Management of Lionfish and Status of Other Marine Invasive Species of Threat to High Biodiversity-value Reef Ecosystems. Final Report. Preventing Costs of Invasive Alien Species (IAS) in Barbados and Countries of the OECS Project. CERMES, UWI, Cave Hill, Barbados, 53pp.

Vallès, H., Gill, D., & Oxenford, H. A. (2015). Parrotfish size as a useful indicator of fishing effects in a small Caribbean island. Coral Reefs, 34, 789-801.

van Dam, J. W., Negri, A. P., Uthicke, S., & Mueller, J. F. (2011). Chemical pollution on coral reefs: exposure and ecological effects.Dennis, M. M., Izquierdo, A., Conan, A., Johnson, K., Giardi, S., Frye, P., & Freeman, M. A. (2019). Scaphanocephalus-associated dermatitis as the basis for black spot disease in Acanthuridae of St. Kitts, West Indies. Diseases of aquatic organisms, 137(1), 53-63.

Van Der Geest M., Meesters E. H., Mücher S. (2020). Impact of terrestrial erosion on coral reef health at Bonaire: a plea for nature-inclusive "watershed-to-reef" based coastal management. Wageningen Marine Research, Den Helder, the Netherlands.

van Duyl, F. C., Post, V. E., van Breukelen, B. M., Bense, V., Visser, P. M., Meesters, E. H., ... & Vermeij, M. J. (2024). Composition and distribution of the near-shore waters bordering the coral reefs of Aruba, Bonaire, and Curaçao in the Southern Caribbean. Marine Pollution Bulletin, 209, 117297.

Van Duyl, F. C. (1985). Atlas of the Living Reefs of Curaçao and Bonaire (Netherlands Antilles). Foundation for Scientific Research in Surinam and the Netherlands Antilles, 117.

van Hooidonk, R. J., Manzello, D. P., Moye, J., Brandt, M. E., Hendee, J. C., McCoy, C., & Manfrino, C. (2012). Coral bleaching at little cayman, Cayman Islands 2009. Estuarine, Coastal and Shelf Science, 106, 80-84.

Van Steveninck, E. D. R., & Bak, R. P. M. (1986). Changes in abundance of coral-reef bottom components related to mass mortality of the sea urchin Diadema antillarum. Marine Ecology Progress Series, 34, 87-94.

Van Tussenbroek, B. I., Arana, H. A. H., Rodríguez-Martínez, R. E., Espinoza-Avalos, J., Canizales-Flores, H. M., González-Godoy, C. E., ... & Collado-Vides, L. (2017). Severe impacts of brown tides caused by Sargassum spp. on near-shore Caribbean seagrass communities. Marine pollution bulletin, 122(1-2), 272-281.

Van Veghel, M. L., & Hoetjes, P. C. (1995). Effects of tropical storm Bret on Curaçao reefs. Bulletin of marine science, 56(2), 692-694.

Van Woesik, R., Shlesinger, T., Grottoli, A. G., Toonen, R. J., Vega Thurber, R., Warner, M. E., ... & Zaneveld, J. (2022). Coral-bleaching responses to climate change across biological scales. Global change biology, 28(14), 4229-4250.

Vandepitte, L., Bosch, S., Tyberghein, L., Waumans, F., Vanhoorne, B., Hernandez, F., ... & Mees, J. (2015). Fishing for data and sorting the catch: assessing the data quality, completeness and fitness for use of data in marine biogeographic databases. Database, 2015, bau125.

Vega Thurber, R. L., Silva, D., Speare, L., Croquer, A., Veglia, A. J., Alvarez-Filip, L., ... & Correa, A. M. (2025). Coral Disease: Direct and Indirect Agents, Mechanisms of Disease, and Innovations for Increasing Resistance and Resilience. Annual Review of Marine Science, 17(1), 227-255.

Velterop, R. (2003). Sedimentation quality change and the presence of indicator species: A study across a seaward sedimentation gradient from the Pearl Cays to the Corn Islands. ReefMap Project Report for the Corn Islands and Pearl Cays.

Velázquez-Ochoa, R., & Enríquez, S. (2023). Environmental degradation of the Mexican Caribbean reef lagoons. Marine Pollution Bulletin, 191, 114947.

Verde, A., Bastidas, C., & Cróquer, A. (2016). Tissue mortality by Caribbean ciliate infection and white band disease in three reef-building coral species. PeerJ, 4, e2196.

Vermeij, M., Marhaver, K., Estep, A., and Sandin, S. (2019): Coral Reefs Baseline Study for Aruba.

Vermeij, M. J., Latijnhouwers, K. R., Dilrosun, F., Chamberland, V. F., Dubé, C. E., Van Buurt, G., & Debrot, A. O. (2019). Historical changes (1905-present) in catch size and composition reflect altering fisheries practices on a small Caribbean island. PLoS One, 14(6), e0217589.

Vermeij, M. J., van Moorselaar, I., Engelhard, S., Hörnlein, C., Vonk, S. M., & Visser, P. M. (2010). The effects of nutrient enrichment and herbivore abundance on the ability of turf algae to overgrow coral in the Caribbean. PloS one, 5(12), e14312.

Verweij, P. J. F. M., Meesters, H. W. G., & Debrot, A. O. (2015). Indicators on the status and trends of ecosystems in the Dutch Caribbean: indicators, monitoring and assessment methods and capacity estimates.

Vicente-Castro, P., Díaz, P. G., Amargós, F. P., Martín, T. F., González, A. G., & Duran, A. (2024). Conducta alimentaria de Scarus coelestinus (Teleostei: Scaridae) en hábitats arrecifales del Parque Nacional Jardines de la Reina, Cuba. Revista de investigaciones marinas, 44(2), 1.

Victoria-Salazar, I., Hernández-Arana, H.A., Meave, J.A., Ruiz-Zárate, M.A., Vega-Zepeda, A., Carricart-Ganivet, J.P., López-Adame, H. (2017). Did the community structure of a coral reef patch affected by a ship grounding recover after 15 years? Merging historical and recent data sets. Ocean Coast. Manag., 144,59-70

Vides, M, Alonso D, Castro E et al (eds) (2016) Biodiversidad del mar de los siete colores. Instituto de Investigaciones Marinas y Costeras—INVEMAR y Corporación para el Desarrollo Sostenible del Archipiélago de San Andrés, Providencia y Santa Catalina—CORALINA. Serie de Publicaciones Generales del INVEMAR, 84, Santa Marta, Colombia

Viehman, T. S., Grove, L. J. W., Edwards, K. F., Blondeau, J., Cain, E., Krampitz, N., Langwiser, C., Sturm, A., Swanson, D., Towle, E. K., &Williams, B. (2025). National Coral Reef Monitoring Program, Biological monitoring summary – U.S. Virgin Islands and Puerto Rico: 2023. NOAA Technical Memorandum NOS CRCP 53.

Viehman, T. S., Grove, L. J. W., Blondeau, J., Cain, E., Edwards, K. F., Groves, S. H., Krampitz, N., Langwiser, C., Siceloff, L., Swanson, D., Towle, E. K., and Williams, B. (2023). National Coral Reef Monitoring Program, Biological Monitoring Summary – Florida: 2022. NOAA Technical Memorandum NOS CRCP 48.

Viehman, T. S., Edwards, K. F., Blondeau, J. W., Cain, E., Grove, L. J. W., Groves, S. H., Krampitz, N., Langwiser, C., Siceloff, L., Swanson, D., Towle, E. K., and Williams, B. (2023). National Coral Reef Monitoring Program, Biological Monitoring Summary – Flower Garden Banks: 2022. NOAA Technical Memorandum NOS CRCP 47.

Viehman, T.S., Nemeth, M., Groves, S. H., Buckel, C. A., Griffin, S., Field, D., Moore, T. D., Moore, J. (2020) Coral assessment and restoration in the US Caribbean after 2017 hurricanes. NOAA Technical Memorandum NOS NCCOS 278.

Villalpando, M. F., Guendulain-García, S. D., Valdez-Trinidad, A., Croquer, A., & Sellares-Blasco, R. I. (2022). Coral reefs of southeastern Dominican Republic hit by two simultaneous epizootic events. Bulletin of Marine Science, 98(4), 507-508.

Villamizar, E. (2024). The Unomia experience in Venezuela. In "A learning exchange on a distinctive non-native soft coral, Unomia stolonifera. Slides 46-53 in https://www.agrra.org/wp-content/uploads/2024/03/Webinar-Slidedeck_Unomia-stolonifera_February-23-2024_Full-Presentations_pdf.pdf

Villamizar, E. (2021). Impacts of the oil spills over the coral reefs and their ecosystems goods and services. Boletín de la Academia de Ciencias Físicas, Matemáticas y Naturales, Vol. LXXXI (1), 45-52.

Villamizar, E., Romero, M., & Boadas, H. (2014b). Estructura de las comunidades de corales y octocorales de Isla de Aves, Venezuela, Caribe Nororiental. Revista de Biología Tropical, 62(3), 115-136.

Villamizar E., A. Yranzo, M. Gonzalez, A.T. Herrera, J. Pérez y H. Camisotti (2014a). Diversidad y condición de salud de corales pétreos en algunos arrecifes del Parque Nacional Archipiélago Los Roques, Venezuela. Acta Biológica Venezuélica, Vol. 34 (2), 257-279.

Villamizar, E., Camisotti, H., Rodríguez, B., Pérez, J., & Romero, M. (2008). Impacts of the 2005 Caribbean bleaching event at Archipiélago de Los Roques National Park, Venezuela. Revista de Biología Tropical, 56(1), 255-270.

Villamizar, E., Posada, J. M., & Gómez, S. (2003). Venezuela. Rapid Assessment of coral reefs in the Archipelago de Los Roques National Park, Venezuela (part 1: stony corals and algae). Atoll Research Bulletin (nov.), Nro. 496, 512-529.

Villamizar, E. (2000). Estructura de una comunidad arrecifal en Falcón, Venezuela, antes y después de una mortalidad masiva. Revista de Biologia Tropical, 48(S1), 19-30.

Voolstra, C. R., Buitrago-López, C., Perna, G., Cárdenas, A., Hume, B. C., Rädecker, N., & Barshis, D. J. (2020). Standardized short-term acute heat stress assays resolve historical differences in coral thermotolerance across microhabitat reef sites. Global Change Biology, 26(8), 4328-4343.

Waitt Institute (2016). Economic valuation of Curaçao's marine resources. Downloadable from: https://www.waittinstitute.org/_files/ugd/47d1fd_93b3e06900f74a6eab2c3c2adb6d50c2.pdf

Wakwella, A., Wenger, A., Jenkins, A., Lamb, J., Kuempel, C. D., Claar, D., ... & Jupiter, S. D. (2023). Integrated watershed management solutions for healthy coastal ecosystems and people. Cambridge Prisms: Coastal Futures, 1, e27.

Walker, B. K., Gilliam, D. S., Dodge, R. E., & Walczak, J. (2012). Dredging and shipping impacts on southeast Florida coral reefs.

Walsh, K. J., Camargo, S. J., Knutson, T. R., Kossin, J., Lee, T. C., Murakami, H., & Patricola, C. (2019). Tropical cyclones and climate change. Tropical Cyclone Research and Review, 8(4), 240-250.

Waterhouse, L., Heppell, S. A., Pattengill-Semmens, C. V., McCoy, C., Bush, P., Johnson, B. C., et al. (2020). Recovery of critically endangered Nassau grouper (Epinephelus striatus) in the Cayman islands following targeted conservation actions. Proc. Natl. Acad. Sci. 117, 1587–1595.

Wecker, P., Lecellier, G., Guibert, I., Zhou, Y., Bonnard, I., Berteaux-Lecellier, V., (2018). Exposure to the environmentally-persistent insecticide chlordecone induces detoxification genes and causes polyp bail-out in the coral P. damicornis. Chemosphere, 195, 190-200.

Wehner, M. F., & Kossin, J. P. (2024). The growing inadequacy of an open-ended Saffir–Simpson hurricane wind scale in a warming world. Proceedings of the National Academy of Sciences, 121(7), e2308901121.

Weil, E., Croquer, A., & Urreiztieta, I. (2009). Temporal variability and impact of coral diseases and bleaching in Puerto Rico. Coral Reefs, 28(1), 47–64.

Weil, E. (2004). Coral reef diseases in the wider Caribbean. In Coral health and disease (pp. 35-68). Berlin, Heidelberg: Springer Berlin Heidelberg.

Weil, E. (2003). The corals and coral reefs of Venezuela. In Latin American coral reefs (pp. 303-330). Elsevier Science.

Weil, E., Urreiztieta, I., & Garzón-Ferreira, J. (2002). Geographic variability in the incidence of coral and octocoral diseases in the wider Caribbean. In Proceedings of the Ninth International Coral Reef Symposium, Bali, 23-27 October 2000, (Vol. 2, pp. 1231-1237).

Whiteman, E. A., Jennings C. A. and Nemeth R. S. (2005). Sex structure and potential female fecundity in a red hind (Epinephelus guttatus) spawning aggregation: applying ultrasonic imaging. Journal of Fish Biology. 66:983-995.

Wickham, H., Averick, M., Bryan, J., Chang, W., McGowan, L. D. A., François, R., ... & Yutani, H. (2019). Welcome to the Tidyverse. Journal of open source software, 4(43), 1686.

Wicquart J., Towle E. K., Dallison T., Staub F., and Planes S. (eds.) (2025). Status and Trends of Coral Reefs of the Pacific: 1980-2023. Global Coral Reef Monitoring Network (GCRMN) and International Coral Reef Initiative (ICRI).

Wicquart, J., Gudka, M., Obura, D., Logan, M., Staub, F., Souter, D., & Planes, S. (2022). A workflow to integrate ecological monitoring data from different sources. Ecological Informatics, 68, 101543.

Wiener, J. (2013). Toward the development of Haiti's system of Marine Protected Areas (MPAs). Fondation pour la Protection de la Biodiversité Marine.

Wijers, T., Hylkema, A., Pilnick, A. R., Murk, A. J., & Patterson, J. T. (2023). Novel shaker bottle cultivation method for the long spined sea urchin (Diadema antillarum; Philippi, 1845) results in high larval survival and settlement rates. Aquaculture, 562, 738855.

Wijers, T., Klokman, O., Murk, A. J., Sijbrandij, J., & Hylkema, A. (2024). Low retention of restocked laboratory-reared long-spined sea urchins Diadema antillarum due to Spanish hogfish Bodianus rufus predation. Restoration Ecology, 32(8), e14264.

Wilkinson, C., Nowak, M., Miller, I., & Baker, V. (2013). Status of Caribbean coral reefs in seven countries in 1986. Marine Pollution Bulletin, 70(1), 7–9.

Wilkinson, C., & Souter, D. (2008). Status of Caribbean coral reefs after bleaching and hurricanes in 2005.

Wilkinson, C. C. (2008). Status of coral reefs of the world: 2008. Australian Institute of Marine Science (AIMS).

Wilkinson, C. C. (2004). Status of coral reefs of the world: 2004. Australian Institute of Marine Science (AIMS).

Wilkinson, C. C. (2002). Status of coral reefs of the world: 2002. Australian Institute of Marine Science (AIMS).

Wilkinson, C. C. (2000). Status of coral reefs of the world: 2000. Australian Institute of Marine Science (AIMS).

Wilkinson, C.C. (1998). Status of coral reefs of the world: 1998. Australian Institute of Marine Science (AIMS).

Williams, S. M. (2022). The reduction of harmful algae on Caribbean coral reefs through the reintroduction of a keystone herbivore, the long-spined sea urchin Diadema antillarum. Restoration Ecology, 30(1), e13475.

Williams, S. M., & García-Sais, J. R. (2020). A potential new threat on the coral reefs of Puerto Rico: The recent emergence of Ramicrusta spp. Marine Ecology, 41(4), e12592.

Williams, S. M. (2016). A novel approach to the restoration of Diadema antillarum on coral reefs in the Caribbean. Reef Encounters, 31, 48-50.

Williams, S.M., P.J. Mumby, I. Chollett, J. Cortés. (2015). Importance of differentiating Orbicella reefs from gorgonias plains for ecological assessments of Caribbean reefs. Marine Ecology Progress Series 530: 93-101

Williams, I., & Polunin, N. (2001). Large-scale associations between macroalgal cover and grazer biomass on mid-depth reefs in the Caribbean. Coral reefs, 19(4), 358-366.

Williamson, C, M.; Williams S., D.; Arróliga, O.; et al. (2019). Índice de salud arrecifal (ISA) en los arrecifes coralinos de Cayos Miskitos. Revista Universitaria del Caribe, 22(1), 16-23.

Wilson, B., Fan, C. M., & Edmunds, P. J. (2020). An unusual microbiome characterises a spatially-aggressive crustose alga rapidly overgrowing shallow Caribbean reefs. Scientific Reports, 10(1), 20949.

Winterbottom, M., Haughton, M., Mutrie, E., & Grieve, K. (2012). Management of the Spiny Lobster Fishery in CARICOM Countries: Status and Recommendations for Conservation.

Woodhead, A. J., Hicks, C. C., Norström, A. V., Williams, G. J., and Graham, N. A. (2019). Coral reef ecosystem services in the Anthropocene. Functional Ecology, 33(6), 1023-1034.

World Meterological Association (2025). WMO global annual to decadal climate update: 2025–2029.

Wynne, S. P. (2017). Trends and other temporal changes recorded in coral reef and Seagrass areas during ten years of the Anguilla marine monitoring Programme (37 pp.). Anguilla Fisheries and Marine Resources Research Bulletin 06.

Wynne, S. P. (2016). Status of Lionfish (Pterois volitans) populations in Anguilla, British West Indies, with additional biometric and stomach content analysis results.

Wynne, S. P., & Côté, I. M. (2007). Effects of habitat quality and fishing on Caribbean spotted spiny lobster populations. Journal of Applied Ecology, 488-494.

Xu, X., Chassignet, E. P., Miron, P., & Zavala-Romero, O. (2024). Seasonality of Marine Litter Hotspots in the Wider Caribbean Region. Journal of Marine Science and Engineering, 12(2), 319.

Yranzo, A., Villamizar, E., Romero, M., & Boadas, H. (2014). Estructura de las comunidades de corales y octocorales de Isla de Aves, Venezuela, Caribe Nororiental. Revista de Biología Tropical, 62, 338-345.

Yranzo-Duque, A., Herrera-Reveles, A. T., Villamizar, E., Cabada-Blanco, F., Pérez-Benítez, J., Boadas, H., ... & Bustillos, F. A. (2025). Revealing the status of Orbicella: Main reef-builder of Morrocoy National Park and Cuare Wildlife Refuge, Venezuela, Southern Caribbean. PloS one, 20(2), e0317728.

Zarco-Perelló, S., Mascaró, M., Garza-Pérez, R., & Simoes, N. (2013). Topography and coral community of the Sisal Reefs, Campeche Bank, Yucatán, México. Hidrobiológica, 23(1), 28-41.

Zhang, W., Ok, Y. S., Bank, M. S., & Sonne, C. (2023). Macro-and microplastics as complex threats to coral reef ecosystems. Environment international, 174, 107914.

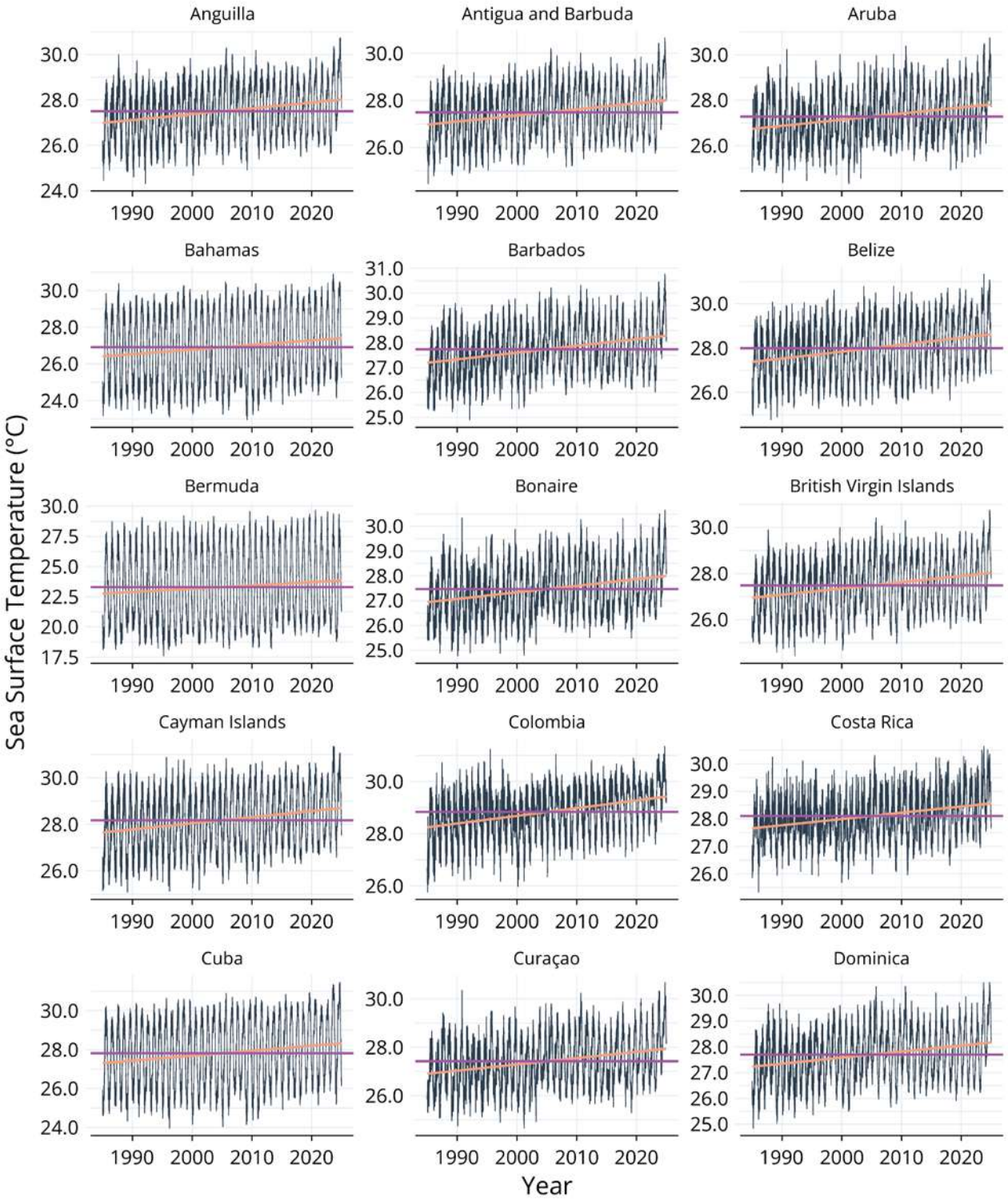
Zhao, H., Zhao, K., Klotzbach, P. J., Chand, S. S., Camargo, S. J., Cao, J., & Wu, L. (2024). Decreasing global tropical cyclone frequency in CMIP6 historical simulations. Science Advances, 10(27), eadl2142.

Zhao, H., Yuan, M., Strokal, M., Wu, H. C., Liu, X., Murk, A., ... & Osinga, R. (2021). Impacts of nitrogen pollution on corals in the context of global climate change and potential strategies to conserve coral reefs. Science of the Total Environment, 774, 145017.

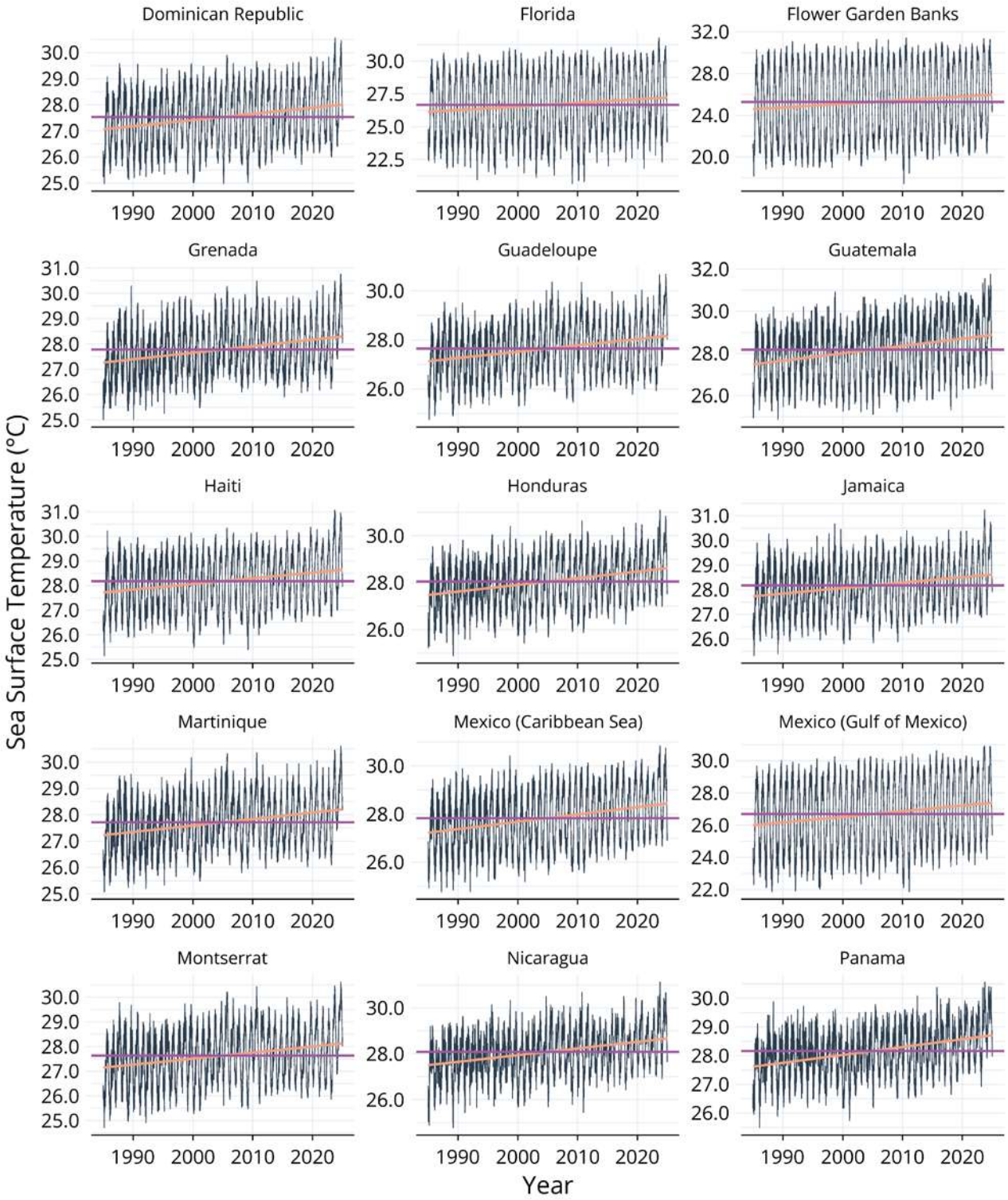
Zirler, R., Leck, L. A., Farkash, T. F., Holzknecht, M., Kroh, A., Gerovasileiou, V., ... & Bronstein, O. (2023). Gaining a (tube) foothold–trends and status following two decades of the long-spined echinoid Diadema setosum (Leske, 1778) invasion to the Mediterranean Sea. Frontiers in Marine Science, 10, 1152584.

Zubillaga, A. L., Márquez, L. M., Cróquer, A., & Bastidas, C. (2008). Ecological and genetic data indicate recovery of the endangered coral Acropora palmata in Los Roques, Southern Caribbean. Coral Reefs, 27(1), 63-72.

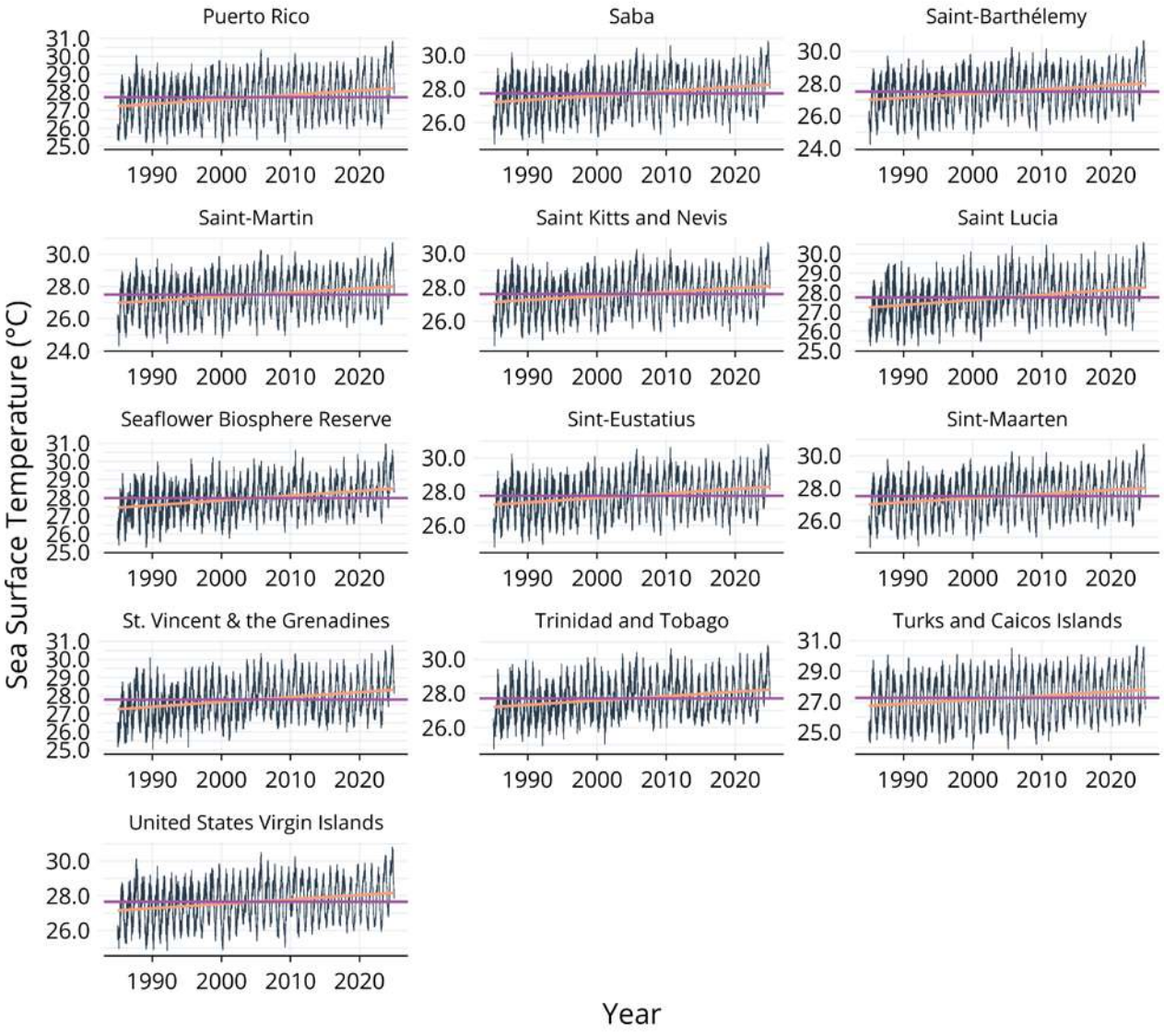
SUPPLEMENTARY MATERIALS



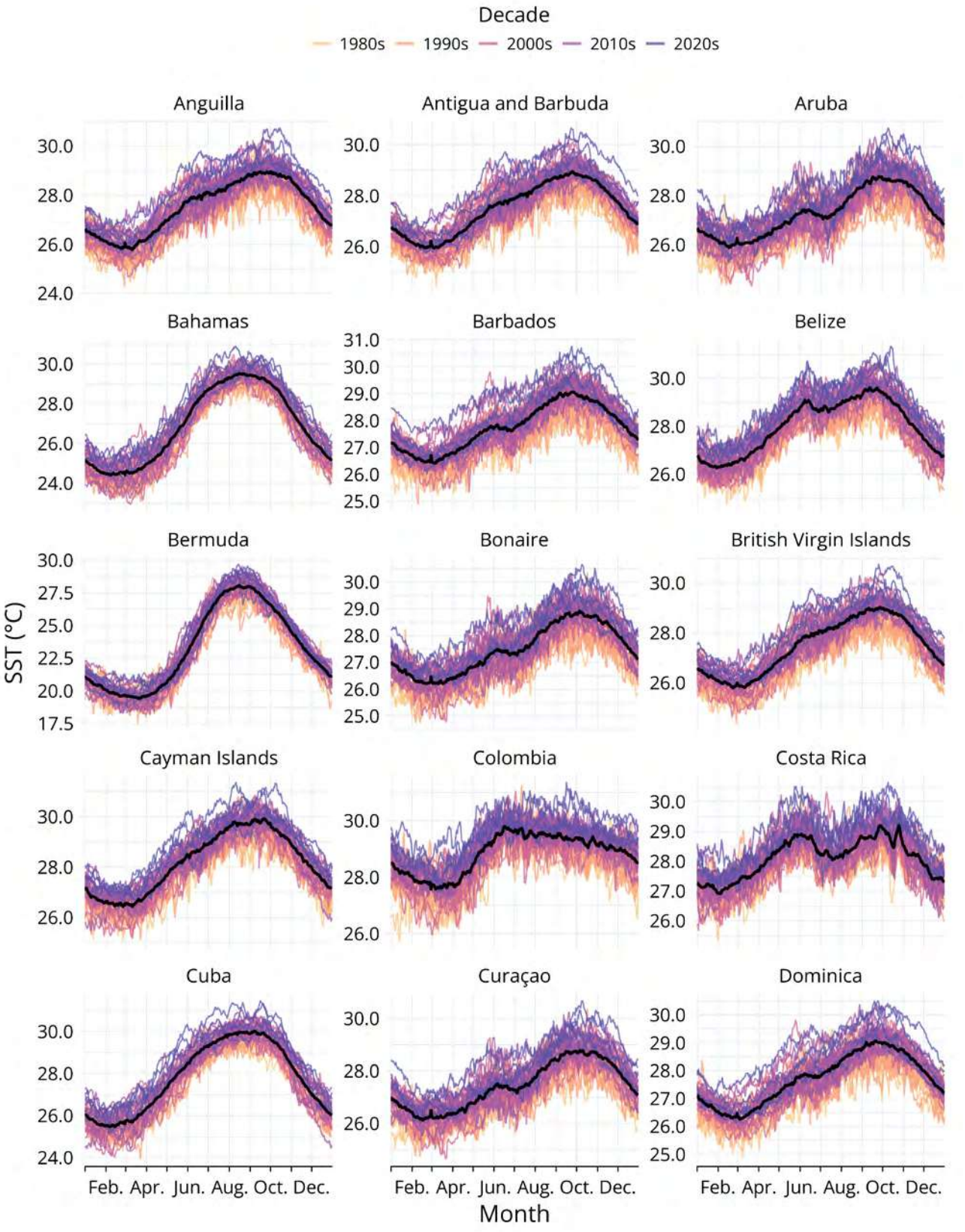
↑ **Supplementary Figure 1A.** Average daily sea surface temperature (SST) from 1985 to 2024 over coral reefs of countries and territories of the Caribbean. Purple lines represent the average SST from over the 1985 to 2024 period, whereas the orange line represents the long-term trend.



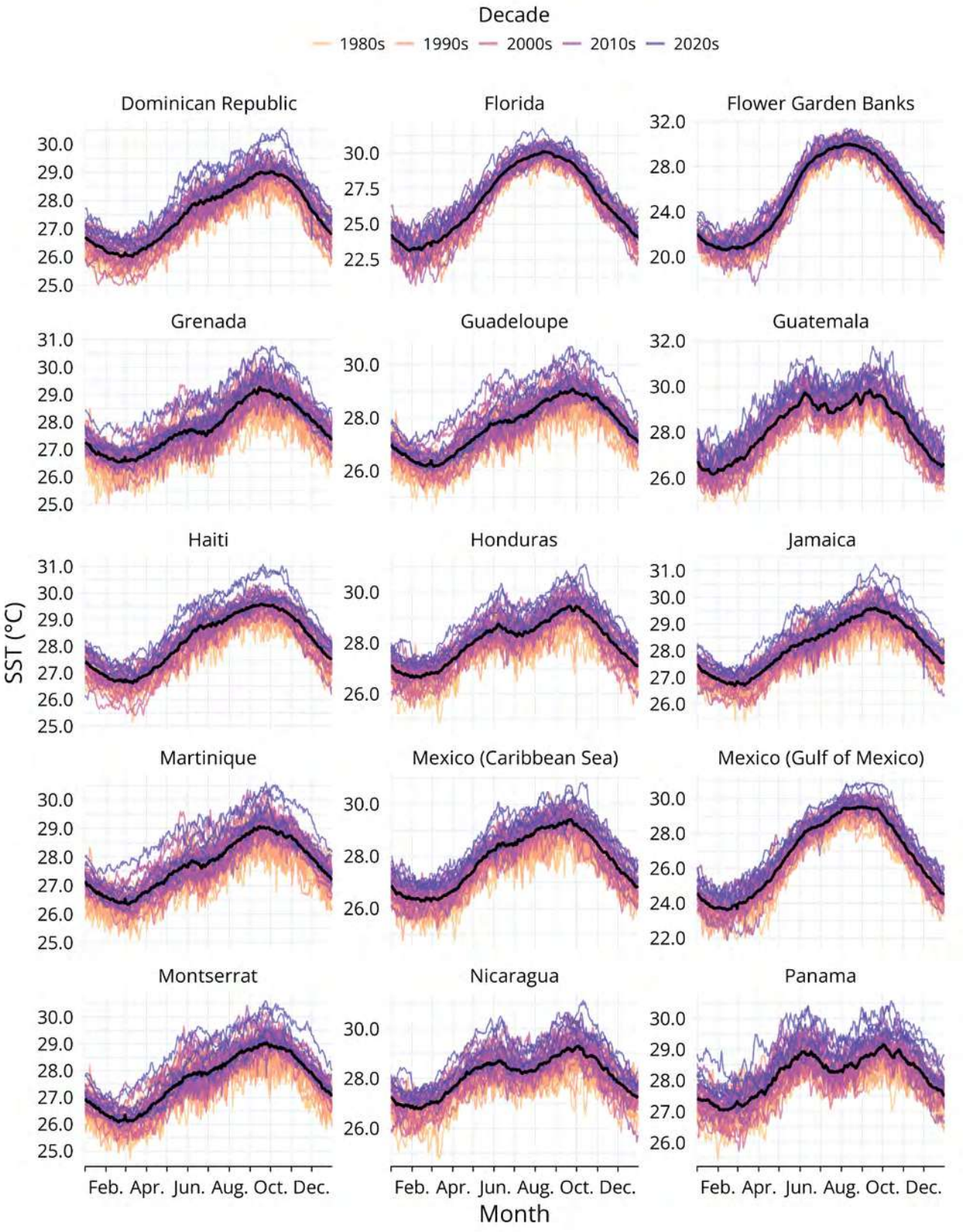
↑ **Supplementary Figure 1B.** Average daily sea surface temperature (SST) from 1985 to 2024 over coral reefs of countries and territories of the Caribbean. Purple lines represent the average SST from over the 1985 to 2024 period, whereas the orange line represents the long-term trend.



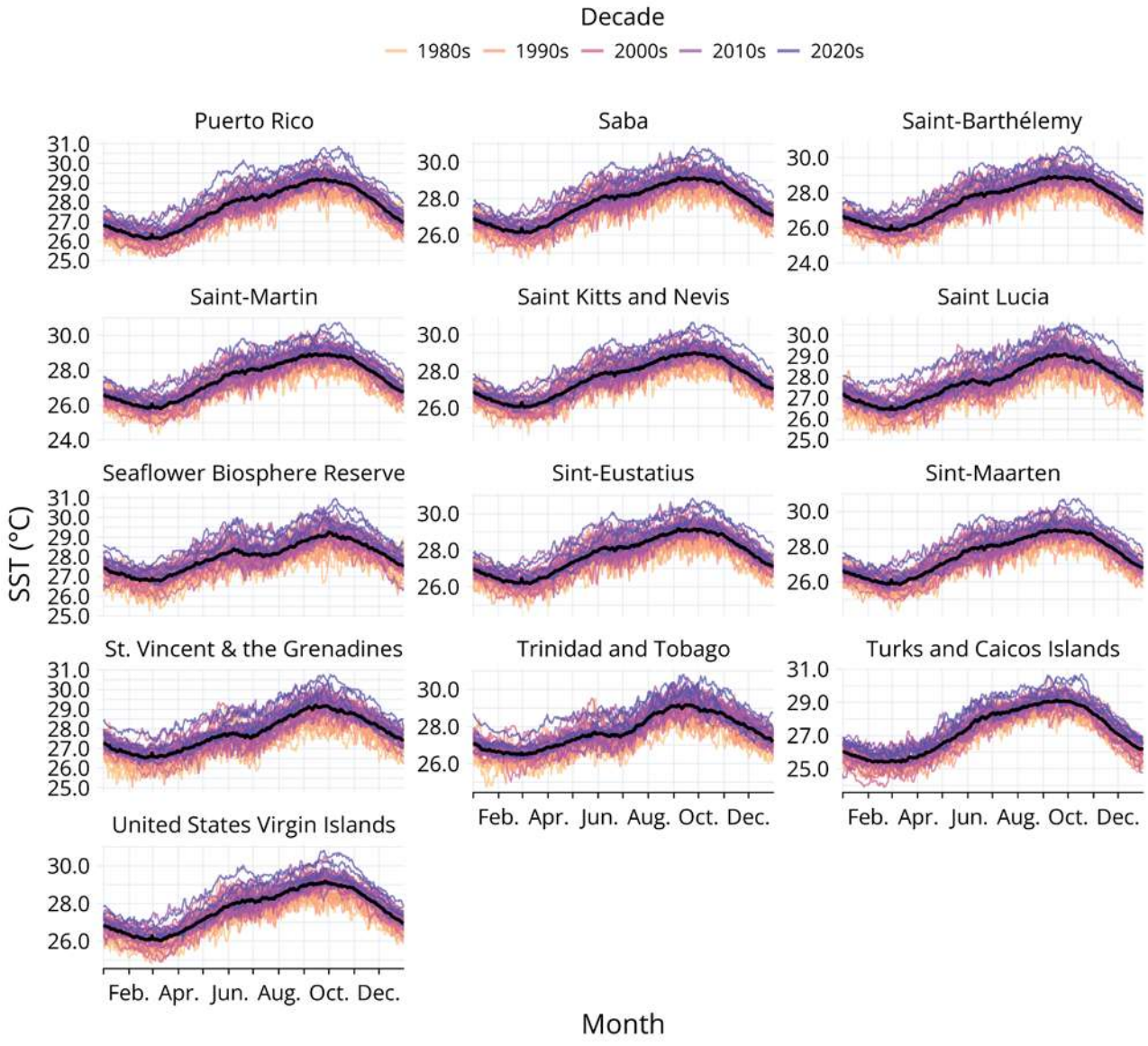
↑ **Supplementary Figure 1C.** Average daily sea surface temperature (SST) from 1985 to 2024 over coral reefs of countries and territories of the Caribbean. Purple lines represent the average SST from over the 1985 to 2024 period, whereas the orange line represents the long-term trend.



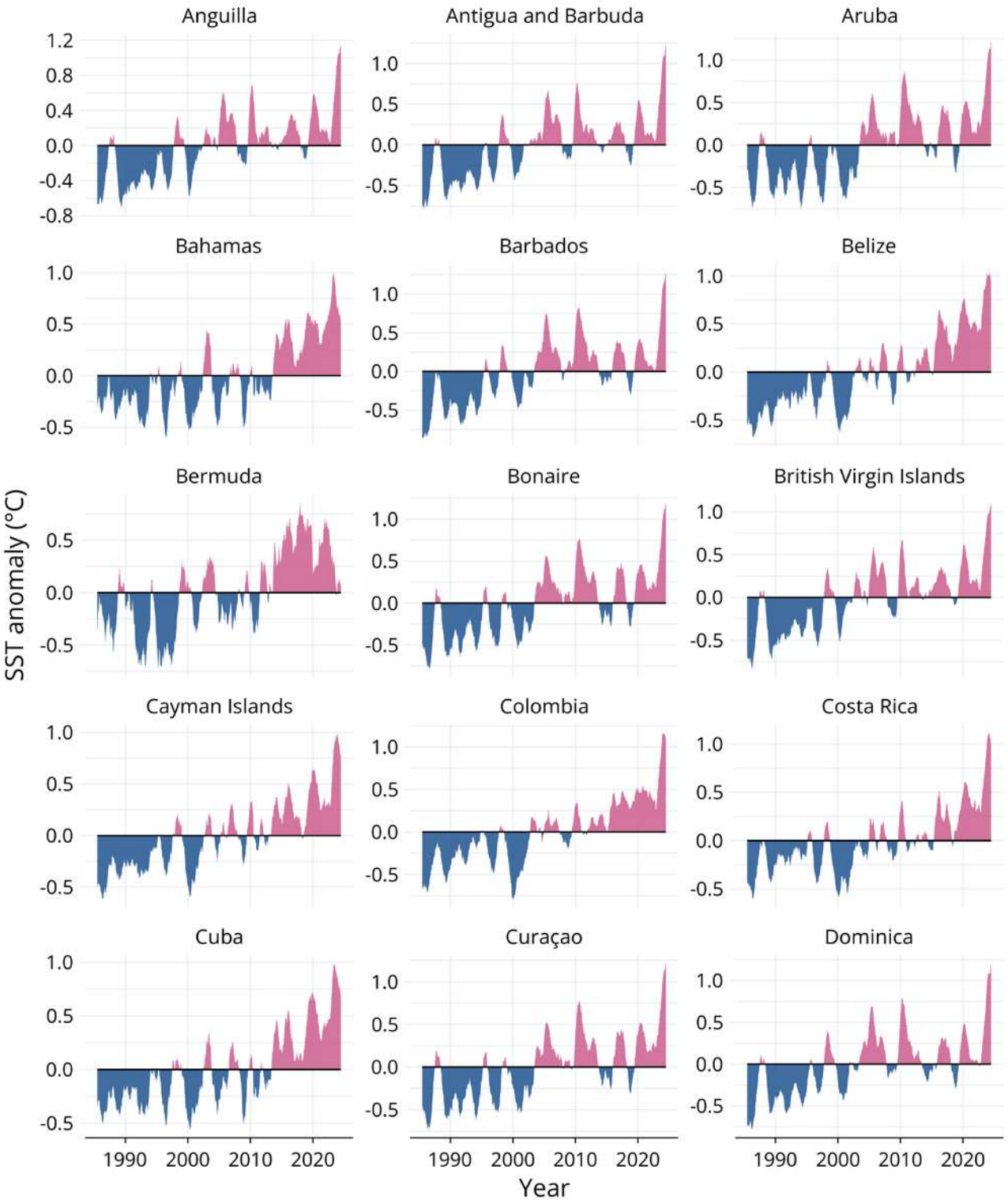
↑ **Supplementary Figure 2A.** Annual Sea Surface Temperature (SST) curves from 1985 to 2024 over coral reefs of countries and territories of the Caribbean. Each line, colored by its decade, corresponds to a unique year. The black line represents the interannual average SST from 1985 to 2024.



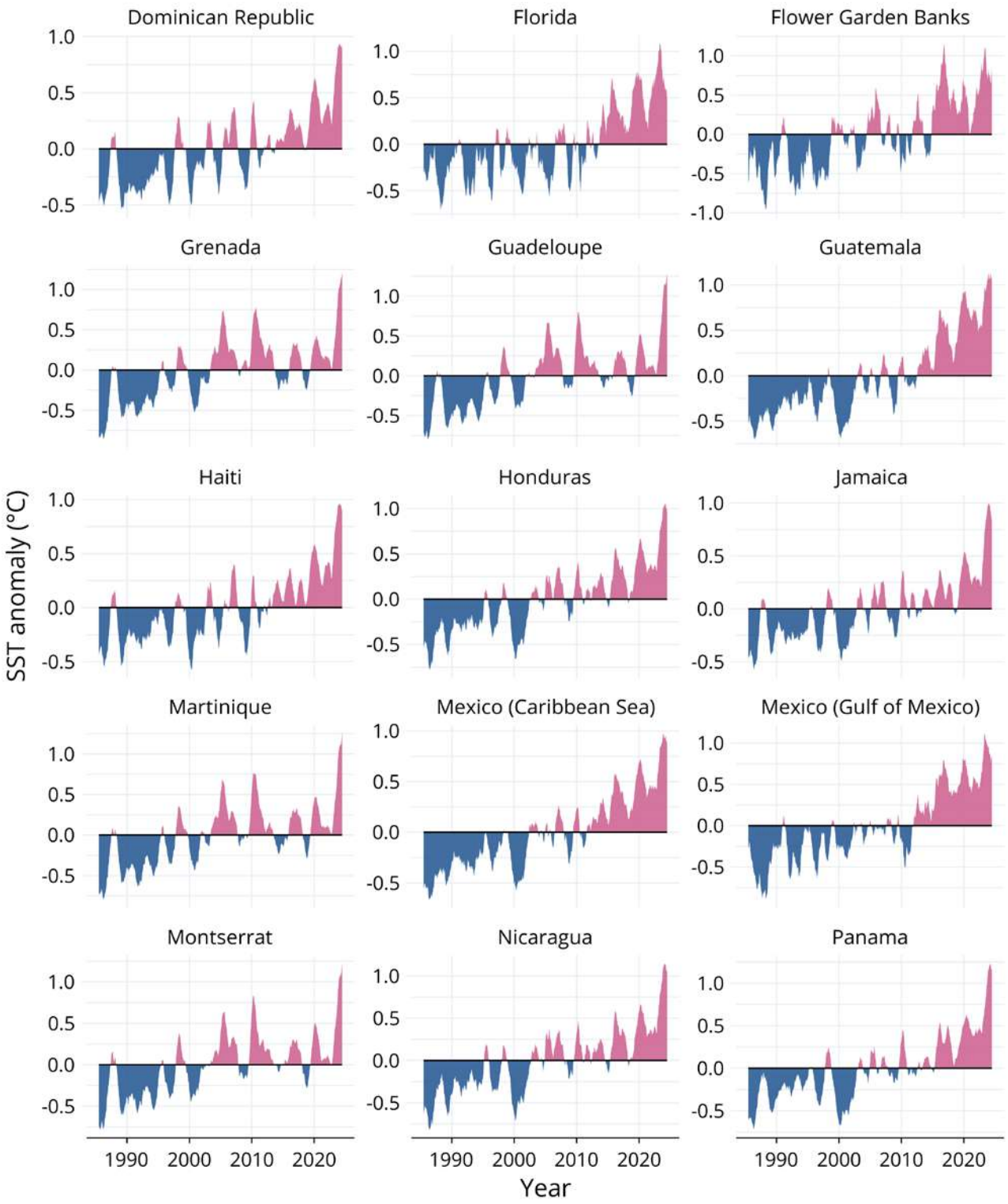
↑ **Supplementary Figure 2B.** Annual Sea Surface Temperature (SST) curves from 1985 to 2024 over coral reefs of countries and territories of the Caribbean. Each line, colored by its decade, corresponds to a unique year. The black line represents the interannual average SST from 1985 to 2024.



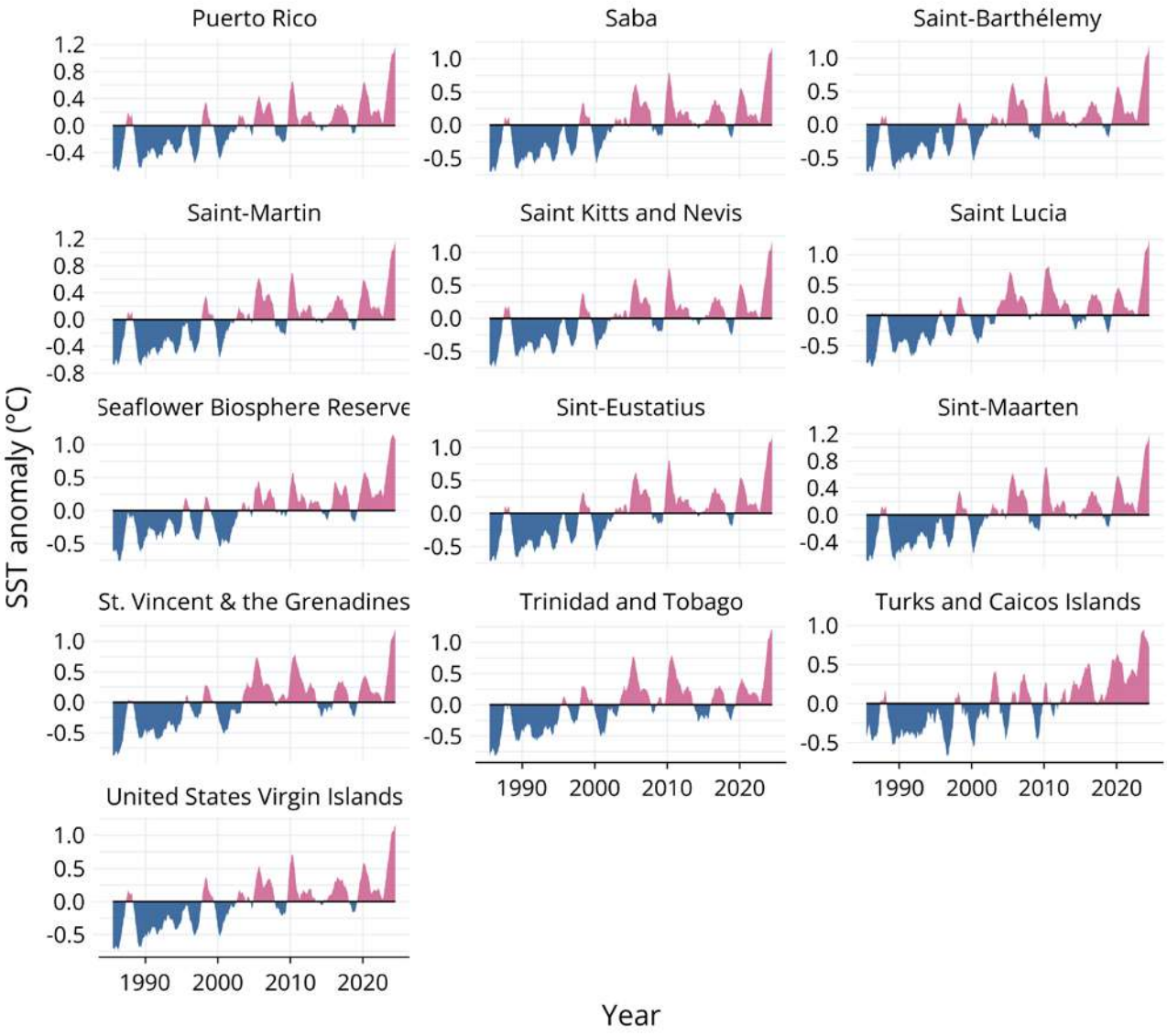
↑ **Supplementary Figure 2C.** Annual Sea Surface Temperature (SST) curves from 1985 to 2024 over coral reefs of countries and territories of the Caribbean. Each line, colored by its decade, corresponds to a unique year. The black line represents the interannual average SST from 1985 to 2024.



↑ **Supplementary Figure 3A.** Average Sea Surface Temperature (SST) anomaly from 1985 to 2024 over coral reefs of countries and territories of the Caribbean. The black line represents a null SST anomaly; values below this line are negative SST anomalies (*i.e.*, cooler than the long-term average), and values above this line are positive SST anomalies (*i.e.*, warmer than the long-term average).



↑ **Supplementary Figure 3B.** Average Sea Surface Temperature (SST) anomaly from 1985 to 2024 over coral reefs of countries and territories of the Caribbean. The black line represents a null SST anomaly; values below this line are negative SST anomalies (*i.e.*, cooler than the long-term average), and values above this line are positive SST anomalies (*i.e.*, warmer than the long-term average).



↑ **Supplementary Figure 3C.** Average Sea Surface Temperature (SST) anomaly from 1985 to 2024 over coral reefs of countries and territories of the Caribbean. The black line represents a null SST anomaly; values below this line are negative SST anomalies (*i.e.*, cooler than the long-term average), and values above this line are positive SST anomalies (*i.e.*, warmer than the long-term average).

↓ **Supplementary Table 1.** List of individual datasets per country and territory used for the estimation of benthic cover. The correspondences of each datasetID are provided in Table 5 of the gcrmndb_benthos.

Country or territory	datasetID
Anguilla	0015, 0267
Antigua and Barbuda	0015, 0091
Aruba	0015
Bahamas	0015, 0091, 0133
Barbados	0015, 0092, 0133
Belize	0015, 0038, 0091, 0133, 0154, 0155, 0270
Bermuda	0085, 0180
Bonaire	0015, 0038, 0062, 0091, 0098, 0099, 0146
British Virgin Islands	0015, 0091, 0152
Cayman Islands	0015, 0038, 0088, 0091, 0115
Colombia	0015, 0094, 0109, 0110
Costa Rica	0091, 0095
Cuba	0015, 0091, 0117
Curaçao	0015, 0084, 0091, 0099
Dominica	0015, 0091
Dominican Republic	0015, 0078, 0080, 0086, 0091, 0145, 0153, 0166
Florida	0015, 0038, 0091, 0097, 0104, 0106, 0107, 0133
Flower Garden Banks	0091, 0101, 0151, 0163, 0164, 0179
Grenada	0015, 0091, 0108, 0270
Guadeloupe	0015, 0047, 0087, 0096
Guatemala	0015, 0091, 0270
Haiti	0015, 0091
Honduras	0015, 0091, 0176, 0270
Jamaica	0015, 0081, 0091, 0111, 0133, 0153, 0181
Jamaica	0015, 0081, 0091, 0111, 0133, 0153, 0181
Martinique	0015, 0067, 0148
Mexico (Caribbean Sea)	0015, 0091, 0113, 0153, 0176, 0235, 0270
Mexico (Gulf of Mexico)	0091, 0113, 0135
Montserrat	0015, 0089
Navassa Island	0091
Nicaragua	0091
Panama	0015, 0038, 0060, 0091
Puerto Rico	0015, 0091, 0093, 0102, 0133, 0151, 0178
Saba	0082, 0091, 0138
Saint Kitts and Nevis	0015, 0091
Saint Lucia	0015
Saint Vincent and the Grenadines	0015, 0091, 0147, 0270
Saint-Barthélemy	0015, 0096
Saint-Martin	0015, 0083
Seaflower Biosphere Reserve	0015, 0038, 0094, 0110
Sint-Eustatius	0015, 0091, 0114
Sint-Maarten	0015, 0066, 0091
Trinidad and Tobago	0015, 0130, 0149
Turks and Caicos Islands	0015, 0091, 0116, 0165
United States Virgin Islands	0001, 0002, 0015, 0079, 0091, 0103, 0133, 0151, 0178, 0181
Venezuela	0015, 0090, 0091

↓ **Supplementary Table 2.** Comparison of the average Root Mean Squared Error (RMSE) and coefficient of determination (R^2) across 20 bootstrap iterations of the machine learning models used to estimate the percentage cover of hard coral and macroalgae categories for each country and territory.

Country or territory	Hard coral		Macroalgae	
	RMSE	R^2	RMSE	R^2
Anguilla	8.77	0.11	12.35	0.3
Antigua and Barbuda	8.14	0.3	11.19	0.32
Aruba	17.42	0.1	4.5	
Bahamas	8.73	0.38	18.14	0.47
Barbados	8.45	0.34	8.48	0.38
Belize	9.45	0.33	12.6	0.45
Bermuda	8.83	0.56	9.59	0.64
Bonaire	13.37	0.38	8.8	0.49
British Virgin Islands	10.03	0.48	6.81	0.31
Cayman Islands	9.51	0.22	18.22	0.46
Colombia	16.62	0.4	13.32	0.5
Costa Rica	13.31	0.39	18.02	0.49
Cuba	11.22	0.28	9.19	0.14
Curaçao	9.38	0.55	5.83	0.37
Dominica	10.71	0.36	4.6	0.25
Dominican Republic	9.28	0.5	13.57	0.45
Florida	5.64	0.59	11.55	0.51
Flower Garden Banks	15.32	0.27	7.51	0.42
Grenada	10.2	0.34	12.75	0.32
Guadeloupe	8.7	0.18	12.66	0.29
Guatemala	13.88	0.39	15.41	0.35
Haiti	10.01	0.25	14.44	0.62
Honduras	12.89	0.53	15.3	0.39
Jamaica	8.75	0.41	15.09	0.52
Martinique	12.4	0.49	13.16	0.62
Mexico (Caribbean Sea)	9.32	0.35	12.2	0.47
Mexico (Gulf of Mexico)	12.05	0.39	13.47	0.49
Montserrat	5.64	0.15	8.02	
Nicaragua	8.69	0.54		
Panama	13.75	0.47	4.33	0.05
Puerto Rico	7.98	0.53	13.05	0.46
Saba	6.36	0.6	11.5	0.5
Saint Kitts and Nevis	5.93	0.15	16.13	0.54
Saint Lucia	10.33	0.25	4.92	0.34
Saint Vincent and the Grenadines	10.09	0.54	16.4	0.33
Saint-Barthélemy	5.94	0.24	13.4	0.18
Saint-Martin	6.86	0.15	16.48	0.24
Seaflower Biosphere Reserve	10.82	0.4	18.64	0.42
Sint-Eustatius	6.35	0.72	10.05	0.45
Sint-Maarten	5.32	0.3	12.1	0.21
Trinidad and Tobago	9	0.29	7.06	0.56
Turks and Caicos Islands	6.41	0.55	9.02	0.54
United States Virgin Islands	6.61	0.59	13.93	0.55
Venezuela	12.48	0.21	10.07	0.36

GCRMN

Global Coral Reef Monitoring Network

Global Coral Reef Monitoring Network
GCRMN Caribbean
gcrmn.net/caribbean