A decade of change on Guam's coral reefs

A report of Guam Long-term Coral Reef Monitoring Program activities between 2010 and 2021

Prepared by David Burdick, M.S.
Principle Investigator
Research Associate, University of Guam Marine Laboratory

with contributions by

Laurie Raymundo, Ph.D.
Director
University of Guam
Marine Laboratory

Diona Drake, M.S.
Research Associate
University of Guam
Marine Laboratory

Andrea Hershberger, B.S.
Research Assistant
University of Guam
Marine Laboratory

and the support of multiple current and previous Guam Long-term Coral Reef Monitoring Program and Raymundo Lab technicians

University of Guam Marine Laboratory Technical Report 170

August 2023

A project currently funded by NOAA Coral Reef Conservation Program cooperative agreement NA21NOS4820018, with previous funding provided through grants NA07NOS4260060, NA10NOS4260060, NA11NOS4820007, NA13NOS4820012, NA15NOS4820039, NA17NOS4820038, and NA19NOS4820057

A decade of change on Guam's coral reefs

A report of Guam Long-term Coral Reef Monitoring Program activities between 2010 and 2021

Prepared by David Burdick, M.S.
Principle Investigator
Research Associate, University of Guam Marine Laboratory

with contributions by

Laurie Raymundo, Ph.D.
Director, University of Guam Marine Laboratory

Diona Drake, M.S.
Research Associate, University of Guam Marine Laboratory

and

Andrea Hershberger, B.S.
Research Assistant, University of Guam Marine Laboratory

University of Guam Marine Laboratory Technical Report 170

August 2023

The Comprehensive Long-term Coral Reef Monitoring at Permanent Sites on Guam project is currently funded by NOAA Coral Reef Conservation Program grant NA21NOS4820018. Previous funding was provided through grants NA07NOS4260060, NA10NOS4260060, NA11NOS4820007, NA13NOS4820012, NA15NOS4820039, NA17NOS4820038, and NA19NOS4820057.

Datasets utilized in this report were obtained by several additional individuals who worked as technicians or monitoring assistants with the GLTMP, including Roxanna Miller, Mari Deinhart, Jordan Gault, Jacques Idechong, Travis Reynolds, Mark Priest, Alyssa Marshell, Pablo Rojas, Roxie Dias, and Paula Lozada. Several graduate students and technicians with the Raymundo Lab have assisted Dr. Raymundo with data collection at the reef flat monitoring sites and with the staghorn mortality assessments; these individuals include Maria Andersen, Nicole Burns, Ashley Castro, Renee Cristostomo, Colin Lock, Farron Taijeron, and Ashton Williams. Special acknowledgement is also owed to Valerie Brown (NOAA), who served as the fish team lead between 2009 and 2019 and provided significant guidance and support to GLTMP efforts since the program's inception. Other members of the Guam Rapid Response Team also provided assistance with island-wide bleaching response surveys, including staff from the Guam Bureau of Statistics and Plans, Guam EPA, and Guam Dept. of Agriculture's DAWR.

Coral bullet point icon by Kerismaker Studio on IconScout, fish bullet point icon by Those Icons, and clam bullet point icon by Mario Bieh on Noun Projects

***Cover**: Bleached coral, mainly Acropora abrotanoides, in Pago Bay during the historically severe coral bleaching event in 2017.* ***Next page**: High structural complexity and high cover of the plate and pillar coral, Porites rus, along the submarine terrace in the East Agana Bay HPRA in 2020.*

ABOUT THIS REPORT

This report provides an overview of activities carried out under the *Comprehensive Long-term Monitoring at Permanent sites on Guam project*, also known as the Guam Long-term Coral Reef Monitoring Program (GLTMP), and presents the results of analyses of data collected by the GLTMP and the affiliated UOG Reef Flat Monitoring Program since 2010 and 2009, respectively. The analyses addressed here are necessarily limited in scope, primarily focused on trends in broad level coral reef ecosystem health indicators, but on-going collaborations with University of Guam Marine Lab researchers, NOAA Pacific Islands Fisheries Science Center researchers and others will continue to yield additional, more detailed information using data collected by the GLTMP and its partners.

Moderate structural complexity but relatively low cover of living coral at a sampling station within the Cocos-East HPRA in 2021. While historical data are limited for this area, data collected by NOAA PIFSC and the GLTMP around the island showed significant declines in the amount live coral along the windward coast between 2003 and 2017. These declines were likely a result of predation by crown of thorns sea stars during the mid-2000s and, more recently, as a result of coral bleaching-associated mortality. Recovery in these areas is likely hindered by continued predation by sea stars, poor water quality, limited coral recruitment, and depauperate herbivorous reef fish populations.

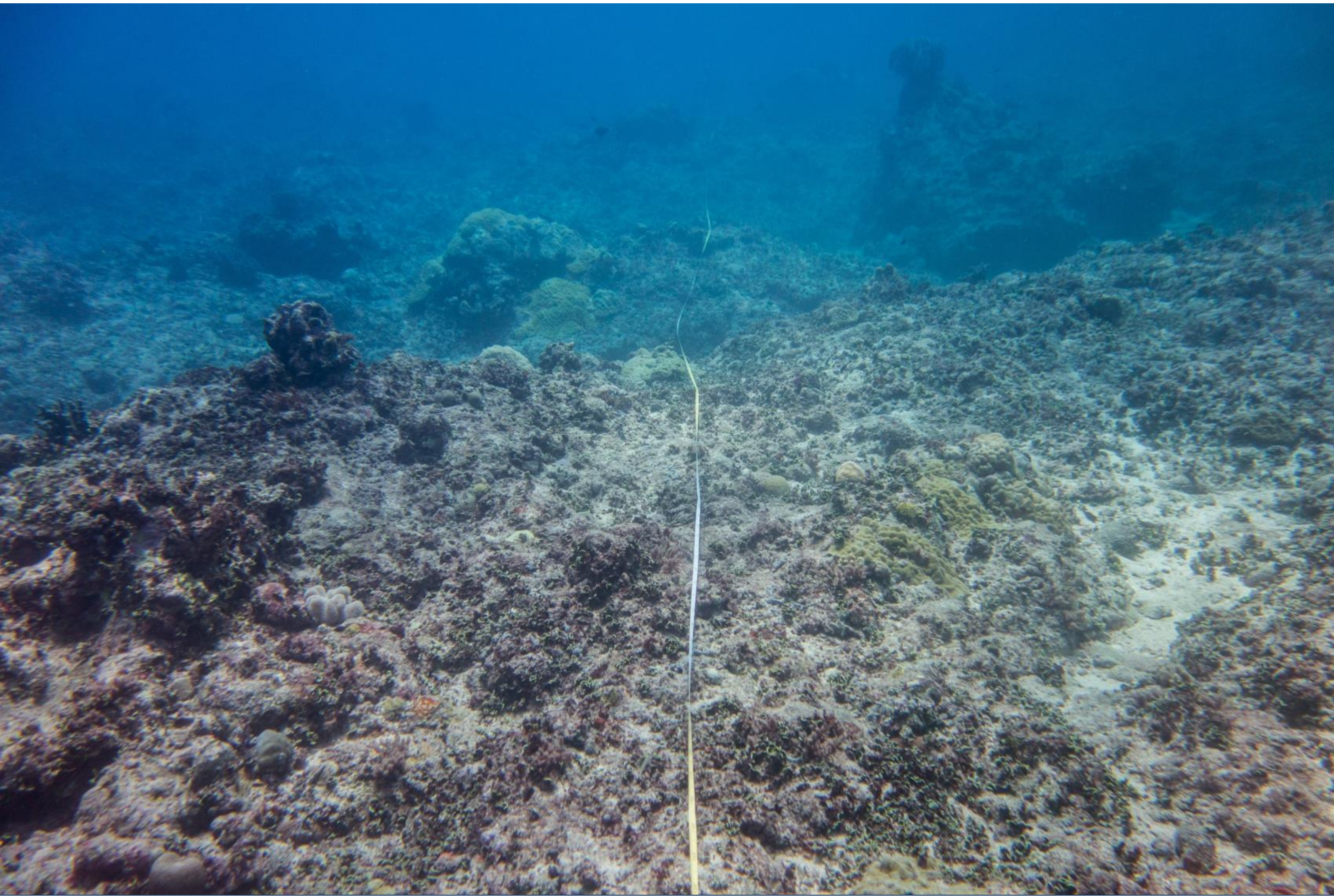
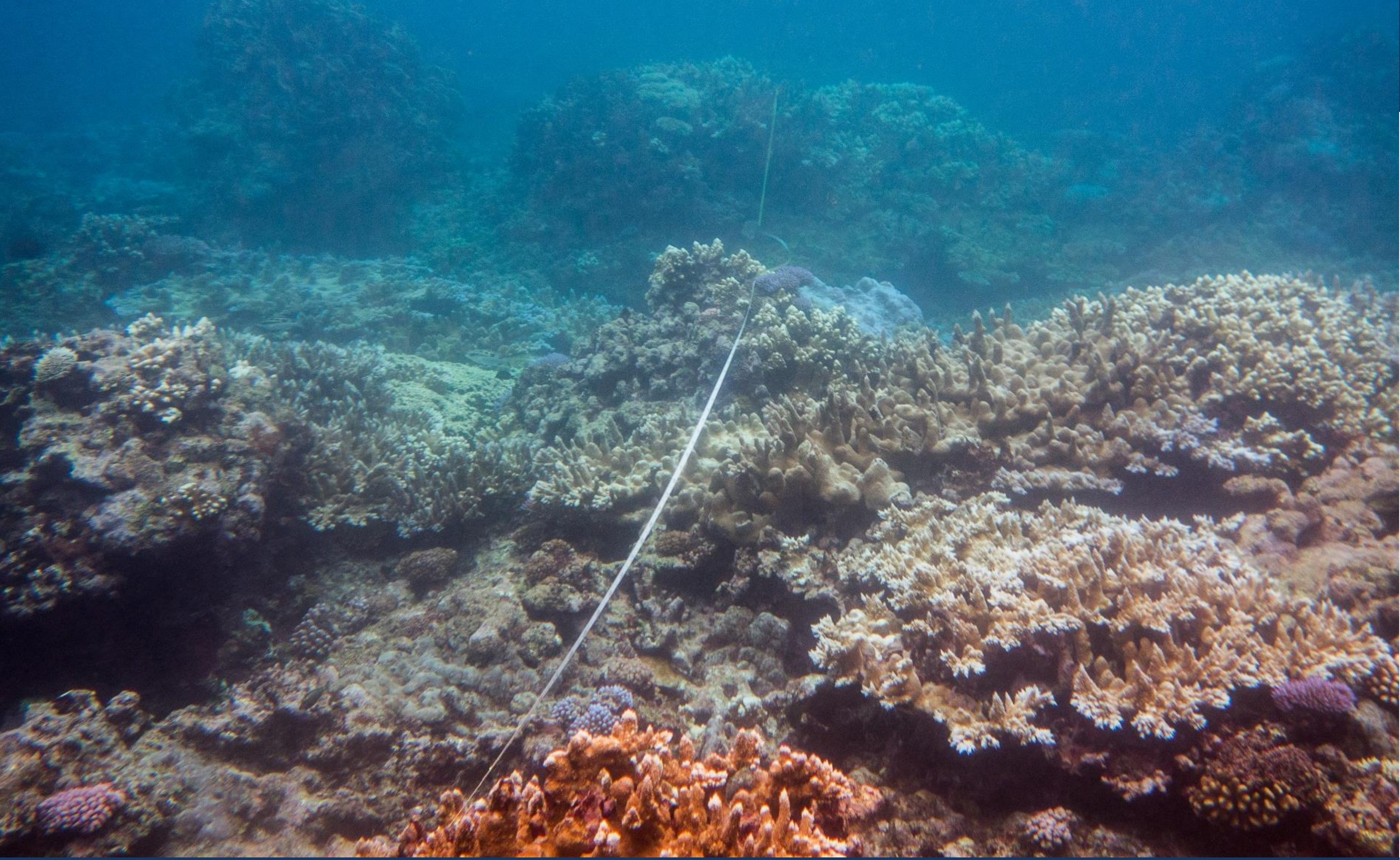
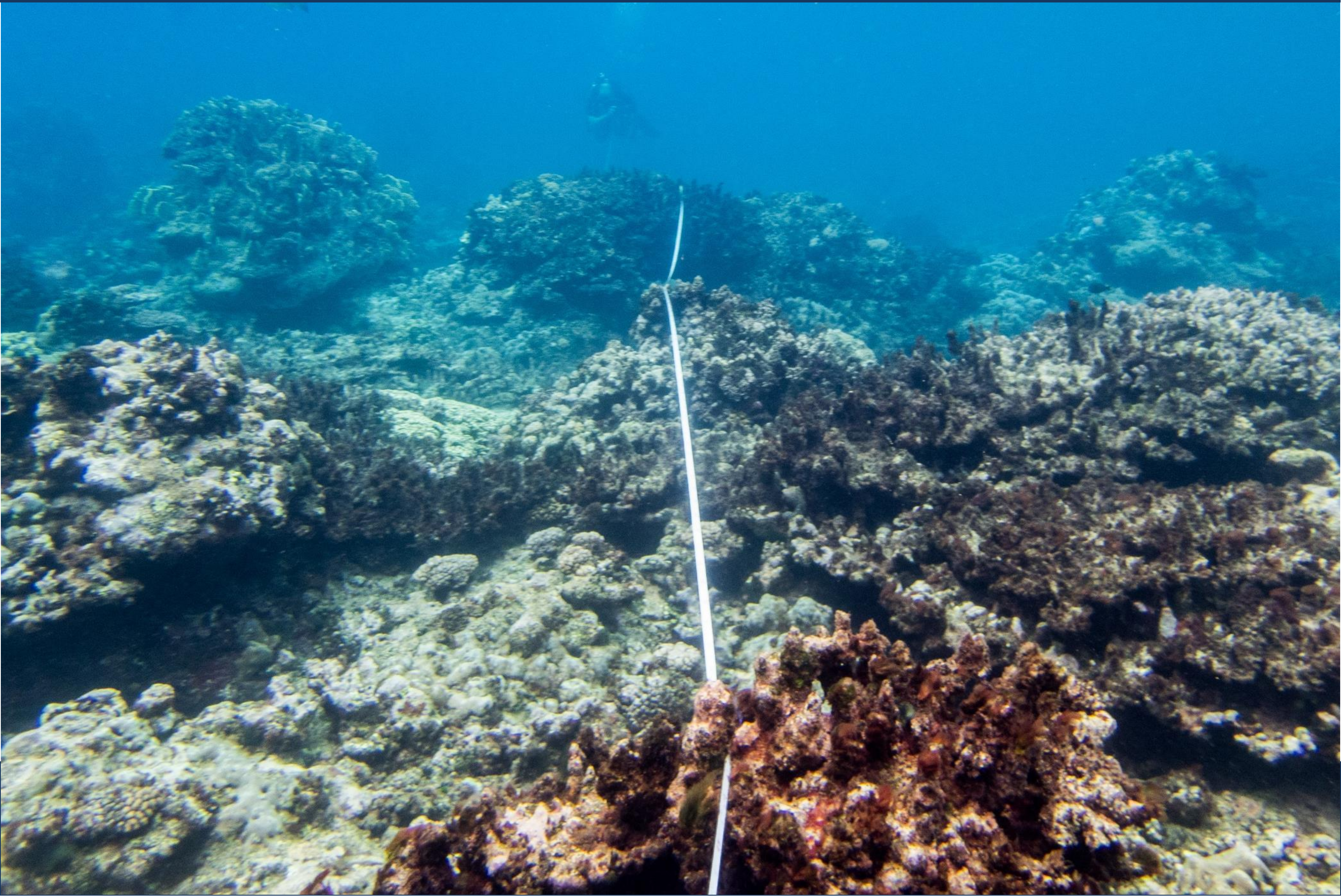


TABLE OF CONTENTS

Next page*: A shallow (2.5 m) wave-exposed reef front community with numerous live Acropora abrotanoides colonies in Pago Bay in August 2017, at the beginning of a record-breaking coral bleaching event (top), and that same community in April 2018, several months after temperatures returned to normal and after most of the corals had died (bottom). Data obtained at the site at the beginning of the 2017 bleaching event and a few months after the event show that live coral cover declined by 54%, from 39% to 18% (t.test, p = 0.02), with most of the loss observed in A. abrotanoides and bushy Acropora spp.; these species suffered near 100% mortality. Coral cover increased slightly, from 18% to 21% (t.test, p = 0.03) between 2018 and 2021, showing that at least some recovery occurred during this period. While more bushy Acropora colonies can be found at the site now than in early 2018, their abundances are still relatively low, and more stress-tolerant—but generally slower-growing and less structurally complex—taxa, such as Goniastrea retiformis, Leptoria phrygia, and Pocillopora spp. are now the dominant coral taxa at the site.*





EXECUTIVE SUMMARY

The *Comprehensive Long-term Monitoring at Permanent Sites in Guam* project, also known as the Guam Long-term Coral Reef Monitoring Program, or GLTMP, involves the regular, intensive collection of data for a suite of coral reef ecosystem health parameters at priority reef areas around Guam, as well as critical support for coral bleaching response and other activities carried out by the multi-partner Guam Coral Reef Response Team. This project also includes support for the Reef Flat Monitoring Program (RFMP), an effort led by Dr. Laurie Raymundo of the University of Guam Marine Laboratory (UOGML) to document changes in coral communities on Guam's shallow reef flats.

The High Priority Reef Areas (HPRAs) selected for monitoring by a group of local and federal agency partners include portions of the Tumon Bay Marine Preserve, East Agana Bay, the Piti Bomb Holes Marine Preserve, the Achang Reef Flat Marine Preserve, the east side of the Cocos Barrier Reef (Cocos-East), and Fouha Bay. Surveys at the HPRAs are carried out along the submarine terrace zone of the seaward slope, at depths between seven and 15 m (23–49 ft). Data collection at the HPRAs began in 2010, and since then reef fish, benthic, and macroinvertebrate survey data has been collected at six sites on a regular basis. The initial focus of the GLTMP was on the HPRAs, but a historically severe mass coral bleaching event in 2013, and subsequent bleaching events in 2014, 2016, and 2017, necessitated a significant shift of resources to island-wide data collection efforts that targeted the shallow reef communities most severely impacted by these events.

Members of the GLTMP have also made essential contributions to the mapping, assessment, and monitoring of Guam's staghorn coral communities; to an island-wide reef resilience assessment in 2016; and to a variety of other monitoring, assessment, outreach, and management-related activities. The program coordinator, as well as other members of the monitoring team, have also provided key input into the development of management strategies, such as the Guam Coral Bleaching Response Plan and the Guam Reef Resilience Strategy, and are currently participating in the development of the Guam Coral Reef Restoration Strategy and the Guam Fisheries Management Plan.

Much of the change in reef condition documented by the GLTMP and its partners over the last decade has occurred within the shallow, wave-washed reef front zone, as well as on Guam's shallow reef flat platforms. These areas hosted much larger numbers of branching corals and other bleaching-susceptible corals prior to the onset of severe bleaching, whereas those corals had become uncommon along much of Guam's seaward slope as a result of decades of crown of thorns sea star outbreaks and limited recovery due to various local stressors. Island-wide surveys carried out by GLTMP staff and other members of the Guam Coral Reef Response Team documented a 34% decline in living coral along the reef front zone as a result of the record-breaking levels of heat stress between 2013 and 2017. Impacts were most severe for coral communities on the eastern side of the island, which suffered an estimated 59% loss in living coral during this period. Recovery following the bleaching events has been mixed, with 60% of reef front sites showing increases in coral cover through 2021 and 40% of sites showing decreases during this period.

All but one of the shallow reef flat sites monitored regularly by the RFMP showed declines in coral cover between 2009 and 2022, with an average loss of 24% during this period. Sites with a greater proportion of the more bleaching-susceptible staghorn corals generally experienced greater losses, with one site losing as much as 50% of its living coral. A series of island-wide staghorn coral mortality assessments, also led by the Raymundo Lab, found a ~60% decline in the extent of living staghorn coral island-wide between 2013 and 2020/21, with some sites losing 100% of their staghorn corals, and all but one species now considered uncommon or rare in Guam's waters.

59%

Decline in living coral at east coast reef front sites 2013–2017

34%

Decline in living coral at reef front sites island-wide 2013–2017

60%

Percentage of reef front sites with increases in coral cover 2017–2021

24%

Decline in living coral at reef flat monitoring sites 2009–2022

~60%

Decline in extent and cover of living staghorn coral island-wide 2013-2020/21

In contrast to the catastrophic loss of bleaching-susceptible corals in the shallower reaches of Guam’s reefs, coral cover along the seaward slope within the HPRAs remained relatively stable throughout the 2013–2017 bleaching events, although modest declines were detected at some of the HPRAs during or after this period. The relative stability of the moderate-to-high levels of live coral cover at the Tumon, East Agana, and Piti HPRAs is likely a result of the near complete dominance of the coral community at these sites by stress-resistant corals such as *Porites rus* and mounding *Porites* species. The HPRAs selected for monitoring happened to include these higher coral cover areas, and were not intended to represent the entirety of Guam’s coral reef ecosystem. The most recent (2022) data collected around the island by NOAA PIFSC showed average coral cover at ~14% along the seaward slope, ranging from 3% in the southwest to 19% in the northwest. Coral cover within the HPRAs in the south of the island, including the Fouha Bay, Achang, and Cocos-East HPRAs, remained very low (< 6%) throughout the study period. The low cover of living coral, and the high cover of cyanobacteria and sediment-laden algae at these HPRAs suggest that these sites had been significantly impacted prior to the commencement of monitoring at these sites, and recovery has been limited or absent.

Data gaps and interobserver bias limited the broad-level analysis of reef fish survey data carried out in this study, but there are indications that food fish biomass decreased significantly within the Tumon and East Agana HPRAs in the first few years of monitoring. Recent data collected by a single observer allows more confidence in the assertion that food fish biomass increased in the Tumon, East Agana, Achang, and Fouha HPRAs between 2019 and 2021, while it remained unchanged in the Piti and Cocos-East HPRAs. Total fish biomass for all of the HPRAs except the Tumon HPRA was low compared to the biomass expected for an unimpaired Guam reef system. The low reef fish biomass values for most of HPRAs suggest human impacts on these reef fish communities, but recent increases are encouraging. Due to the complex nature of reef fish communities and the challenges inherent in surveying them, it is recommended that the GLTMP reef fish data be considered along with other available datasets in a more robust analysis carried out by specialists.

0%

Change in live coral cover at the Tumon, East Agana, and Piti sites between 2010/12 and 2020

<6%

Cover of live coral at the Achang, Cocos-East, and Fouha Bay HPRAs between 2014/15 and 2018/19

6–77%

Proportion of reef fish biomass expected for a healthy Guam reef observed at the HPRAs

69–93%

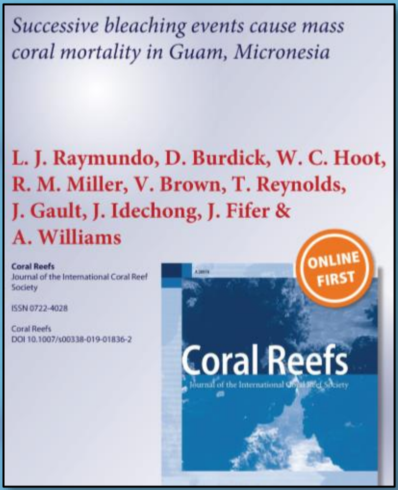
Decline in sea cucumber densities at 4 of the 6 HPRAS between 2012 and 2017

Sea cucumbers and edible shells (mainly top shells and giant clams) suffered significant (69–93%) losses in most of the HPRAs, likely as a direct or indirect result of the severe heat stress events. Anecdotal observations in 2014 suggest the decline in sea cucumber densities may have been caused by a marine disease that affected a single once-abundant species. The possible link between the apparent disease-related mass mortality of the spiky sea cucumber (*Stichopus chloronotus*) and ocean warming should be investigated further. Ocean warming is projected to increase over the coming decades and is expected to increase the virulence and prevalence of marine disease, potentially resulting in the mass mortality of sea cucumbers and other marine invertebrates that may be important sources of protein for coastal communities, or which may play key roles in maintaining coral reef function.

The results of this broad-level analysis of survey data collected by the GLTMP and its partners over the last decade show that Guam's reefs continue to shift toward coral communities dominated by a small number of generally stress-tolerant species, a trend seen throughout Micronesia and across the globe. This shift had already occurred across large portions of the seaward slope prior to the onset of severe coral bleaching, but the multiple heat stress events between 2013 and 2017 caused substantial losses of stress-susceptible corals in Guam's shallow reef areas, where these corals still thrived. While recent fish survey data showed positive trends at most of the HPRAs, the reef fish communities within these areas still appear to be impacted, likely by some combination of habitat degradation and fishing pressure. The copious survey data, as well as anecdotal observations made at numerous reef areas around the island, also demonstrate the complex, heterogeneous nature of Guam's reefs, and highlight the importance of recognizing the distinct reef communities associated with each reef zone and reef habitat, and how those distinct communities, even those within the same bay, may be exposed to different stressors or respond differently to the same stressors.

Other publications produced or supported by the GLTMP

Reports available at guamcoralreefs.info, guamcoralreefmonitoring.wordpress.com, and coris.noaa.gov.
Journal articles available upon request. A comprehensive list of publications is provided in Appendix B.


Successive bleaching events cause mass coral mortality in Guam, Micronesia

L. J. Raymundo, D. Burdick, W. C. Hoot, R. M. Miller, V. Brown, T. Reynolds, J. Gault, J. Idechong, J. Fifer & A. Williams

Coral Reefs
Journal of the International Coral Reef Society

ISSN 0722-4028

Coral Reefs
DOI 10.1007/s00338-019-01836-2


Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Changes in coral reef community structure along a sediment gradient in Fouha Bay, Guam

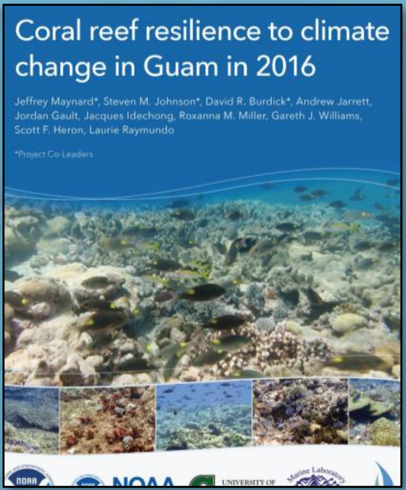
Dwayne Minton[a,*], David Burdick[b], Valerie Brown[c,1]

ARTICLE INFO

Keywords:
Coral reef
Guam
Sediment
Moving Window Analysis
Restoration
Gradient
Algae

ABSTRACT

1. Introduction


Coral reef resilience to climate change in Guam in 2016

Jeffrey Maynard*, Steven M. Johnson*, David R. Burdick*, Andrew Jarrett, Jordan Gault, Jacques Idechong, Roxanna M. Miller, Gareth J. Williams, Scott F. Heron, Laurie Raymundo

*Project Co-Leaders


An interim report of the *Comprehensive Long-term Coral Reef Monitoring at Permanent Sites on Guam* project

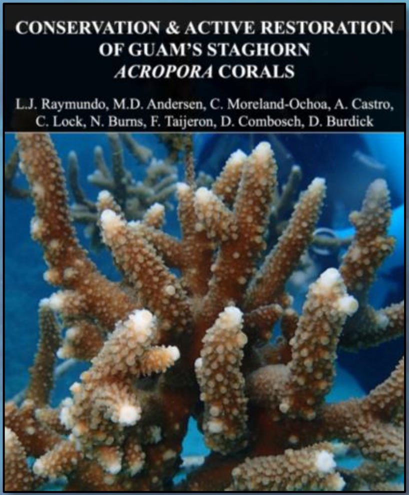
ncluding the results of an initial analysis of coral reef bleaching response survey dat 013–2017), data collected by the UOGML Reef Flat Monitoring Program (2009–2018 and a staghorn mapping and mortality assessment effort (2009–2017)

Prepared by David Burdick
Principle Investigator
Research Associate, University of Guam Marine Laboratory

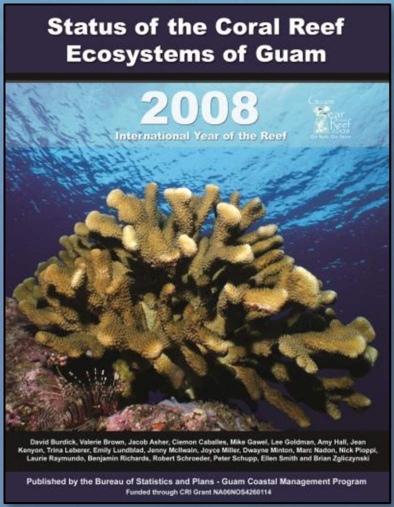
With contributions by
Dr. Laurie Raymundo, University of Guam Marine Laboratory


A report of the *Comprehensive Long-term Coral Reef Monitoring at Permanent Sites on Guam* project

Including the results of an analysis of data collected at high priority reef areas between 2010 and 2018


CONSERVATION & ACTIVE RESTORATION OF GUAM'S STAGHORN *ACROPORA* CORALS

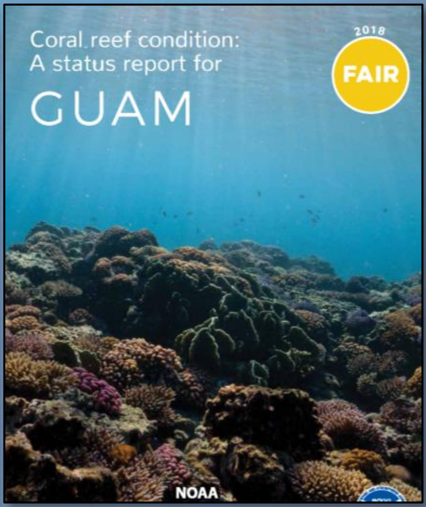
L.J. Raymundo, M.D. Andersen, C. Moreland-Ochoa, A. Castro, C. Lock, N. Burns, F. Taijeron, D. Combosch, D. Burdick


Status of the Coral Reef Ecosystems of Guam

2008
International Year of the Reef

David Burdick, Valerie Brown, Jacob Asher, Ciemon Caballes, Mike Gawel, Lee Goldman, Amy Hall, Jean Kenyon, Trina Leberer, Emily Lundblad, Jenny McIlwain, Joyce Miller, Dwayne Minton, Marc Nadon, Nick Pioppi, Laurie Raymundo, Benjamin Richards, Robert Schroeder, Peter Schupp, Ellen Smith and Brian Zgliczynski

Published by the Bureau of Statistics and Plans - Guam Coastal Management Program
Funded through CRI Grant NA06NOS4260114


Coral reef condition:
A status report for
GUAM


ol. 564: 47–55, 2017
i: 10.3354/meps12005

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published February

Anomalous temperatures and extreme tides: Guam staghorn *Acropora* succumb to a double threat

L. J. Raymundo[1,*], D. Burdick[2], V. A. Lapacek[1], R. Miller[2], V. Brown[3]

[1]University of Guam Marine Laboratory, UOG Station, Mangilao, Guam 96923, USA
[2]NOAA Guam Coral Reef Monitoring Program, University of Guam Marine Laboratory, UOG Station, Mangilao, Guam 96923, USA
NOAA Fisheries Habitat Conservation Division, Pacific Island Region, Guam Field Office, Tiyan, Guam 96913, US

ABSTRACT: Anomalously warm sea surface temperature events are increasing in frequency, generating global concern regarding the adaptive and acclimatizing capacities of corals. Staghorn *Acropora* corals, important ecologically as habitat structurers, are particularly vulnerable to temperature-related bleaching. Here, we report a catastrophic mass mortality event that affected shallow staghorn communities in Guam, Micronesia. Mortality began in conjunction with a mass bleaching event in late 2013, initiated by anomalous warm sea surface temperatures and doldrum winds over a 4 mo period. A second warming event followed less than 8 mo later, concurrent with a period of extreme low tides resulting in repeated periods of subaerial exposure of shallow corals. This combination of stressors acted synergistically to trigger an extended mass mortality event. In 2015, we conducted rapid assessment surveys of 7 species in 21 previously mapped populations to determine mortality extent and pattern. Mortality from these combined environmental stressors resulted in a 53 ± 10% reduction in Guam's staghorn population, covering an estimated 17.5 ha of coral communities. Greater water circulation appeared to be associated with higher survival during both warm temperature periods and extreme low tides; populations in slightly deeper water, closer to well-flushed reef margins, showed lower mortality. A better understanding of the environmental drivers of the mortality patterns we observed is currently being applied to developing strategies to restore and manage remaining populations.

KEY WORDS: Staghorn · *Acropora* · Bleaching · Guam · Mortality · ENSO

Resale or republication not permitted without written consent of the publisher

INTRODUCTION

(Golbuu et al. 2007, Maynard et al. 2008, Fabric

PROJECT OVERVIEW

Project background

The *Comprehensive Long-term Monitoring at Permanent Sites in Guam* project, also known to as the Guam Long-term Coral Reef Monitoring Program (GLTMP), involves the regular collection of data for a suite of coral reef ecosystem health parameters at High Priority Reef Areas (HPRAs) around Guam, as well as support for coral bleaching response and other monitoring and assessment activities carried out by the Guam Coral Reef Response Team. The project is funded as part of a NOAA Coral Reef Conservation Program jurisdictional cooperative agreement for Guam. Data are currently collected and managed by a small team of highly trained biologists from the University of Guam Marine Laboratory (UOGML), but in previous years data collection efforts were supported by the NOAA Pacific Islands Regional Office (PIRO) Guam Field Office, and with occasional assistance by staff from the Bureau of Statistics and Plans (BSP), the Guam Environmental Protection Agency (GEPA) and the Department of Agriculture's Division of Aquatic and Wildlife Resources (DAWR). The core GLTMP team is currently comprised of three UOGML employees, including a part-time Monitoring Program Coordinator (MPC) and two full-time Monitoring Technicians (MT). In her previous role as a coral reef ecologist with NOAA PIRO's Guam Field Office, Valerie Brown also served as a core member of the monitoring team, providing significant input into project planning as well as leading the collection and analysis of reef fish survey data. All GLTMP staff are highly skilled scientific divers trained to carry out a variety of coral reef survey methods. Since 2013 the GLTMP's activities have been supported by the staff, boats, AAUS-compliant scientific diving program, and other infrastructure available at the UOGML.

Guam's reefs have been the subject of numerous studies since the mid-twentieth century, but not until the establishment of the GLTMP had there been a coral reef monitoring program that comprehensively addressed changes in the condition of permanently established reef monitoring sites through regular data collection efforts. This lack of baseline and time series information limited the ability of managers to evaluate natural and anthropogenic impacts to specific reef areas around Guam and to gauge the effectiveness of management activities at the scale of individual management areas. In order to address this major gap the Government of Guam's natural resource agencies, with the assistance of the UOGML and NOAA PIRO, developed a long-term monitoring strategy aimed at addressing the management needs of local resource agencies and the objectives set by the National Coral Reef Ecosystem Monitoring Program (NCREMP). A program coordinator was hired in January 2007 to further develop and implement the monitoring strategy and to coordinate the development of the 2008 State of the Coral Reef Ecosystems of Guam report. A large amount of baseline data for a number of key ecosystem health parameters is now available for several High Priority Reef Areas (HPRAs) that are currently the focus of a number of management actions aimed at improving reef condition. Subsequent data collection, which is on-going at six of these reef areas, provides critical information to managers about the effectiveness of management efforts, and alerts managers to emerging threats that may not otherwise be detected. The data collected at HPRAs are also essential for evaluating the relative resilience of the reef communities to climate change impacts and for tracking changes in resilience over time. GLTMP data collection efforts within three of the five locally managed Marine Preserves provide the only long-term monitoring data from the seaward slope available for evaluating the effectiveness of individual preserves.

The initial focus of the GLTMP was on the HPRAs, but a historically severe mass coral bleaching event in 2013, and subsequent bleaching events in 2014, 2016, and 2017, necessitated a significant shift of the already-limited resources to island-wide data collection efforts that targeted the shallow reef communities that were most severely impacted by the event. The data collected by the GLTMP and other members of the Guam Coral Reef Response Team at sites around the island during and between coral bleaching events are critical for understanding how Guam's reefs are changing in response to warming seas. Ultimately, these data will improve our understanding of what the future of Guam's reefs will look like and what impacts those changes will have on the people of Guam.

It soon became clear that the limited capacity of the small GLTMP team was not sufficient to carry out annual monitoring at all of the HPRAs while also participating in island-wide, survey-intensive data collection effort aimed at documenting the extent and severity of the bleaching events. Recently implemented changes are helping the program adapt to the shifting management priorities in this period of catastrophic climate change-associated coral reef impacts, and further efficiencies are being explored.

***Previous page**: A school of brassy trevally (Caranx papuensis) at a sampling station within the Tumon Bay High Priority Reef Area in 2020.*

In addition to carrying out the monitoring of the HPRAs and the island-wide bleaching response and recovery sites, members of the GLTMP have also made essential contributions to the mapping, assessment, and monitoring of Guam's staghorn coral communities; an island-wide reef resilience assessment in 2016 (see Maynard et al. 2017); and a variety of other monitoring, assessment, outreach, and management-related activities. The MPC, as well as other members of the monitoring team, have also provided key input into the development of management strategies, such as the Guam Coral Bleaching Response Plan and the Guam Reef Resilience Strategy, and are currently participating in the development of Guam's Fisheries Management Plan as well as the Guam Coral Reef Restoration Strategy.

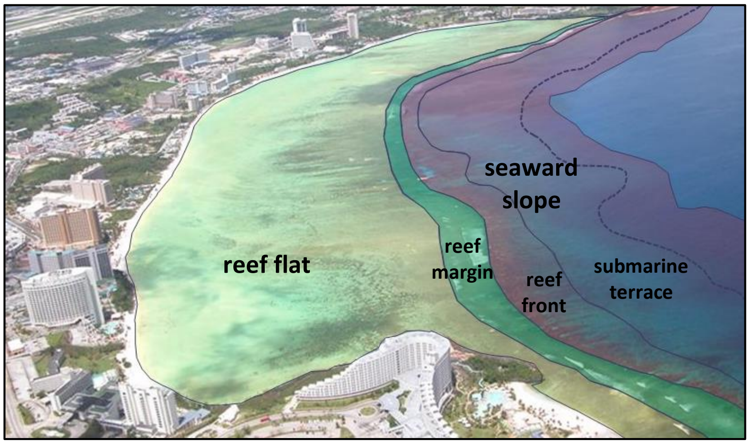
The *Comprehensive Long-term Monitoring at Permanent Sites in Guam* project also includes support for the Reef Flat Monitoring Program carried out by Dr. Laurie Raymundo of the UOGML, which has tracked changes in the condition of Guam's reef flat coral communities since 2009. The results of a broad-level analysis of coral cover data collected by the Reef Flat Monitoring Program between 2009 and 2022 are presented beginning on p.40 of this report; the results of a previous analysis of both coral cover and coral health data were presented in the December 2018 Interim report (see Burdick and Raymundo 2018), which can be found at the links on p. 3.

Regular monitoring at High Priority Reef Areas

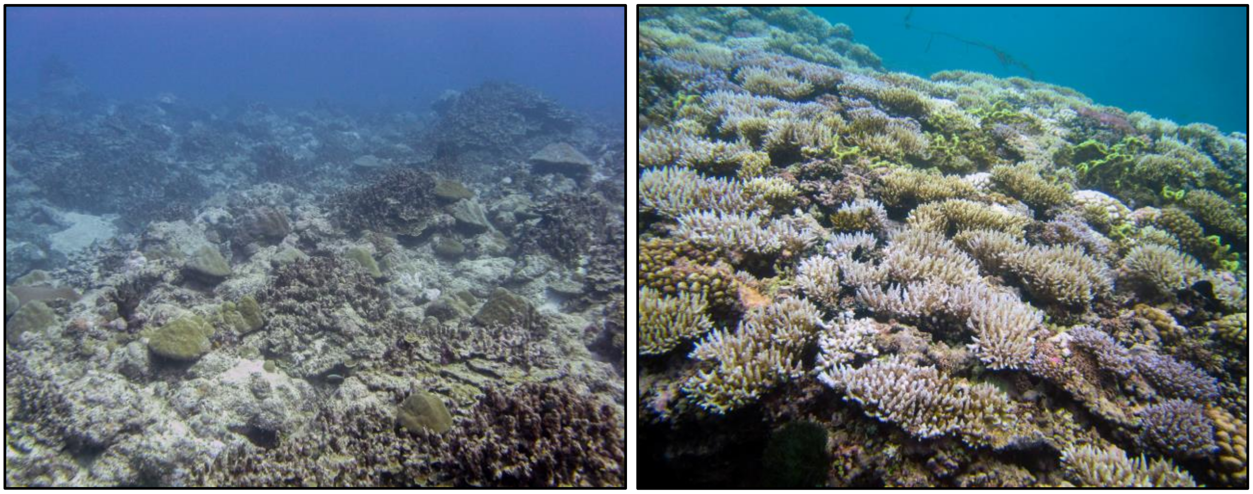
The High Priority Reef Areas, or HPRAs, targeted for monitoring by the GLTMP include the Tumon Bay Marine Preserve, East Agana Bay, the Piti Bomb Holes Marine Preserve, the Achang Reef Flat Marine Preserve, the eastern portion of the Cocos barrier reef (Cocos-East), and Fouha Bay. Data collection at the Tumon and East Agana HPRAs began in 2010, and since then data has been collected at a total of seven HPRAs, with data collected from six sites on a regular basis. Baseline surveys were carried out at Western Shoals, in Apra Harbor, in 2011 but further data collection at this site has been postponed indefinitely. A detailed account of GLTMP data collection activities at the HPRAs between 2009 and 2022 is provided in Appendix A.

The GLTMP's comprehensive approach to ecological monitoring at the HPRAs, combined with the high density of samples within the relatively large (0.1–0.2 km^2) reef areas, provide data critical to understanding changes in reef condition at these areas, for the development of management strategies specific to each area, and for evaluating the effectiveness of management actions in improving the condition of these high value marine ecosystems. The sampling design and survey protocols used at the HPRAs are described beginning on p. 16 of this report. The relatively large size of the HPRAs and the large number of sampling stations regularly surveyed within their boundaries allow the program to obtain data across a sizeable portion of high priority reef systems and avoid the risk of missing significant impacts along a reef tract or overgeneralizing geographically isolated impacts (e.g., a limited COTS outbreak) that can occur when using a small number of closely spaced transects.

The archipelago-wide scope of the NOAA Pacific Island Fisheries Science Center's (PIFSC) triennial Reef Assessment and Monitoring Program (RAMP) cruises, which GLTMP staff have supported through multiple deployments, provide important snapshots of benthic and reef fish community condition across a much broader geographic scale. The data collected during the RAMP cruises is not intended for use in assessing the status and trends in coral reef condition at individual reef sites, such as within the HPRAs monitored by the GLTMP, or for documenting the extent and severity of coral bleaching as a result of thermal stress events, a task taken up by the GLTMP and other members of the Guam Response Team. The GLTMP and PIFSC staff continue to work closely with each other, sharing data and methods, on studies that take advantage of the strengths of the data generated by our respective programs and



View of Tumon Bay from Two Lovers' Pt., showing location of different reef zones and subzones typical of fringing reefs



Reconnaissance surveys conducted at the beginning of the 2013 coral bleaching event revealed minimal or no bleaching at mid-depth, Porites-dominated reefs, such as that found within the Tumon Bay HPRA (left) and other previously established HPRAs. In contrast, numerous pale and bleached corals were observed at shallow, wave-exposed reef areas, such as the reef front at Sharks Hole, in northwestern Guam (right). Monitoring had not previously been conducted at these shallower, bleaching-susceptible reef areas, thus necessitating a major shift in GLTMP priorities and the commencement of island-wide monitoring at these shallow reef areas.

yield important information for Guam and the Marianas more broadly that cannot be attained by each program alone.

The results of analyses of baseline data collected at each the HPRAS, as well as time series analyses of data obtained at some of the HPRAs, were presented in previous reports (available at the links provided on p. 3). The results of time series analyses of benthic cover, reef fish, and macroinvertebrate data obtained at all six regularly visited HPRAs between 2010 and 2021 are presented beginning on p. 47 of this report.

Coral bleaching response

Researchers at the UOGML and scientists from government agencies have documented coral bleaching on Guam's reefs since the phenomenon was first reported in 1994 (Paulay and Benayahu 1999, Burdick et al. 2008). GLTMP staff and other members of the Guam Coral Reef Response Team contributed to efforts to track the relatively modest impacts of past coral bleaching on Guam's reefs, such as the bleaching observed in 2006 and 2007. In 2013, the coral reefs of Guam and other islands in the Mariana Archipelago experienced a significant bleaching event associated with anomalously high sea surface temperatures and an extended period of calm water conditions. At the time, the 2013 mass coral bleaching event registered as the most severe to have affected Guam's reefs, but within just a few years that major event would itself be overshadowed by an even more devastating bleaching event.

A large-scale data collection effort was initiated by UOGML researchers and other members of the Guam Coral Reef Response Team in order to document the extent and severity of the historic 2013 event, and to collect datasets that would prove critical to understanding the impact of mass coral bleaching on Guam's reefs and inform predictions about the future of Guam's reefs. Members of the GLTMP made significant contributions to the organization and implementation of the UOGML-led island-wide assessment. Follow-up surveys were carried out with the assistance of GLTMP staff in 2015 at a subset of the original 48 bleaching response sites in order to assess the full impact of the 2013 event, as well as assess additional mortality that may have resulted from a subsequent bleaching event in 2014. Beginning in July 2016 one of the MTs assisted the NOAA Coral Fellow with numerous qualitative and quantitative bleaching response surveys at various reef flat sites around the island, and several GLTMP staff documented bleaching observed along the seaward slope in conjunction with surveys for an island-wide reef resilience assessment[1]. In response to the record-breaking 2017 bleaching event GLTMP team members

---

[1] The MPC served as Co-PI with Dr. Jeffrey Maynard on a NOAA Saltonstall-Kennedy grant-funded project aimed at assessing the resilience of reef areas around island. The results of this effort were presented in a report available at guamcoralreefs.info and coris.noaa.gov.

participated in surveys at 12 previously surveyed sites located along the northern, western, and southern coasts. The team also established three shallow (2 m), shore-accessible, 30 m-long permanent transects at Tanguisson Pt., on the western leeward side of the island, and in Pago Bay, on the eastern windward side of the island, at the beginning of the 2017 bleaching event. In addition, the team re-established four 50-m transects originally established in 2007 along the 10 m depth contour in Pago Bay by Dr. Peter Schuup of the UOGML. Baseline and multiple follow-up benthic photo transect surveys were carried out in an effort to document changes in the benthic communities at these three sites—all of which hosted a high proportion of bleaching-susceptible species—with greater temporal resolution than could be achieved with an island-wide effort.

The data collected with the significant support of GLTMP team members during and between the multiple coral bleaching events that have impacted Guam's reefs since 2013 provide the only record of the extent and severity of these historic events. These data are being used to assess the susceptibility of different coral species to bleaching, determine the environmental and human-associated drivers of differences in bleaching impacts observed across different reef areas, and make predictions about the future composition and function of Guam's reefs under a range of climate change scenarios. GLTMP staff and collaborators have published several journal articles based on initial analyses of the data and continue to further analyze the data to better understand the impact of ocean warming on Guam's reef ecosystem. Bleaching survey data collected in 2013 provided the foundation of a UOGML graduate student thesis project completed in 2016, and data collected in 2013 and 2015 were also provided to Dr. Jeff Maynard and colleagues for a NOAA-funded study of the human and natural drivers of coral reef resilience to climate-induced coral bleaching in Guam and the identification of potential climate refugia. The results of an initial analysis of coral bleaching survey data collected between 2013 and 2017 was published in the journal Coral Reefs in 2019; those data were also submitted to NOAA Coral Reef Watch staff for inclusion in an analysis of the 2014–2017 global coral bleaching event, the results of which were published in the same special issue of the journal Coral Reefs in 2019. Those data were also used for a more in-depth analysis of the 2014–2017 global coral bleaching event carried out by an international team of more than 120 researchers, led by Dr. Mark Eakin, formerly with NOAA Coral Reef Watch.

Summaries of the results of analyses of coral bleaching response and recovery data collected during the island-wide shallow seaward slope surveys, staghorn coral mortality assessments, and reef flat monitoring surveys were provided in the Interim report submitted to NOAA CRCP in December 2018 (see Burdick and Raymundo 2018), and are again summarized in this report (beginning on p. 26), with the results updated to include more recently obtained data.

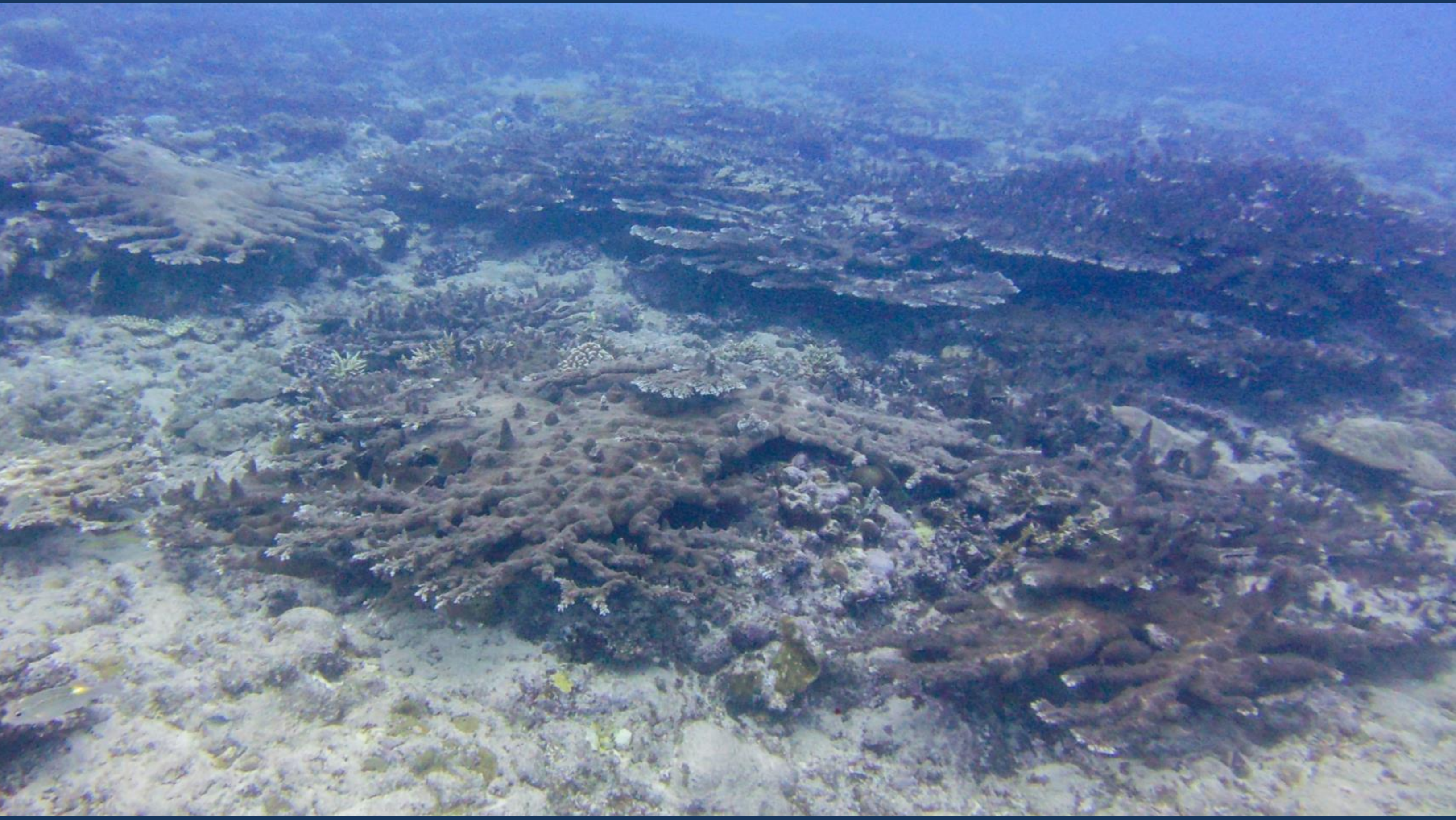
Other rapid response activities

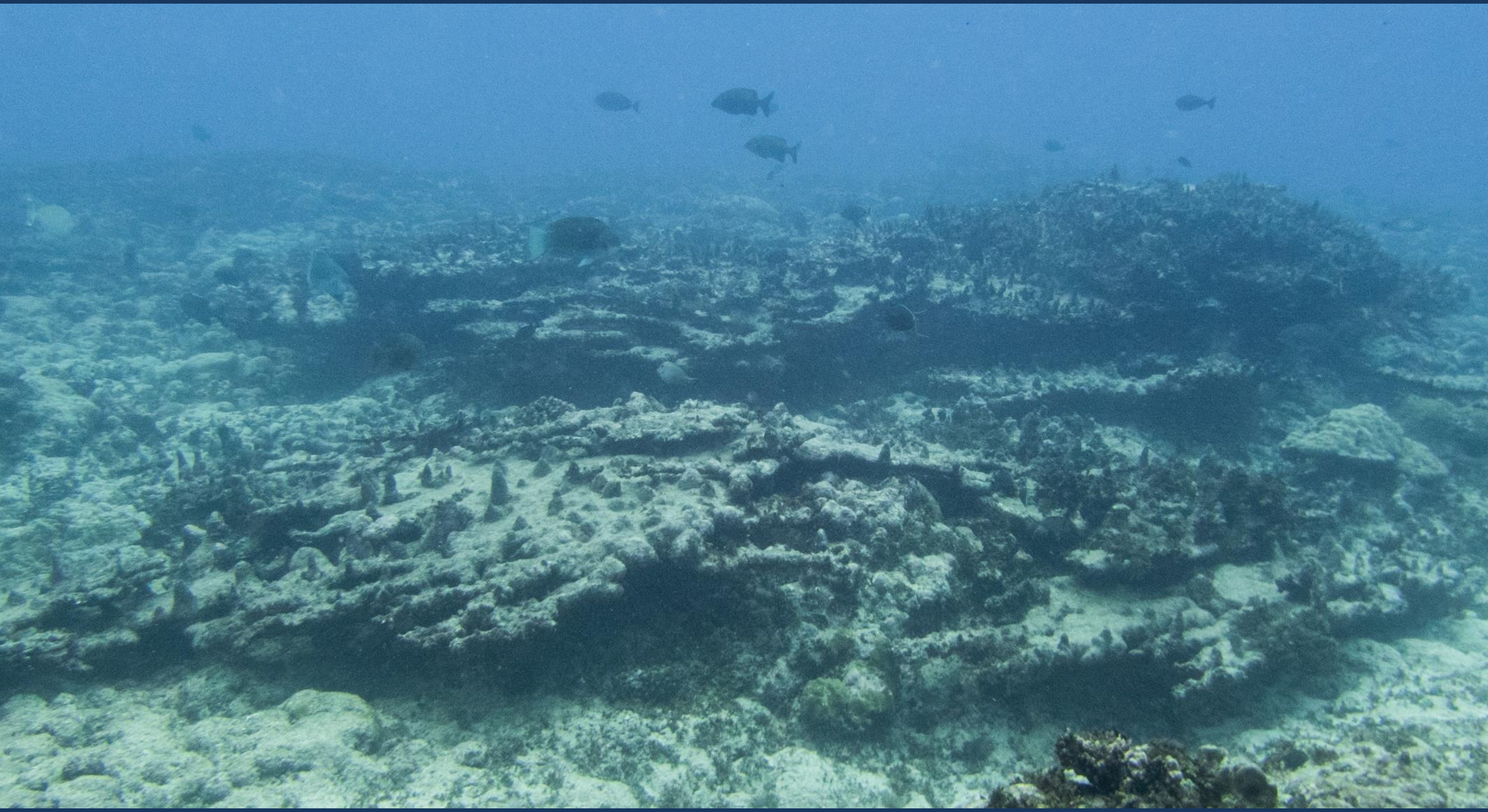
While coral bleaching response has comprised the bulk of the GLTMP's rapid response efforts, monitoring team members have also contributed to other Guam Coral Reef Response Team activities. Examples of GLTMP staff involvement in other Guam Coral Reef Response Team activities include support for crown-of-thorns sea star (COTS) surveillance and control efforts, contributing to the development of response strategies, and providing technical assistance to resource agencies in the assessment of coral reef damage caused by multiple vessel groundings.

Mapping and assessment of staghorn coral communities

Between 2009 and 2013 GLTMP staff opportunistically recorded the presence and species composition of staghorn communities around the island in an effort to generate a comprehensive, geographically referenced inventory of these communities. The in-water observations and GPS data were used to inform the development of a spatial data layer that included all known staghorn coral communities on Guam. The staghorn corals spatial data layer and related information provided the foundation for a study led by Dr. Laurie Raymundo, and supported by GLTMP staff,

***Next page**: A stand of Acropora abrotanoides colonies just north of Anao Pt., in northeastern Guam, during the 2013 bleaching event (top), and the algae-covered skeletons of the same colonies in 2019, following the 2017 bleaching event. This important structure-providing coral appeared to have withstood the heat and light stress of the bleaching events in 2013, 2014, and 2016, but many large colonies, especially along the northeastern coast and in shallow (<4 m depth) reef front areas around the island, succumbed to the historically severe heat stress of the 2017 event.*





to assess the amount of staghorn coral loss associated with the back-to-back bleaching events in 2013 and 2014. The results of the study were published in Marine Ecology Progress Series in 2017 (see Raymundo et al. 2017). The results of a follow-up staghorn mortality assessment conducted by Dr. Raymundo's lab, also with support from GLTMP staff, in 2017, were summarized in Burdick and Raymundo (2018), and the results of data obtained by the Raymundo Lab in 2019 were presented in a 2022 UOGML technical report (see Raymundo et al. 2022, links provided on p. 3). A summary of the results of these staghorn mortality assessments is provided beginning on p. 36 of this report.

Beyond long-term monitoring and rapid response

The work of the small GLTMP team extends beyond monitoring at the HPRAs, reef resilience assessments, coral bleaching assessments and other monitoring and assessment activities. The broad array of activities presented in the abridged list below is illustrative of the critical capacity GLTMP team members provide to the assessment and management of Guam's coral reefs. Additional activities are listed in the 2019 report (links on p. 3).

- Provided data and expert input via document review, workshops, and other communications regarding the initial listing and subsequent management of ESA-listed coral species
- Participated in the development of the Guam Coral Reef Status Report (https://www.coris.noaa.gov/monitoring/status_report)
- Assisted with multiple SECORE (SExual COral REproduction) workshops, coral colony out-plantings, coral nursery establishment, and methods testing experiments on Guam (http://www.secore.org/site/our-work/detail/project-guam.20.html)
- Carried out surveys as part of the 2009, 2011, 2017 and 2022 NOAA PIFSC Marianas Reef Assessment and Monitoring Program cruises
- Carried out all field work and contributed to the development of a final report for a 2015 NOAA PIRO Moving Window Analysis of benthic communities in Fouha Bay; a paper documenting the results was published in Marine Pollution Bulletin in 2022 (https://doi.org/10.1016/j.marpolbul.2022.113816)
- Provided expert review of major environmental impact assessments pertaining to the planned U.S. marine relocation and the construction of a nuclear aircraft carrier wharf and turning basin in Apra Harbor; also served as a member of the Natural Resources Subcommittee of the Civilian-Military Taskforce and participated in numerous meetings, outreach events, and site inspections related to the buildup
- Contributed to the development of measures of success for marine resources under the Micronesia Challenge
- Contributed to a collaborative NSF RAPID-funded study of bleaching susceptibility and resilience in 2014
- Assisted with Guam Community Coral Reef Monitoring Program training activities
- Carried out surveys in support of a RARE Pride project in Tepungan (Piti) Bay in 2012 and 2014
- Assisted the Guam Department of Agriculture's Forestry and Soil Resources Division with the development of a State-wide Assessment and Resource Strategy
- Assisted the Guam Department of Agriculture's Aquatics and Wildlife Division with the planning and implementation of a series of community workshops in which the Limits of Acceptable Change framework was used to solicit public input into the development of regulations for a Marine Preserve Eco-permitting law
- Assisted the Guam Bureau of Statistics and Plans and partners in the review of National Environmental Policy Act-related documents and other documents pertaining to the proposed military buildup
- Carried out coral biodiversity surveys at 38 sites in Apra Harbor as part of an independent assessment by NOAA and USFWS of coral reef resources within the impact area of a planned nuclear aircraft carrier turning basin and wharf
- Assisted Dr. Steve Kolinski with a NOAA PIRO-led assessment of an area proposed for a U.S. Navy Underwater Electromagnetic Measuring Range and three potential coral transplantation locations in Apra Harbor

Outreach

GLTMP staff have also carried out numerous outreach activities in an effort to raise awareness about the status and trends in the condition of Guam's reefs. These outreach activities include online and printed articles, a website, an

informational brochure, reef status reports, and numerous presentations provided to audiences of all ages. Examples of outreach materials generated by monitoring team members and by others who wished to highlight the work of the monitoring team can be found in the GLMTP's 2019 report available at the links on p. 3. In addition to the brochure, the online and printed articles, and the presentations mentioned above, the MPC and the 2012 NOAA Technical Support Specialist/2013 NOAA Coral Fellow (later one of the MTs) presented information about the monitoring program and closely related topics to audiences of all ages.
For example, presentations were provided to local and federal officials, talks for high school career days, regular (nearly every semester) talks to a University of Guam Environmental Biology class, presentations to a University of Guam Introduction to Geography class, presentations to a University of Guam Scientific Photography class, presentations for the NOAA PIRO and NPS-supported Friends of Reefs Guam Science Sunday lecture series, presentations for the University of Guam's POETS lecture series, a lecture on recent coral reef research for a workshop aimed at Guam educators, as well as several other presentations to school students of various ages.

Conference, workshop, and training participation

GLTMP staff have also presented at several local and international conferences and provided presentations, facilitation support, or were otherwise actively involved in numerous workshops and training opportunities since the program's inception. For example, presentations pertaining to the status of Guam's coral reef ecosystems and related topics were given at several Guam Coral Reef Symposia, the 2023 Conference on Island Sustainability, the 3rd Asia Pacific Coral Reef Symposium in Taiwan in 2014, and the 13th International Coral Reef Symposium in Honolulu, HI, in 2016.

The MPC has also provided training in, and developed materials for, the identification of Guam's coral species, and has supported workshops conducted by Dr. Raymundo and her colleagues aimed at improving the detection and identification of coral diseases among resource agency personnel and UOGML researchers, staff and students. The MPC has also provided presentations for two Guam Coastal Climate Resilience workshops (2016 and 2017), a Recreational Tour Operator workshop (2016), two SECORE [SExual COral REstoration] workshops (2014 and 2017), among others. The MPC and a previous MT also contributed to several Micronesia Challenge Measures Working Group meetings between 2008 and 2016.

Participation by the GLTMP staff in numerous workshops and training opportunities have contributed significantly to imparting to the staff the skills and knowledge critical to effectively and efficiently conduct coral reef assessment and monitoring, to incorporating new technologies and approaches, and to contributing to effective coral reef management on Guam. Examples of workshop and training opportunities provided to the MPC and/or other GLTMP staff include Reef Brigades reef restoration training (2021/22), Climate Smart Design Workshop (2018), three SECORE workshops (2013, 2014, 2017), PRIMER multivariate statistics software training (2013, 2016, 2018), and a Motorboat Operators Certification Course (2012).

A comprehensive list of conference participation through 2018 is provided in Appendix J of the April 2019 report (links on p. 3), and a list of workshops and training opportunities to which GLTMP contributed or in which they participated through 2018 is provided in Appendix K of that report.

Publications

Despite the significant workload associated with administering the program, collecting and managing the copious amount of coral reef survey data, carrying out broad level data analyses for progress reports, and the various other tasks described above, the MPC and other GLTMP staff have co-authored or otherwise supported multiple technical reports and peer-reviewed publications, most of which also relied upon data collected wholly by, or with the significant support of, the GLTMP. A comprehensive list of publications through 2022 is provided in Appendix B.

Collaborations

The GLTMP staff have engaged in collaborations with numerous partners over the years, with many of these collaborations aimed at gaining additional, more detailed insights into the copious data produced by the program, or by examining broader-scale trends by combining GLTMP data with data collected by other programs. Examples of recent and on-going collaborations include:

- An on-going collaboration with Dr. Romina King (UOGML) and Dr. Ved Chirayath (University of Miami) on the development and integration of high-resolution drone-based remote sensing, fluid lensing technology, and crowd-sourced benthic habitat classification into reef assessment and monitoring activities on Guam (see box on the following page)
- A NOAAO PIFSC-led investigation of the impacts of recent coral bleaching events on benthic and fish communities across the Marianas archipelago (on-going)
- A NOAA PIFSC-led investigation of the effectiveness of Guam’s marine preserves (on-going)
- A collaboration with Dr. Gareth Williams (Bangor University/SymbioSeas) and Dr. Laurie Raymundo (UOGML) to examine shifts in benthic communities in response to repeated coral bleaching events
- A follow-up collaboration with Dr. Williams and Dr. Raymundo to examine the impacts of coral bleaching events on coral communities
- A collaboration with Dr. Peter Houk (UOGML) to examine observer biases in GLTMP reef fish survey data and develop data quality control and analysis workflows in R (completed in 2020)
- A collaboration with Dr. Tom Schils of UOGML to investigate recent shifts in coralline algae communities (on-going)
- GLTMP staff deployments on NOAA Mariana RAMP cruises in 2009, 2011, 2017, and 2022
- A 2021 collaboration with Dr. Monica Moritsch (USGS) and PacIOOS to develop coral reef resilience map products for Guam (completed in 2021)
- A collaboration with NOAA PIRO and Dr. Dwayne Minton to examine the impacts of sediment on benthic communities in Fouha Bay and quantify the expected shifts in benthic communities in response to reduced sediment load (completed in 2015)

NOAA research vessel Hi’ialakai in front of Asuncion, in the northern CNMI, during the 2017 MARAMP cruise.

Reef Flat Monitoring Program

The Reef Flat Monitoring Program (RFMP) currently involves semi-annual data collection at five reef flat sites that extend along the western coast of Guam. The RFMP emerged following a study carried out by Dr. Laurie Raymundo and colleagues examining sewage nutrient eutrophication and its impact on coral disease along Guam’s northwestern coast (Redding et al. 2013). At the time of the original study seven sites were selected for the assessment and monitoring of disease impacts. Two of these sites, Tanguisson and West Hagåtña, were nearshore to recently upgraded sewage outfalls. The study examined whether there was any improvement in coral condition over the course of the year following the upgrade. The study concluded that all sites showed indications of sewage-based nitrogen pollution, little overall change in the concentration of nitrogen over the year of monitoring, and a positive correlation between the degree of nitrogen eutrophication and the severity of the coral disease, White Syndrome, in resident corals at each site. Upon completion of the study local management agencies requested that the monitoring of a subset of these sites be continued as part of the *Comprehensive Long-term Coral Reef Monitoring at Permanent Sites on Guam* project.

The RFMP has provided important, detailed information about trends in coral community health, with particular attention on coral diseases, predators, bleaching and other coral health concerns and the relationship with water temperature and nutrients. The data generated by the RFMP provides a strong complement to the data

EMERGING TECHNOLOGIES TO AUGMENT CORAL REEF MONITORING

USING FLUID LENSING AND MIDAR TO MAP NEAR-SHORE ECOSYSTEMS

By: Romina King, PhD
Associate Director
NASA Guam EPSCoR and NASA Guam Space Grant

In-depth look under the sea

In support of assisting natural resource managers with the monitoring of Guam's coral reef ecosystems, an ongoing campaign is using cutting-edge remote sensing technology to produce centimeter-scale resolution maps of Guam's coral reefs. Researchers Dr. Romina King of the University of Guam and Dr. Ved Chirayath of the University of Miami and are leading a multi-agency project that utilizes fluid lensing and Multispectral Imaging, Detection, and Active Reflectance (MiDAR) instruments to augment long-term coral reef monitoring — giving more high-resolution geographic coverage of the spaces between the existing monitoring sites in order to provide a wider, holistic context of the state of reef.

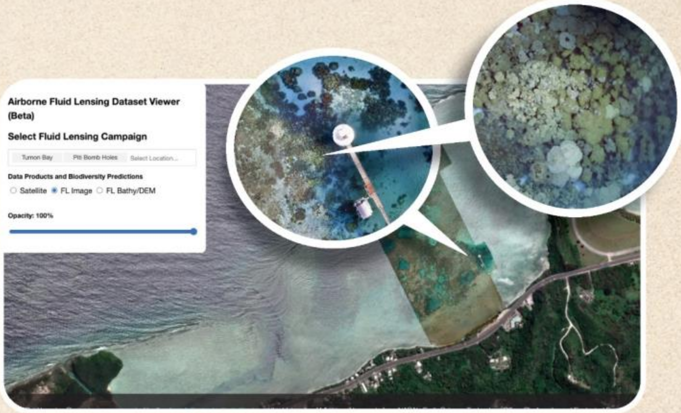
Advanced sensing technology

The fluid lensing algorithm post-processes aerial imagery collected by unmanned aerial vehicles (UAV) to reduce the distortion of water. The resulting clear images of shallow marine habitats down to 20 meters are stitched together to create georeferenced orthomosaics and can provide reef managers a more informed, perspective of the heterogeneity of Guam's reefs.

MiDAR, which won NASA's Invention of the Year in 2019, is an active multi-spectral sensor with a focus on the UV bands that potentially can be used to conduct rapid assessments of bleaching events. This experimental sensor may be able to augment existing coral reef bleaching assessments and response protocols by capturing information over a wide geographic area in a relatively short amount of time. This data could address questions about post-bleaching recovery times and the ability to quickly measure the severity and extent of an event, potentially allowing coral reef managers to monitor bleaching events in real time. MiDAR was flown on Guam in 2023 and demonstrated a high potential to detect marine debris, such as plastics.

Current available fluid lensing products include high-resolution benthic orthomosaic maps of priority areas, which can potentially assist field marine biologists with an update to Guam's benthic habitat maps as well as showcase reefs not regularly visited by the long-term coral reef monitoring team. Additionally, the University of Guam Drone Corps, which has been instrumental in advancing this project, will build on this new age of coral-reef monitoring by conducting regular aerial surveys of near-shore coral reef ecosystems. These efforts are intended to supplement the overall efforts of reef managers and provide tools to streamline monitoring efforts over a wide geographic area. Currently, a data viewer is being built to see these memory intensive images for those without high-power computing capabilities.

Special acknowledgements to Dr. Ved Chirayath, Keanno Fausto, Jonelle Sayama, Dong Won Lee, John I. Borja, and the UOG Drone Corps for contributing to this piece.



○ Satellite ● FL Image ○ FL Bathy/DEM

Opacity: 100%

One of the products of this project is a data viewer that coral reef managers can use to access high-resolution orthomosaic maps of surrounding reefs, such as the Piti Bomb Holes. (Image courtesy of University of Guam and University of Miami.)

3	8	504.2
summer campaigns	coral reef sites surveyed	hectares covered

- **2019** - Piti Bomb Holes, Tumon Bay
- **2022** - Piti Bomb Holes, Tumon Bay, Achang Reef, Cocos Lagoon
- **2023** - Piti Bomb Holes, Tumon Bay, Agat Cemetery, Luminao Reef, Togcha Bay, Ague Cove

AS OF JUNE 2023

During the summer campaigns, teams of drone pilots and free divers head out to the mapping sites to target and collect data. Mapping missions are also conducted by the UOG Drone Corps yearlong. (Left and bottom right photos by Keanno Fausto. Top right photo by Farron Taijeron.)

PARTNERS

collected at the seaward reef slope sites targeted by the GLTMP. The RFMP provides information to managers for a critical, yet vulnerable reef zone, and is an essential component of a comprehensive coral reef monitoring strategy.

The results of a preliminary analysis of benthic cover data collected between 2009 and 2014 as part of the RFMP were presented in a 2012 progress report for the *Comprehensive Long-term Coral Reef Monitoring at Permanent Sites on Guam* project. The results of further analysis of RFMP data collected between 2009 and 2018 were presented in the December 2018 interim report (links on p. 3), and the disease prevalence data obtained by the RFMP since 2009 provided the basis for an international collaborative study on the drivers of the common coral disease, White Syndrome, the results of which were presented in a 2020 paper in the scientific journal Frontiers in Marine Science (see Green et al. 2020). The results of an updated analysis of benthic cover data obtained at the reef flat monitoring sites between 2009 and 2022 are summarized beginning on p. 40 of this report.

Outcomes and benefits

Despite decades of research and numerous individual assessments of some of Guam's coral reef areas, it wasn't until the establishment of the GLTMP that regular data collection occurred at permanently established monitoring sites with the aim of comprehensively tracking changes in the condition of the reef communities at these reef areas. A large amount of baseline and time series data for a number of key ecosystem health parameters is now available for several High Priority Reef Areas, providing critical information to managers for the development of management strategies, for the evaluation of the effectiveness of management efforts, and for alerting managers to emerging threats that may not otherwise be detected. Notably, GLTMP data collection efforts at three of the five locally managed Marine Preserves provide the only long-term monitoring data from the seaward slope available for evaluating the effectiveness of individual preserves. The data collected at HPRAs are also essential for evaluating the relative resilience of the reef communities to climate change impacts and for tracking changes in resilience over time. The Reef Flat Monitoring Program carried out by Dr. Laurie Raymundo has tracked changes in the condition of several of Guam's reef flat coral communities since 2009, providing the only long-term monitoring of these important, but vulnerable, reef communities.

The data collected with the significant support of GLTMP team members during and between the multiple coral bleaching events provide the only record of the extent and severity of these historic, catastrophic events. These data are being used to assess the susceptibility of different coral species to bleaching, determine the environmental and human-associated drivers of differences in bleaching impacts observed across different reef areas, and make predictions about the future composition and function of Guam's reefs under a range of climate change scenarios. All of this information is important for understanding what the future of Guam's reefs will look like, how the people of Guam will be affected by these changes, and for developing proactive management strategies to help mitigate these impacts.

In addition to carrying out the monitoring of the HPRAs and the island-wide bleaching response and recovery sites, members of the GLTMP have also made essential contributions to the mapping and monitoring of Guam's staghorn coral communities, to an island-wide reef resilience assessment, and numerous other monitoring, assessment, research, outreach, and management-related activities. The GLTMP has contributed to local coral reef monitoring and management capacity through the continued employment of program staff and the formal and informal transfer of knowledge and skills to partnering organizations. The MPC, as well as other members of the monitoring team, have also provided key input into the development of management strategies, such as Guam Coral Bleaching Response Plan and Guam Reef Resilience Strategy, and are currently participating in the development of Guam's Fisheries Management Plan as well as the Guam Coral Reef Restoration Strategy.

The results of analyses of GLTMP data are presented in journal articles, periodic reports, outreach materials, and presentations to various audiences. Datasets collected through GLTMP efforts are provided upon request to resource agencies, research teams, and others, and once archived with NCEI will reach an even larger community of end-users. Anecdotal reports and the results of preliminary analyses of data obtained through GLTMP activities are also presented to resource agency staff at relevant multi-partner meetings, workshops, and conferences, and communicated via email, telephone conversations, and in-person meetings.

Adapting to a warming world

The major coral bleaching-associated changes in benthic communities observed at shallow sites around the island, and the lack of significant changes in reef condition at the HPRAs, necessitated a re-evaluation of GLTMP priorities. By significantly reducing the frequency of coral quadrat surveys within the HPRAs, using the CoralNet website to increase the efficiency of benthic image analysis, and counting and sizing only food fish species, among other program changes, the GLTMP team is now able to regularly survey 20 of the island-wide bleaching response and recovery sites while also continuing to regularly monitor the HPRAs.

Another emerging goal of the GLTMP in response to the expected continuation of rapid changes in the health of Guam's reefs as the average sea surface temperature increases is to actively integrate more threat-reduction and reef restoration activities into the program. GLTMP team members possess the technical skills required to carry out many of these activities and can provide technical guidance and supplement restoration and threat-reduction efforts when necessary. These activities include, but are not limited to, opportunistic COTS control at monitoring sites; COTS monitoring and control at other reef areas; continued participation in vessel grounding site response and rehabilitation; and assisting with coral nursery, coral outplanting, and other active coral restoration activities.

The GLTMP has long been an important component of Guam's response to coral bleaching events and COTS outbreaks and has made significant contributions to restoration efforts. In recent years, the MPC has assisted with the development of Guam's coral restoration strategy and, with support provided by The Nature Conservancy's Reef Resilience Program, attended the 2022 Reef Futures conference in Florida to learn about restoration efforts underway around the world. GLTMP staff have also provided technical support for recent COTS surveillance and control activities and have participated in coral restoration training, such as the Reef Brigades Micronesia training provided in 2021/2022 by The Nature Conservancy's Reef Resilience Network.

A GLTMP biologist, Andrea Hershberger, drilling a hole to accommodate a transplanted coral fragment (top) and another GLTMP biologist, Diona Drake, cementing a coral fragment to the substrate during the Reef Brigades training provided by The Nature Conservancy in 2021/22.

METHODS

Methods overview

The fundamental sampling design and survey protocols for the HPRAs were developed with the aim of contributing to the stated goals of the GLTMP. These goals include:

- Determining the status and trends in selected coral reef ecosystem indicators to better inform resource managers' decision-making process and increase the effectiveness of natural resource management
- Providing managers with early notice of abnormal conditions of selected resources to encourage effective mitigation measures and reduce the costs of management
- Providing data to better understand the dynamic nature and condition of the island's coastal ecosystems
- Allowing natural resource agencies to meet certain legal and Congressional mandates related to coastal resource protection
- Measuring progress towards performance goals

Regular, high intensity data collection was originally intended to occur only at a small number of relatively large priority reef areas, and was intended to complement the island-scale, ship-based, rapid ecological assessments carried out by NOAA PIFSC every two years (now three or more years) beginning in 2003. However, the onset of repeated bleaching events beginning in 2013 necessitated the implementation of sampling by GLTMP staff and other members of the Guam Coral Reef Response Team at an island-wide scale during and between these events. The sampling design used for data collection efforts at the island-wide bleaching response and recovery sites is necessarily different than that used at the HPRAs, but still contributes to achieving the broad GLTMP goals listed above. The protocols for the smaller number of survey methods employed at the island-wide sites are nearly identical to the equivalent survey methods used for the HPRAs, with only minor modifications. The Reef Flat Monitoring Program's methods are briefly described on pp. 40–41, and in more detail in previous reports, while the various methods used for the staghorn mortality assessments are briefly described on p. 38 and more comprehensively addressed in Raymundo et al. (2022).

Coral reef condition parameters

The GLTMP collects data about a number of important parameters related to ecosystem condition within the HPRAs. These parameters include benthic cover, rugosity, reef fish abundance and biomass, reef fish diversity, and the abundance of ecological or commercially important macroinvertebrate taxa. Coral colony size, density, and condition observations were obtained at the HPRAs between 2010 and 2020 and may be obtained again if capacity is available. Horizontal growth rates can be determined for some coral species using planar view images of permanently marked coral quadrats, but the GLTMP currently does not have the capacity to carry out this analysis and is seeking collaborators to assist. An analysis of coral quadrat images could also yield important insights into the dynamics between corals, algae, and other benthic organisms, especially as these relationships may be influenced by coral bleaching events, predation by COTS, and other acute and chronic stressors. Observations of protected or otherwise notable species, such as marine mammals, sea turtles, and tangison (*Cheilinus undulatus*) have also been recorded since data collection began at the HPRAs. Surveys at the island-wide bleaching response and recovery sites are currently focused on obtaining benthic cover estimates and data on the abundance of ecological or commercially important macroinvertebrate taxa, but reef fish survey data was obtained for a subset of sites in 2015 and 2016, and coral colony size and condition data was extracted from benthic photo transects images at sites surveyed in both 2013 and 2017.

Water temperature data obtained from in situ data loggers is available for several of the HPRAs between 2014 and 2020, but due to limited program capacity and budget new loggers have not been purchased and deployed. Additional water quality parameters have not yet been incorporated into field surveys due to program capacity limitations that prevent the regular deployment and maintenance of multiparameter datasondes and the organization and analysis of the collected data. In recognition of the importance of water quality monitoring, the

***Previous page**: GLTMP biologists, Andrea Hershberger and Diona Drake, carrying out benthic cover and macroinvertebrate belt transect surveys at an island-wide bleaching response and recovery monitoring site near Pati Pt., in northeastern Guam.*

MPC and the Guam CRI Program Coordinator are currently exploring funding opportunities to support a specialist that would contribute to the water quality monitoring needs of the GLTMP and local coral restoration projects. The MPC has also sought out collaborations with researchers interested in collecting water quality data as part of research projects carried out at the HPRAs.

HPRA site selection

The HPRAs currently targeted for long-term monitoring include the Tumon, East Agana Bay, Western Shoals, Piti, Achang, Cocos-East, and Fouha Bay sites (Fig. 1). Sites were selected by the multi-partner Guam Coral Reef Monitoring Group (GCRMG) after consideration of cultural and economic importance, the number of other management activities in the watershed, the amount and quality of available historical data, protected status, accessibility, and other factors. In order to achieve a relatively high level of detection and significant statistical power with a practical number of transects, sampling at most sites (with the exception of Western Shoals and Fouha Bay) is restricted to the submarine terrace, which is an area of relatively gentle slope that is found between the base of the high wave energy reef front and the steeper lower reef slope. Monitoring is not likely to be resumed at the Western Shoals site unless Department of Defense efforts to construct a nuclear aircraft carrier wharf and turning basin in Apra Harbor are revived, or another major threat to this important reef emerges.

Sampling design

High Priority Reef Areas

Data collection at the HPRAs is carried out using a stratified random sampling approach and the combined use of both permanent and non-permanent sampling stations across reef areas of significant size (0.1 to 0.2 km^2). This general approach is used for most of the original monitoring sites (with the exception of Fouha Bay), but the details of the sampling design may differ somewhat between sites. However, sampling strategies and survey methods are made consistent across sites to the fullest extent possible in order to maximize the ability to make between-site comparisons. This split-panel sampling approach was originally selected after consultation with the National Park Service, which uses this approach at sites comparable in size to the HPRAs. The approach is intended to provide robust data across relatively large reef areas that could be influenced by management activities carried out within their respective watersheds. These data would be more robust than data collected at a typical set of three to five closely spaced transects established in a small area that may not represent the broader benthic community influenced by the watershed. This approach was also selected because there had been a recent move towards site-based, ridge-to-reef reef management on Guam at the time the GLTMP was established, and because the island-scale data collected during the biennial (now triennial) NOAA PIFSC RAMP cruises were not intended to detect changes in reef condition at individual priority reef areas.

The locations of sampling stations within the boundaries of the HPRAs were generated randomly using Geographic Information System (GIS) software. Even-numbered stations are fixed, marked with rebar, while odd-numbered stations are unfixed; a new set of re-randomized, non-permanent stations are generated for each subsequent data collection effort. Site boundaries were initially defined using LIDAR bathymetry data and benthic habitat data within a GIS, but were subject to revision (as was the case with the Tumon Bay site) if analysis of baseline benthic data indicated that the site included two or more distinct benthic communities, resulting in high variances for key parameters. The delineation of boundaries for more recently established sites occurred after a series of exploratory/mapping dives were carried out across a large area of interest.

Data collection for most of the HPRAs is focused on hardbottom habitat of the outer reef slope terrace, between depths of 7 m and 15 m. Baseline monitoring at the Western Shoals site was focused within three strata, including the reef slope on the western half of the shoals, the reef flat margin on the western half of the shoals, and the reef flat margin on the eastern half of the shoals. Baseline data collection in the Fouha Bay site focused on areas towards the inner portion of the bay (three transects) as well as the outer portion of the bay (five on north side and five on south side).

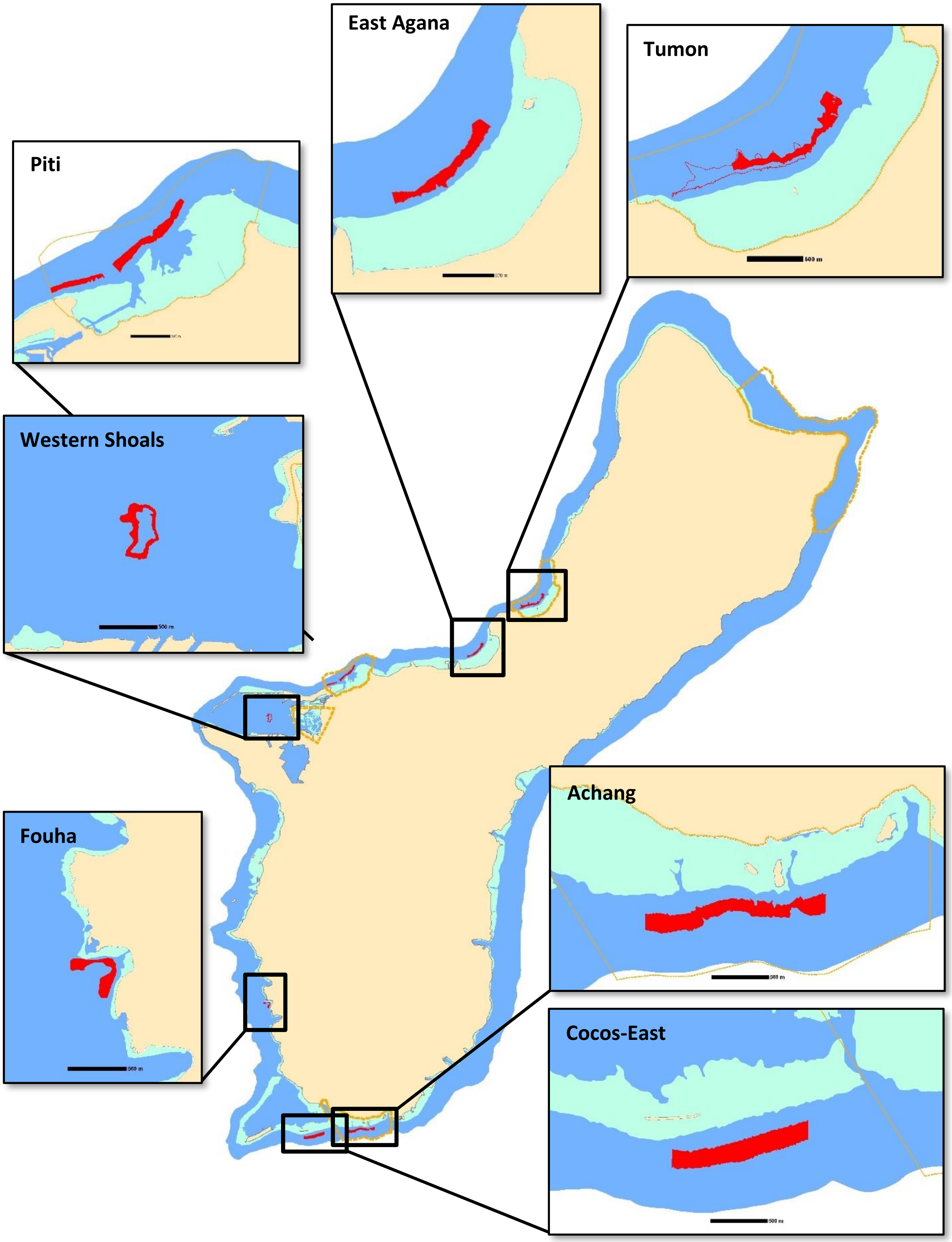


Figure 1. Location of the High Priority Reef Area long-term monitoring sites around Guam. Detailed site maps that display the location of all sampling stations surveyed between 2010 and 2021 are provided in the section beginning on p. 47 of this report.

Island-wide bleaching response and recovery

The split-panel approach is currently used only for high-intensity data collection at the HPRAs; monitoring at the island-wide sites instead occurs along three 30 m-long transects laid end-to-end along a depth contour at each site. The locations of the 20 sites were selected from a larger set of survey sites previously generated by NOAA PIFSC using a random stratified approach; the sites were selected in order to maximize coverage around the island, with priority placed on sites visited multiple times since 2013. Transects at the island-wide sites occur near the lower extent of the reef front (~5 m depth). A second set of three transects may also be placed immediately down-slope at a depth of 12 m if enough capacity is available to carry out this task. Data collection has occurred within the reef front zones of most of the island-wide sites beginning in 2013, while data collection along the 12 m depth contour was initiated at some of the sites as part of the 2016 reef resilience assessment and in 2017 as part of the bleaching response effort. A map depicting all island-wide bleaching response and recovery sites visited since 2013, including the locations of the sites marked with rebar in 2021, is provided in Fig. 3, on p.28.

Site monitoring

Site monitoring overview

Data collection efforts at the original HPRAs and the new set of 20 island-wide sites each target a different, but overlapping, set of parameters (as described above). Monitoring at the HPRAs has included, and will continue to include, the collection of reef fish, benthic cover, and macroinvertebrate data. Permanent coral quadrat markers will be opportunistically maintained, and coral quadrat surveys may resume if capacity is available to do so. Surveys at the island-wide sites have primarily been limited to benthic photo transect surveys and macroinvertebrate belt transect surveys; reef fish surveys may be conducted if capacity is available.

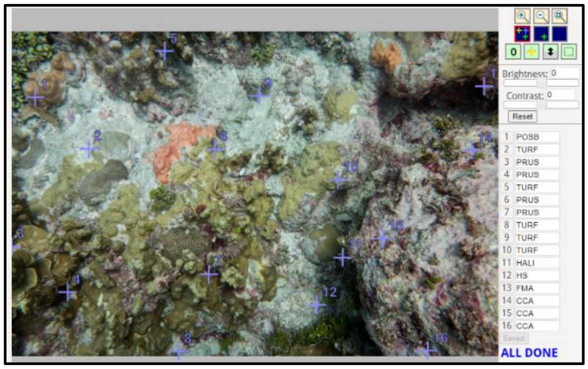
The original intent was to visit all of the HPRAs annually, but it has become clear that the large number of sampling stations required to attain a reasonably high level of detection and statistical power with limited capacity and resources make this target unrealistic, especially when considering the resources required to carry out surveys at the island-wide sites during and between coral bleaching events. Assuming the current level of program capacity is maintained, the Tumon and East Agana HPRAs will continue to be visited on an annual basis, while the Piti, Achang, and Cocos-East HPRAs will be visited on a biennial basis. Data collection efforts are currently planned for the Fouha Bay HPRA in 2023, but limited program capacity necessitates a re-evaluation of future data collection in the bay. The 20 island-wide sites will be visited on an annual basis when possible. Surveys at the island-wide sites will be carried out at the estimated peak of an active bleaching event; in years when bleaching is minimal or absent an effort will still be made to visit the sites between September and October, when bleaching is typically observed.

Survey logistics

Each sampling station is located using a GPS receiver. In optimal situations where at least four divers are available, two divers enter the water first to carry out the reef fish surveys. The transect is laid out in a clockwise direction (clockwise from a planar view of Guam), following the depth contour or at a previously determined heading parallel to the reef margin if the area is relatively flat and a depth contour is not readily discernable. For previously established permanent stations, divers locate the rebar marking the beginning of the transect and lay out the tape in line with the existing rebar. The two or more divers conducting the benthic surveys enter the water after the fish surveys are completed. For fixed sampling stations, rebar is installed at the beginning, middle, and end of the transect; four-inch concrete nails are installed at two or more of the corners of each coral quadrat location.

Benthic community

Benthic cover



Brightness: 0
Contrast: 0

1 POSB
2 TURF
3 PRUS
4 PRUS
5 TURF
6 PRUS
7 PRUS
8 TURF
9 TURF
10 TURF
11 HALI
12 HS
13 FMA
14 CCA
15 CCA
16 CCA

Benthic cover and coral relative abundance are currently assessed using digital photo transect surveys. Non-overlapping digital photos are taken every meter along each transect with a digital point and shoot camera mounted on a PVC frame. The percent cover for various benthic cover types was estimated from benthic photo transect images using the software application CPCe through 2022. Percent cover data is currently generated from the images using CoralNet, a free, open-source, web-based coral analytics solution hosted by the University of California San Diego and developed with support by NOAA and NSF (top right). Images are currently analyzed entirely by human observers, using 16 points per image, but an investigation by the NOAA PIFSC (see Williams et al. 2019) indicated that a fully automated implementation of the CoralNet machine-learning algorithm can now be relied upon to provide rapid, accurate results for total coral cover and the cover of common coral genera. Internal testing by GLTMP staff confirmed that estimates of total coral cover, as well as of dominant coral cover types (e.g., *Porites rus* and mounding *Porites* spp.) made by the CoralNet robot is highly comparable to those arrived at by human observers. Focused training of the CoralNet robot is expected to improve estimates of less common cover types. Further analysis of benthic photo transect images, such as analyses focused on rare or uncommon benthic classes, or the analysis of images obtained during bleaching events, will likely continue to remain primarily dependent on human observers, albeit alleviated to some degree by the CoralNet robot, until the technology is able to generate more accurate estimates for these benthic classes.

The MPC has also received training by NOAA PIFSC staff in the use of structure-from-motion (SfM) technology to generate 3D models and 2D orthomosaics of benthic communities (lower right). Due to staff capacity limitations, the GLTMP will likely continue to implement standard benthic photo transect surveys for the HPRAs and the island-

Examples of benthic organisms targeted by GLTMP surveys

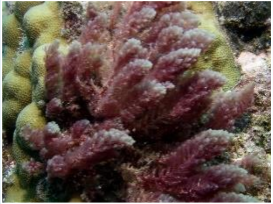
Hard corals (*Porites* spp.)

Hard corals (*Acropora* spp.)

Coralline algae (*Halimeda* spp.)

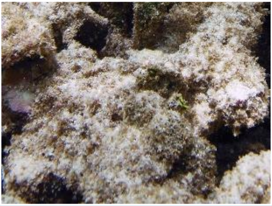
Crustose coralline algae

Fleshy macroalgae (*Asparagopsis*)



Encrusting sponges (*Terpios*)

Turf algae



Cyanobacteria

wide sites over the next few years, while opportunistically obtaining SfM imagery for a small subset of HPRA sampling stations and island-wide sites. As the technology and methodologies are further developed and efficiency is further improved, SfM surveys may eventually replace the benthic photo transect surveys for all sites.

Coral colony size and condition

As discussed above, the significant time and skill required to carry out the coral quadrat survey and the limited availability of the part-time program coordinator to conduct these surveys has necessitated a pause in the collection of coral quadrat survey data within the HPRAs. Permanent coral quadrat markers have been opportunistically maintained and planar view images of the quadrats obtained at some sampling stations in order to preserve the long-term viability of the quadrats, in the event that an increase in capacity or shift in priorities result in the re-addition of coral quadrat surveys to data collection efforts. A detailed description of the coral quadrat survey method used through 2020 is provided in the April 2019 report (links on p. 3). With the development of structure-from-motion technology and methodologies, the automated or semi-automated extraction of coral colony size, and possibly even coral condition, data from the SfM models may eventually be possible. If this can be achieved, the full in situ coral quadrat surveys may no longer be necessary, and limited in situ coral recruitment surveys could be carried out when capacity is available.

Rugosity

Beginning with the 2012 field season, rugosity has been measured at each sampling station using the standard chain-and-tape method. A diver carefully drapes a 15 m light chain over the substrate along a taut transect tape, paying out as much chain as is necessary to conform to the substrate profile along the length of the tape. The diver then records the length of the tape under which the chain was placed. A rugosity index value is calculated as the ratio between the length of chain and the linear distance between the chain endpoints. Using this method, a perfectly flat area would yield a rugosity index value of 1 and an area of more complex terrain would yield rugosity index value greater than 1.

Associated biological communities

Reef fishes

Between 2010 and 2019 the fish team used a Stationary Point Count Method (SPC) adapted from Ault et al. (2006)

Examples of food fish groups targeted by GLTMP surveys

Parrotfishes, *Låggua*
Groupers, *Gadao*
Rabbitfishes, *Sesyon*
Unicornfishes, *Hangon*
Rudderfishes, *Guili*
Jacks, *tarakitu*
Emperors, *M’atan hågon*
Snappers, *Fafa’et*

and the NOAA Pacific Islands Fisheries Science Center (Williams et al., 2011; Heenan et al., 2017) at the HPRAs. A modified version of this method (described below) was implemented beginning in 2020, although the original SPC method was used in 2022 in order to maximize comparability to reef fish SPC survey data collected by NOAA PIFSC biologist during the RAMP cruise that year. The original and modified SPC survey methods share the same fundamental structure (Fig. 2). A pair of fish divers deploy a 30 m transect across the substratum, position themselves at 7.5 m and 22.5 m, and survey fish within a 7.5 m radius cylinder extending from the substrate to the limits of vertical visibility. The SPC surveys are conducted in two parts. During the first five minutes, divers record all species observed within the cylinder, but do not count or size fish. All fish are identified to species level or the next lowest taxonomic level possible. After the first five minutes divers enumerate fish, one species grouping at a time, using rapid visual sweeps of the plot. The counts are designed to be "instantaneous" to avoid double counting. All fish of the target species within the SPC boundaries are counted and sized to the nearest centimeter. This process is continued until all of the target species are assessed.

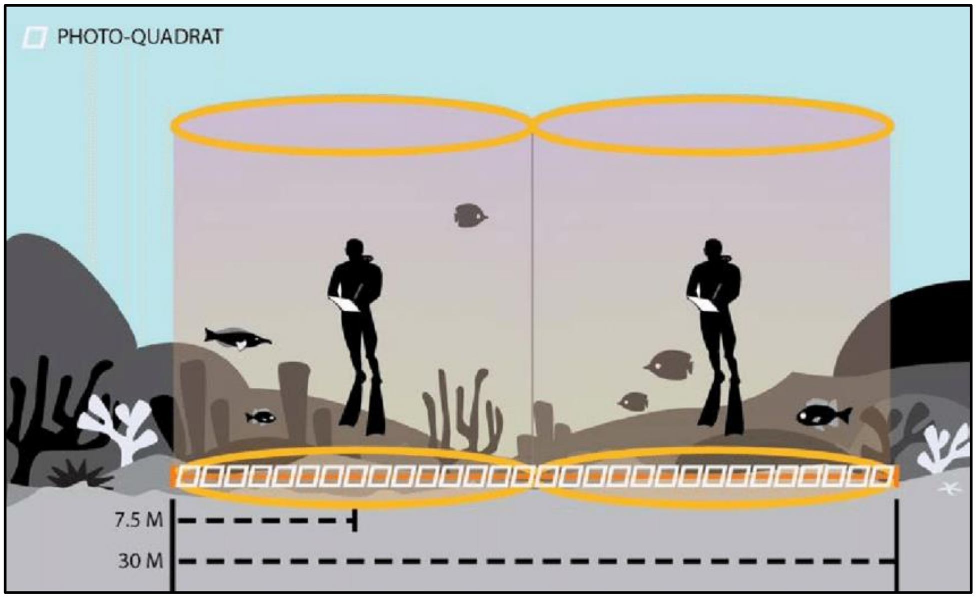


***Figure 2**.* Diagram illustrating diver position and cylinder size for the stationary point count reef fish survey method currently used by the GLTMP and NOAA PIFSC. Illustration provided by NOAA PIFSC.

The "full" SPC method targeted a broad array of food and non-food fish species, called for the non-instantaneous enumeration of fishes that enter the cylinder after the 5-minute listing phase, and for denoting the presence of additional species occurring outside the cylinder. In recognition of the insufficient capacity of the small GLTMP team to carry out the full method at the number of HPRAs targeted each year, and to allow for the collection of benthic cover and macroinvertebrate data at the 20 island-wide sites on a regular basis, the fish SPC method was modified to significantly shorten survey time. The method was modified in such a manner as to maximize to the extent possible the comparability to data collected using the full method for past surveys, and to data collected by NOAA PIFSC. The primary modification to the method involves the counting and sizing of only food fishes, and the elimination of observations of fish species that first enter the survey cylinder after the 5-minute listing phase. Most fish taxa surveyed as part of the full method are considered food fishes on Guam, but the small number of non-food fish groups include taxa (e.g., damselfishes and small wrasses) that can be abundant, small, and time-consuming to count and size. By excluding these taxa, each dive is shortened by up to 20 minutes, often allowing the team to visit an additional sampling station during a single boat trip. The 5-minute listing phase has not been modified—all species originally considered for listing with the full method are still listed; as such, these lists can still be directly compared to previous GLTMP and NOAA PIFSC survey data. However, even though most fish groups are still counted and sized, the exclusion of non-food fishes from the counting and sizing phase precludes direct comparison of total reef fish biomass as recorded using the full and modified methods. The biomass of specific groups, including those typically considered to be of greatest important to managers, can still be directly compared.

Macroinvertebrates

Counts of commercially and ecologically important macroinvertebrate species (COTS, echinoids, holothurians, *Tridacna*, etc.) are made within a 4 m wide belt (2 meters on either side of the transect). The size of each *Tridacna* clam is measured to the nearest cm. Boring urchins (e.g., *Echinostrephus aciculatus* and *Echinometra* spp.) and small sea stars (e.g., *Fromia* spp. and *Gomophia egyptiaca*) were eliminated from the survey beginning in 2020 due to the difficulty in accurately detecting these organisms with rapid assessments of relatively large belt transect areas.

Data management, archival, and distribution

Several datasets collected at the HPRAs currently reside in a relational database hosted by the University of Guam Office of Information Technology and made accessible to GLTMP staff through a web-based data entry portal. The database and accompanying data entry web portal were originally developed through a collaboration with the NOAA PIFSC and BSP and hosted by the Department of Administration. The data management system was migrated to UOG in 2020. Other datasets generated by the GLTMP may be integrated into the data management system in the future.

Metadata records developed for datasets obtained from the HPRAs have been posted to CoRIS in order to facilitate broader awareness of the available data sets. In addition, updated and reformatted metadata records have been submitted to InPort, a web-based National Marine Fisheries Service data management application. Full datasets and associated metadata for reef fish, coral colony, and macroinvertebrate data collected at the HPRAs between 2010 and 2019 have been archived with the National Centers for Environmental Information (NCEI); benthic cover data and more recently collected observations for the datasets that had already been submitted to NCEI will be submitted in 2023. Metadata for other datasets collected by the GLTMP, such as coral bleaching survey data, will be developed and submitted to CoRIS, InPort, and NCEI in the future. While all GLTMP imagery, data, and analysis products are available upon request, the submission of the metadata and full datasets to CoRIS, InPort, and NCEI broaden data discovery and facilitate the efficient distribution of these data to a broader community of end users.

Examples of macroinvertebrates targeted by GLTMP surveys

Crown-of-thorns sea star
Acanthaster* cf. *solaris

Common comet star
Linckia guildingi

Spiky sea cucumber, balåte
Stichopus chloronotus

Black teatfish, balåte
Holothuria whitmaei

Giant clam, *hima*
Tridacna maxima

Spider conch
Lambis chiragra

Top shell, *aliling*
Tectus niloticus

Diadema urchin, lå'on
Echinothrix diadema

Data analysis

Baseline data collected at each of the HPRAs was first explored in multivariate space using PRIMER and the PERMANOVA add-on, and power analyses were conducted to determine optimum sample sizes and to adjust HPRA site boundaries if two or more distinct benthic communities were found to occur within the original boundaries. More detailed descriptions of the multivariate and power analysis methods and the results of these analyses are available in previously published GLTMP reports (links on p. 3).

Percent cover values of all benthic cover classes were obtained for each HPRA sampling station or island-wide site for each sampling period by pooling the CPCe- or CoralNet-generated point count observations for all photo transect images from each station/site visit. Mean percent cover values of benthic cover types were calculated for individual HPRAs or island-wide site groups by averaging the percent cover values from all stations/sites surveyed within an HPRA/site group during a given sampling period. Mean cover values for the Tumon Bay HPRA in 2010 were calculated using only those stations occurring within the new site boundaries established beginning in 2012; these stations only occur within the southern half of the new site boundaries, however, and thus may not be representative of the entirety of the site. The relative abundance values for coral genera were generated using the benthic cover estimates. Coral community size structure and aspects of coral condition, such as the extent of partial mortality, sources of mortality, and coral disease prevalence, have been examined using baseline data obtained for some of the HPRAs; the results of these analyses were presented in previous reports.

Reef fish biomass values were calculated using length-weight regression factors and the visually estimated length of each observed fish. Length-weight regression values and length measurement conversion values were obtained from NOAA PIFSC to facilitate comparison with regional estimates. Biomass values for all fish were aggregated to species or family by station and then converted to an area-based measure. Observations of sharks, rays, barracuda, and eels were excluded from the biomass calculations, as their rarity and the large biomass values can obscure trends in the biomass of other fish groups. Spadefish (Ephipiidae) were also excluded from the analysis of data obtained within the Tumon Bay HPRA, as the biomass of a large school observed in 2021 obscured potential trends in more commonly observed (and more commonly harvested) reef fish species. Non-food fishes were not counted and sized beginning in 2020, so a total fish biomass value cannot be reported for HPRAs beyond 2019 [although total reef fish biomass can be reported for 2022 once those data are analyzed, as the full SPC survey method was used that year]. Reef fish species richness was calculated as the number of species observed at a given station. Reef fish density was calculated for each sampling station by combining the number of fish observed within the replicate SPC cylinders at each station, dividing the sum by the total area sampled, and converting to a more commonly used area-based measure (# of fish per 100 m^2). Observations of gobies and blennies were excluded from species richness calculations because of the difficulty in detecting the generally small members of these families and inconsistency in the observers' approach to surveying them. Mean reef fish biomass, density, and species richness were calculated for each HPRA. The density of macroinvertebrates and densities of four broad macroinvertebrate groups (e.g., edible shell, sea cucumber, sea star, and sea urchin) were calculated for each station as the number of individuals per 100 m^2, and mean density values were calculated for each HPRA.

Two-sample comparisons of mean values for selected benthic community, reef fish, and macroinvertebrate community parameters between time periods for individual HPRAs, or for coral cover for the island-wide sites, were carried out using the Partover.test function available in the Partiallyoverlapping package for R. Data were square root or square root (x+1) transformed, and tested for normality using a Shapiro-Wilk's test (Shapiro.test) prior to using the Partover.test function. Ranks were used when one or both paired samples were non-normal; ranks were calculated by pooling observations from both samples and using the mid-point for tied ranks.

Direct comparisons between HPRAs have not yet been conducted, but such comparisons may be made at a later date—perhaps in collaboration with researchers such as Dr. Peter Houk--once an adequate understanding of factors influencing the biological communities at each site is achieved and comparisons of one or more datasets is determined to be appropriate. In addition to analyses of GLTMP data carried out in-house by GLTMP personnel for journal articles, technical reports, and presentations, collaborations with outside experts are also pursued. These collaborations have included more in-depth or novel analyses of GLTMP data and the incorporation of GLTMP data into regional or global analyses.

***Next page**: The GLTMP coordinator carrying out a coral community bleaching assessment at a shallow (5 m) reef front site near Orote Pt. during the 2017 mass coral bleaching event.*

CHANGES AT ISLAND-WIDE CORAL BLEACHING RESPONSE AND RECOVERY SITES: 2013–2021

Key findings

- Guam's reefs were affected by four bleaching events between 2013 and 2017, resulting in a loss of about one-third of all coral in the shallow, wave-exposed reef front zone
- The eastern windward side of the island, which hosted more bleaching susceptible species, suffered a 60% decline during this period, while little change was detected along the western leeward coast
- Notable coral recovery was detected at more than half of the sites in the four years following the 2017 bleaching event, but coral cover at most sites had not yet returned to levels seen before the 2013 bleaching event

Introduction

Members of the Guam Coral Reef Response Team, including GLTMP staff, Dr. Laurie Raymundo (UOGML), UOGML graduate students, Valerie Brown (NOAA PIRO), Guam EPA staff, and Whitney Hoot (BSP), carried out multiple data collection efforts aimed at assessing the impact of thermal stress events on Guam's reefs communities between 2013 and 2017. Surveys targeted a randomized selection of seaward slope sites island-wide, priority reef flat sites, and all known major staghorn coral communities. GLTMP staff were primarily involved in data collection efforts targeting the seaward slope communities, but the MPC also provided significant contributions to the assessment of staghorn coral communities, and some staff support was provided for surveys at priority reef flat sites. In 2019 GLTMP staff carried out benthic photo transect surveys at nine (9) sites along the northern and eastern coast that could not be visited during the 2017 bleaching event due to hazardous surf conditions, and in 2021 twenty of the island-wide bleaching response and recovery sites were marked with rebar and benthic photo transect surveys were carried out. As mentioned earlier, these 20 sites represent the core sites at which surveys will be conducted during and between future bleaching events.

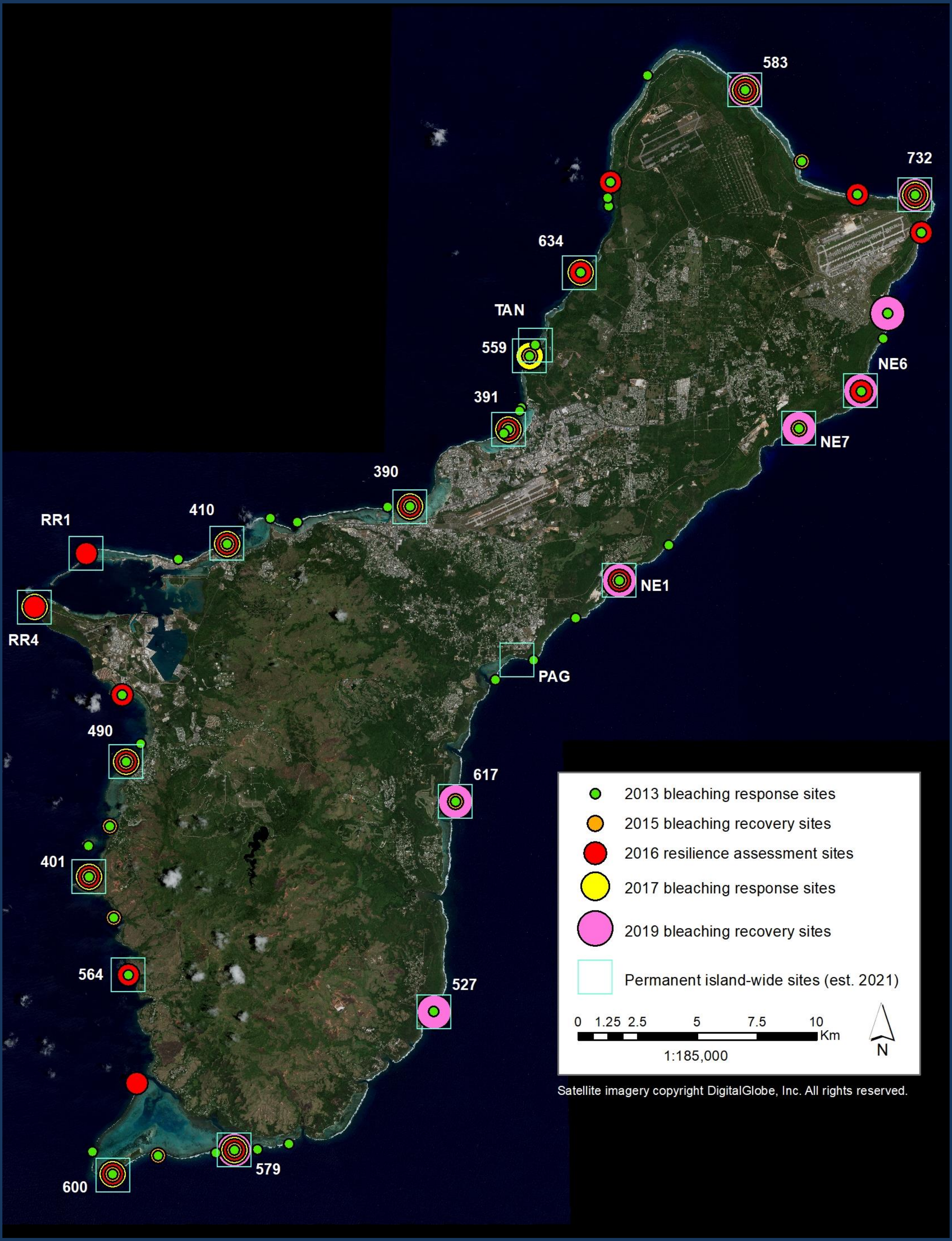
The results of an analysis of coral bleaching survey and staghorn mortality assessment data were included in a paper that was published in a special edition of the journal Coral Reefs focused on the 2014–2017 global coral bleaching event (see Raymundo et al. 2019). Because GLTMP staff played significant roles in the collection and analysis of data collected at the island-wide seaward slope sites, and the MPC made significant contributions to the development of the manuscript itself, a summary of the study's findings are presented below. These results were also presented in the Spring 2019 GLTMP report (see Burdick, Brown and Miller 2019, links on p. 3), but are updated here to include more recent benthic cover data and a brief summary of more recent environmental data. The impacts of the bleaching events on coral communities at staghorn thickets island-wide and the RFMP sites are presented on pages 36 and 40, respectively.

Methods

Environmental data analysis

Valerie Brown and the MPC obtained satellite-derived sea surface temperature (SST) and degree heating weeks (DHW) data, as well as bleaching alerts and predictions, from the NOAA Coral Reef Watch website (https://coralreefwatch.noaa.gov/satellite/index.php) in order to assess thermal conditions in the waters around Guam during periods of coral bleaching. Ms. Brown also obtained temperature and wave data from wave buoys near Ipan, eastern Guam, and Ritidian, in northern Guam, via the PacIOOS website (https://www.pacioos.hawaii.edu). Reef flat temperatures were recorded at three RFMP sites using loggers maintained by Dr. Raymundo since 2009.

Figure 3, *next page*. Location of coral bleaching response and reef resilience assessment sites surveyed between 2013 and 2017 with the significant support of GLTMP biologists, and the now permanently marked island-wide bleaching response and recovery sites surveyed in 2019 (windward sites only) and 2021.



Satellite imagery copyright DigitalGlobe, Inc. All rights reserved.

Benthic cover and bleaching prevalence surveys

Benthic photo transect surveys were carried out at 48 shallow (~5 m) seaward slope sites located around the island during the 2013 bleaching event and at 12 seaward slope (5 m and 12 m depths) sites located along the western and northern coasts during the 2017 bleaching event (Fig. 3). In 2015, benthic photo transect surveys were carried out at 15 of the 48 sites surveyed in 2013 in order to assess the cumulative impacts of the 2013 and 2014 bleaching events, and in 2016 benthic photo transect surveys were carried out at 20 seaward slope sites (5 and 12 m depths, including 17 of the sites surveyed in 2013) in conjunction with a NOAA S-K grant-funded reef resilience assessment for which the MPC served as Co-PI. In 2019, benthic photo transect surveys were carried out at a total of nine (9) sites located along the north and east coast, and in 2021 at 20 sites located around the island. The benthic photo transect surveys were carried out along three 25 (later 30) meter-long transects placed end-to- end across the target depth contour. Coral bleaching condition was also semi-quantitatively assessed in the vicinity of the benthic photo transects during the 2013 and 2017 bleaching events, and coral colony size and condition surveys were carried out in 2015. Macroinvertebrate belt transects were carried out at the 48 sites in 2013, and reef fish surveys were carried out by V. Brown at the 17 sites in 2015, but these data have not yet been analyzed and will not be discussed in this report. A detailed description of the coral bleaching condition survey methodology and the results of the assessments can be found in the Raymundo et al. (2019) paper.

Results and Discussion

Environmental conditions

Sea surface temperatures and accumulated heat stress (degree heating weeks) acquired from NOAA coral Reef Watch are provided for each year between 2013 and 2017 in Fig. 4. An analysis of satellite-derived SST and DHW data indicates that thermal stress conditions during the 2013 bleaching event were, at the time, the most severe since satellite measurements began. The predicted coral bleaching threshold for Guam (30°C) was exceeded for most of the period between June and October, with SST reaching a peak of 31.5°C in August. Accumulated heat stress reached a peak of 12 DHW in early October and did not fully dissipate until late December. In May 2014, less than six months after temperatures dissipated in 2013, satellite-derived SST exceeded the bleaching threshold. A maximum SST of 31°C was recorded in June, and although satellite-derived temperatures remained between 29°C and 30°C through December temperatures recorded by in situ loggers exceeded 31°C on several occasions between June and September. Accumulated heat stress reached a peak of 9 DHW in mid-September and dissipated by mid-

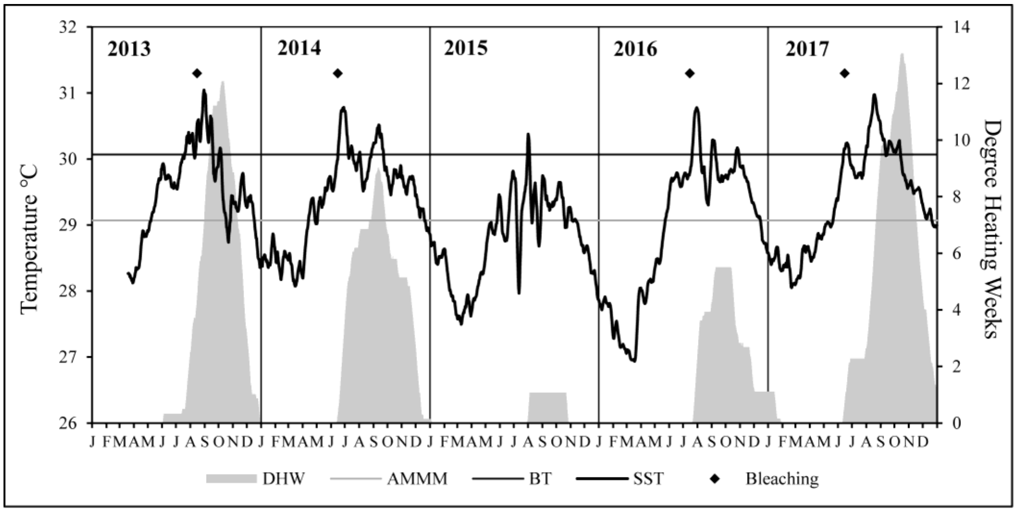


***Figure 4**.* Sea surface temperature from the NOAA CRW virtual station for Guam plotted against annual maximum monthly mean SST (AMMM), bleaching threshold (BT), and degree heating weeks (DHW) between 2013 and 2017. Figure reproduced from Raymundo et al. (2019).

December. Thermal stress in 2015 was low compared to the two previous years, reaching a peak of 30.8°C in late July and with accumulated heat stress never exceeding 1 DHW. While thermal stress in 2015 was limited, the onset of a strong El Nino event resulted in a 0.35 m decrease in sea level between late 2014 and 2015. The ENSO-associated extreme low tide events caused the repeated, prolonged subaerial exposure of shallow reef flat communities, which resulted in significant mortality to the upper extent of coral growth across much of Guam's reef flats. Thermal stress conditions in 2016 were moderately severe, with the 30°C bleaching threshold first exceeded in July and a maximum SST of 30.9°C recorded in late July. Accumulated heat stress peaked at 5.5 DHW between September and October and didn't fully dissipate until the end of January 2017. Water temperature at reef flat sites greatly exceeded the offshore satellite-derived SST measurements, with a maximum of 35.6°C recorded from a logger deployed in Tumon Bay. Thermal stress conditions in 2017 were more severe than the record-breaking conditions of 2013, with satellite-derived and buoy temperatures both exceeding 31°C in August and accumulated heat stress reaching 13 DHW in mid-October. Maximum water temperatures of between 34°C and 35°C were recorded from reef flat sites in Tumon Bay and Agat between June and August. While a comprehensive analysis of environmental conditions post-2017 was not conducted for this report, NOAA Coral Reef Watch Regional Virtual Station data for Guam showed no-to-limited heat stress in 2018 (max DHW = 0) and 2019 (max DHW = 3.3). Notable heat stress (max DHW = 9.2) was recorded between August and December 2020, but significant bleaching was not observed at the Piti, Tumon and East Agana HPRAs surveyed during that period. Bleaching may have impacted shallower coral communities in 2020, but unfortunately surveys at the shallower island-wide sites did not occur due to COVID-19 pandemic-related restrictions and staff capacity limitations. Heat stress in 2021 was minimal, reaching a max DHW of 1.6.

Bleaching prevalence at shallow seaward slope sites

Nearly a third (32 ± 19%) of coral cover at the shallow seaward slope sites was impacted by bleaching in 2013, with an average of 11 ± 9% of coral cover exhibiting bleaching-associated mortality. While the amount of bleaching-associated mortality was similar for the eastern windward sites (10 ± 7%) and the western leeward sites (12 ± 10%), the percentage of pale or bleached coral cover was significantly greater for eastern sites (31 ± 17%) than for western sites (11 ± 7%; Fig. 5). An analysis of the 2013 dataset by Travis Reynolds, a former graduate student of Dr. Raymundo, suggested that this difference was driven in large part by the greater proportion of bleaching-susceptible coral taxa in eastern windward coral communities (Reynolds 2016). While thermal conditions associated with coral bleaching were observed in 2014, seaward slope surveys were not carried out due to the lack of warning of the unusually early onset of bleaching conditions, which were not detected at the time by the older 50-km NOAA Coral Reef Watch product, and because of limited program capacity. However, significant bleaching, including bleaching-associated mortality, was observed at several reef flat sites. Consistent with the low levels of thermal stress recorded in 2015, shallow seaward slope communities exhibited low levels of bleaching (3 ± 3% of coral cover was pale or bleached), with no observed bleaching mortality. The mean percentage of pale or bleached coral at shallow seaward slope sites in 2016 was even lower than in 2015 (0.1 ± 0.5%), although it should be noted that significant bleaching was recorded at reef flat sites during this same period. The percentage of bleaching-impacted coral cover at seaward slope sites in 2017 was even greater than in 2013, with an island-wide mean of 48 ± 17% impacted coral cover, including 15 ± 17% of coral cover that exhibited bleaching-associated mortality. Semi-quantitative surveys of coral communities revealed that 92% of all surveyed coral taxa and 98% of coral genera exhibited paling, bleaching, or bleaching-associated mortality. A quantitative analysis of coral condition was not carried out using images obtained at the island-wide bleaching response and recovery sites surveyed in 2019 or 2021, as indications of heat stress were limited or absent those years.

Changes in benthic cover at shallow seaward slope sites

When data from all sites surveyed each year are considered, the percentage of living coral cover at shallow seaward slope sites island-wide declined by 34%, from 25 ± 13% to 17 ± 9% (Partover.test, t = 0.03) between 2013 and 2017 (Fig. 6); this decline is very likely attributed in large part to mortality associated with the multiple thermal stress events recorded during this period. The decline in mean island-wide coral cover was driven primarily by a nearly 60% decline (from 29 ± 13% to 12 ± 1%) at the eastern windward sites (Partover.test, t = <0.001), whereas no significant

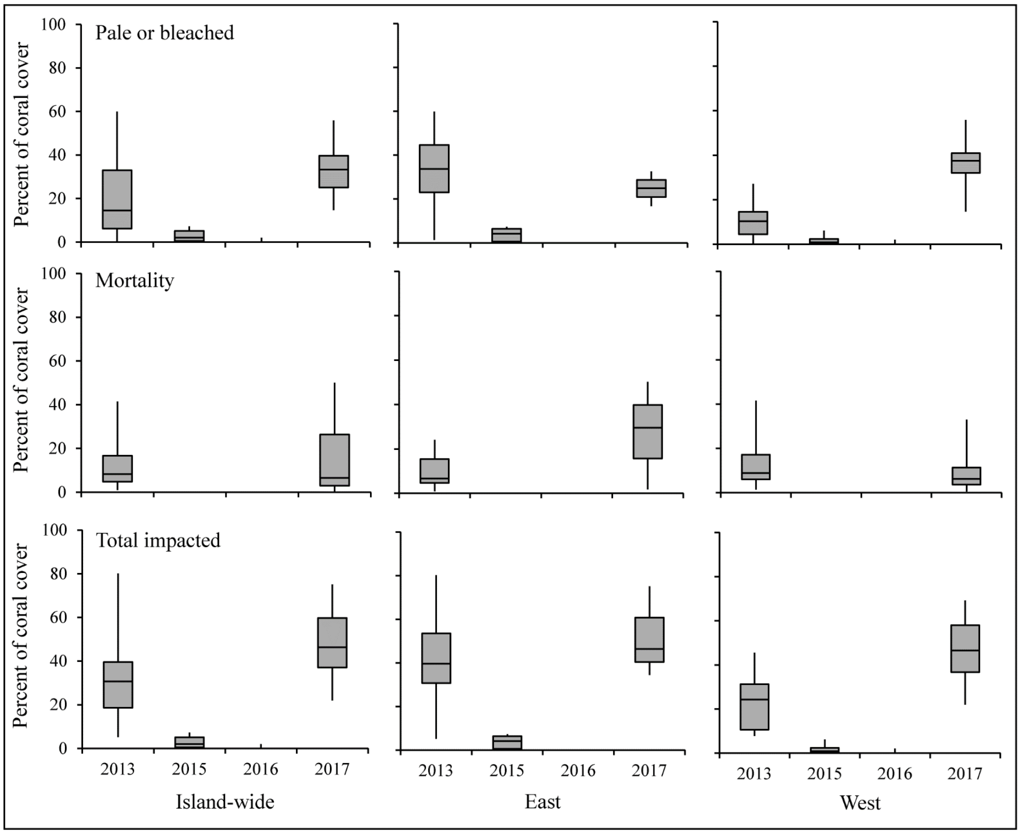


***Figure 5**.* Percentage of bleaching-impacted coral cover from shallow (5 m) seaward slope benthic photo transect surveys between 2013 and 2017 for all sites ("Island-wide"), eastern windward sites ("East") and western leeward sites ("West"). Figure reproduced from Raymundo et al. (2019).

difference in coral cover was observed at western sites during this period. However, it should be noted that the small number of eastern seaward slope sites surveyed in 2017 may not be representative of the full extent of the windward side of the island. The declines detected when data from only those sites surveyed in both 2013 and 2017 are considered were highly similar to the values reported above, with a 35% decline, from 26 ± 11% to 17 ± 9% (Partover.test, t = 0.05) in mean coral cover island wide. Mean coral cover for the three east coast sites surveyed in both 2013 and 2017 declined by 63% (from 32 ± 16% to 12 ± 1%), but this change was only significant at the 0.1 level. Mean coral cover for seven west coast sites surveyed in both 2013 and 2017 remained stable at 22%.

Mean coral cover at shallow seaward slope sites across the island in 2021 had not recovered to pre-2013 levels, at 19 ± 10% in 2021 compared to 25 ± 13% in 2013 (Partover.test, t = 0.02; Fig. 6). The island-wide coral cover values are similar when data from only those sites surveyed in both 2013 and 2021 are considered, at 20 ± 10% in 2021 compared to 25 ± 11% in 2023 (Partover.test, t = 0.048). When considering data from only those east coast sites surveyed in both 2013 and 2021, coral cover was 39% lower in 2021 (28 ± 12%) compared to 2013 (17 ± 11%)(Partover.test, t = 0.03), while the coral cover at paired west coast sites remained at 22%.

Examining changes in coral cover at the island scale or along one side of the island, while useful, can miss important changes that are occurring within the reef communities at individual survey sites, as reef communities and their environments at different parts of the island can be remarkably different from each other, and thus these communities may respond differently to stress events. Indeed, changes in mean coral cover at sites surveyed in both 2013 and 2021 were variable (Fig. 7). While most sites did see declines during this period, indicating that coral cover at these sites had not recovered to 2013 levels following the multiple bleaching events between 2013 and 2017, these declines were variable in magnitude, ranging from -80% to <1%. Surprisingly, coral cover actually increased at five of the 17 sites, with increases ranging from 5% to 67%. Two out of the nine east coast sites exhibited increases in coral cover between 2013 and 2021, while three out of the eight west coast sites exhibited increases in coral cover during this period. It is possible, likely even, that variation in the placement of some of the transects from one year

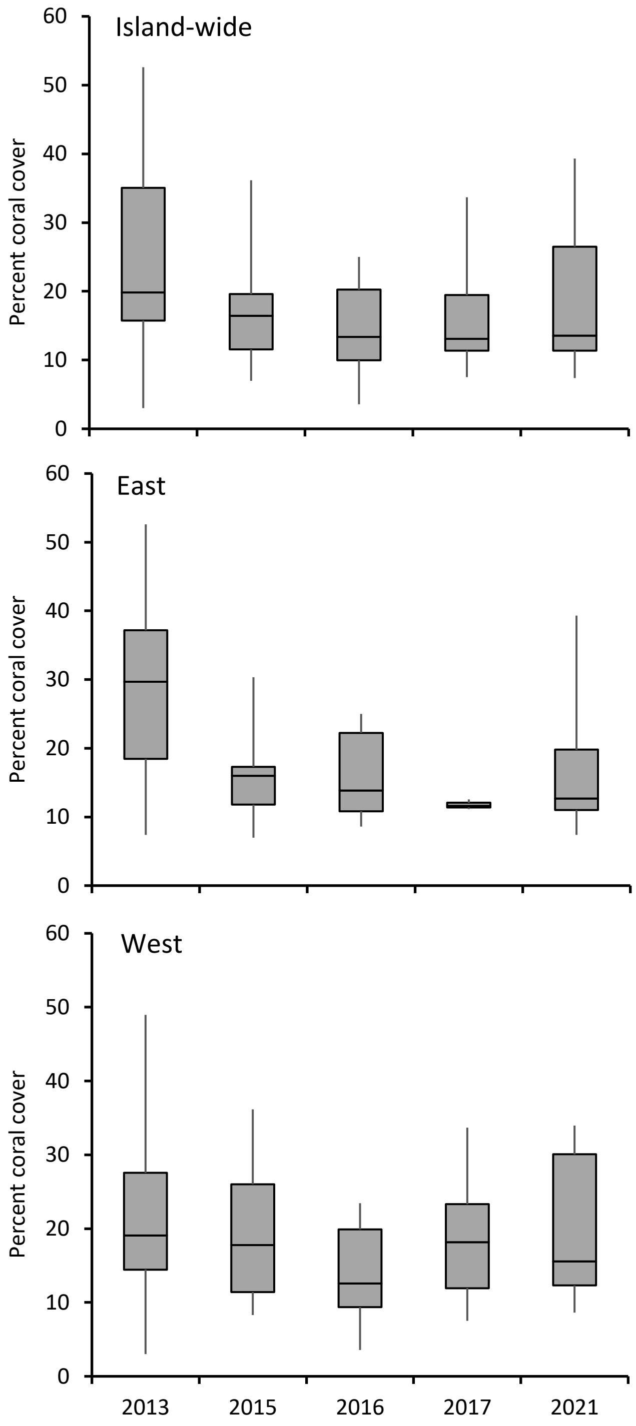
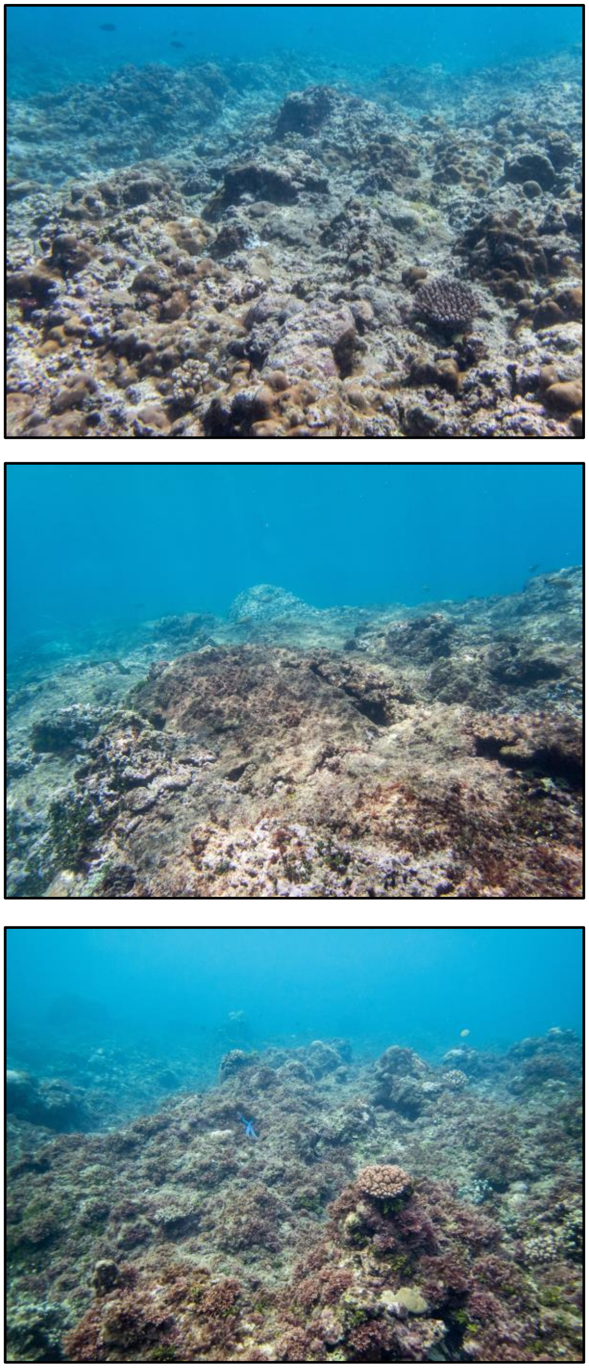


Figure 6. Change in percent coral cover from shallow (5 m) seaward slope benthic photo transcept surveys between 2013 and 2021 for all sites island-wide, eastern windward sites, and western leeward sites. Figure modified from Raymundo et al. (2019) to include 2021 data.



Benthic communities at some of the island-wide bleaching survey sites in 2021, including moderate cover of mostly stress-tolerant taxa along the reef front in East Agana Bay in 2021 (top); low coral cover of a reef front site along the northeastern coast that previously hosted a greater percentage of stress-susceptible coral taxa (middle); and a fleshy algae-dominated benthic community with relatively low coral cover near Ga’an Pt, in southwestern Guam.

to the next contributed to the large magnitude of some of the changes, as rebar was not installed until 2021, but the direction of change and some proportion of the estimated magnitude of the change likely reflect actual changes on the reef.

While coral cover at most of the sites had not returned to 2013 levels by 2021, examining the changes in coral cover between 2017 and 2021—a period with relatively limited thermal stress—can provide an indication of the trajectory of coral cover at the island scale, and at the scale of island side and individual survey sites, following the mass mortality associated with the 2013–2017 bleaching events. When data from all sites surveyed each year are considered, mean coral cover was slightly higher in 2021 compared to 2017 (19 ± 10% compared to 17 ± 9%), but this change was not statistically significant due to the large degree of variability at such a broad spatial scale. Similarly, apparent increases in mean coral cover along the east (12 ± 1% in 2017 to 17 ± 10% in 2021) and west (14 ± 7% in 2017 to 19 ± 10% in 2021) coasts were not statistically significant. When data from only those sites surveyed in both 2017 and 2021 are considered, mean coral cover island wide increased 14% (17 ± 8% to 19 ± 9%), but this change was only significant at the 0.1 level. No statistically significant differences were detected in mean coral cover at east coast or west coast sites surveyed in both 2017 and 2021.

However, the absence of significant gains in coral cover averaged across numerous sites hosting a variety of different reef communities masks notable changes at the scale of individual sites. For example, coral cover increased between 2017 and 2021 at seven of the 12 sites surveyed both years, while it decreased at five of the sites (Fig. 7). Relative to coral cover recorded at each site towards the end of the 2017 bleaching event, percent increases in coral cover by 2021 ranged from 8% to 72%. Five of the eight west coast sites surveyed in both 2017 and 2021 exhibited increases in coral cover, while two of the four east coast sites exhibited increases, one exhibited a decrease, and another remained unchanged.

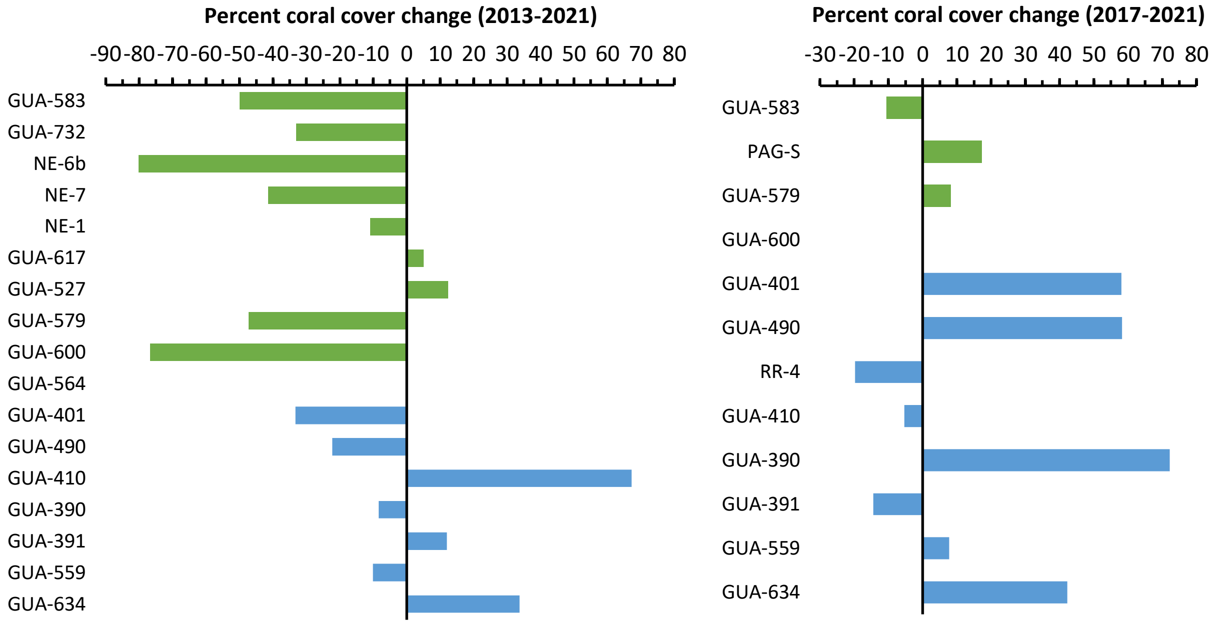


Figure 7. Percent change in coral cover between 2013 and 2021 (left) and between 2017 and 2021 (right) for individual bleaching survey sites. Site locations are shown in the map in Fig. 3. Green bars represent sites along the eastern windward coast, while blue bars represent sites along the western leeward coast.

Next steps

The GLTMP staff are intending to return to the shallow (5 m) transects at the 20 island-wide bleaching response and recovery sites in 2023; once analyzed, the benthic cover data obtained from these now permanently-marked sites should provide a more comprehensive, statistically robust assessment of recovery that may have occurred in recent years. Annual or biennial monitoring is expected to continue at the shallow transects as long as program capacity is sufficient to do so. The establishment and regular monitoring of permanent transects along the 12 m depth contour immediately downslope from the shallow transects—transects along which photo transect surveys were conducted in 2016 and 2017—will also be carried out if program capacity is available.

The GLTMP is also collaborating with Dr. Laurie Raymundo of UOGML and Dr. Gareth Williams of SymbioSeas and Bangor University on a deeper analysis of the benthic cover and coral colony-level data collected at the island-wide sites since 2013. The first analysis project, which was funded by NOAA CRCP, focused on changes in the benthic community beyond just coral cover at a selection of sites for which data were available between 2013 and 2017. While these sites were limited to the west and north coasts, the results of the analysis indicate that the composition of benthic communities at reef front sites along the west coast was generally consistent throughout this period, although a decline in total hard coral cover was detected and some uncommon coral species became rarer or were absent by 2017. A second, still ongoing analysis led by Dr. Williams and funded by NOAA CRCP, is focused primarily on changes in the coral community in response to repeated bleaching events using coral colony size and condition data extracted from the benthic photo transect images.

Due in large part to the relatively small number of sites surveyed during the 2017 event, but also to the high degree of variability across sites at the island-scale, the power to detect change in a metric as coarse as total coral cover between 2013 and 2017 has generally been limited. In addition to establishing permanent transect markers to ensure the same area of reef is surveyed each subsequent year to reduce variability between years and increase statistical power, repeated surveys across several years at numerous sites are required to detect increases in coral cover associated with post-bleaching recovery, as those increases, if they occur at all, are likely to be relatively small and incremental—especially in comparison to the rapid, high magnitude losses associated with severe bleaching events.

Next page: *While the multiple coral bleaching events between 2013 and 2017 resulted in significant coral loss around Guam, particularly in shallow reef areas that were once dominated by stress-susceptible taxa such as Acropora spp. and Montipora spp., in 2021 GLTMP biologists encountered some reef areas, such as a site near Pati Pt. (top) and another south of Talofofo Bay (bottom), that still hosted significant cover of these stress-susceptible taxa, suggesting that these sites are more resilient to thermal/light stress events and have not been significantly impacted by COTS.*

Examples of stress-susceptible corals


Staghorn *Acropora* spp.


Bushy *Acropora* spp.


Stylophora mordax


***Montipora* spp.**

Examples of stress-tolerant corals


Mounding *Porites* spp.


Goniastrea retiformis


Porites rus


Pocillopora setchelli

SUMMARY OF ISLAND-WIDE STAGHORN MORTALITY ASSESSMENT RESULTS: 2013-2021

Key findings

- More than half of Guam's staghorn corals died as a result of coral bleaching events in 2013 and 2014
- Surveys in 2017 showed that the amount of live staghorn coral at individual sites declined by 29–100% compared to pre-2013 levels
- Surveys in 2020/2021 showed further declines, with live staghorn no longer found at Sharks Hole and Pugua Patch Reef (Double Reef), and the lack of recovery at major staghorn sites in Tumon Bay, Cocos Lagoon, and West Hagåtña Bay
- Staghorn communities in Guam's waters have become increasingly reliant on a single species; other staghorn species may soon be lost without substantial intervention

Introduction

Arborescent *Acropora* species, known commonly as staghorn corals, provide important habitat for fishes (Johnson et al. 2011) and invertebrates on tropical coral reefs and can reduce the amount of wave energy that impacts shorelines (Kuffner and Toth 2016). Staghorn corals are among the coral species considered most vulnerable to the effects of climate change, particularly to thermal stress events, which are expected to increase in frequency and severity as sea surface temperatures continue to rise. Prompted by the observed decline in Guam's staghorn coral communities in recent decades and the increasing threat of climate change, members of the GLTMP, in collaboration with UOGML researchers and the National Park Service (NPS), opportunistically mapped the majority of Guam's staghorn coral thickets between 2009 and 2013. This baseline inventory of Guam's staghorn thickets has been critical for understanding the contribution of these unique coral communities to Guam's coral reef ecosystem and in assessing changes in their extent and condition over time.

The staghorn mapping effort provided the foundation of a 2014/2015 staghorn mortality assessment led by Dr. Laurie Raymundo of the UOGML and supported by the MPC. This initial island-wide mortality assessment was prompted by observations of catastrophic mortality at several of Guam's staghorn coral communities in association with bleaching events in 2013 and 2014, as well as mortality associated with subaerial exposure during a period of ENSO-related extreme low tides in 2014/2015. The results of the assessment were published in Raymundo et al. (2017). A second mortality assessment was carried out in 2017 following observations of additional mortality associated with a bleaching event in 2016. The cumulative results of the 2015 and 2017 assessments were presented in Raymundo et al. (2019). A third, and the most comprehensive and quantitative, assessment of the condition of Guam's staghorn corals was carried out by the Raymundo Lab in 2020/2021 to establish a new baseline against which the success of active restoration efforts could be measured (Raymundo et al. 2022). A brief summary of the methods and results of the mapping effort and the mortality assessments are presented below.

Next page: *The largest known thicket of the staghorn coral, Acropora muricata, in Guam's waters (Apra Harbor) at the peak of the severe bleaching event in October 2017, with much of the thicket bleached or recently killed by the heat stress and a concurrent disease outbreak (top). The algae-covered skeletons of the same thicket in March 2021. Some live coral was observed in the thicket in 2021, but the remaining coral represents less than 5% of the live coral prior to the 2017 event.*

Methods

Shallow reef areas around Guam where staghorn coral thickets were known or expected to occur were investigated opportunistically between 2009 and 2013 by GLTMP team members and an NPS intern. The GPS data recorded during the surveys, in combination with satellite imagery, were used within a Geographic information System by the MPC to create polygon features delineating the estimated extent of staghorn thickets. Between November 2014 and February 2015 the MPC assisted Dr. Raymundo with a semi-quantitative assessment of the extent and mortality of all known staghorn *Acropora* populations around Guam. The location and pre-bleaching extent of the staghorn communities were derived from the spatial data layer resulting from the above-mentioned mapping effort. Surveys involved the visual estimation of percent mortality, which was multiplied by the pre-bleaching areal extent to arrive at post-bleaching areal extent and calculations of net staghorn coral loss. Follow-up surveys of the same 21 populations were carried out between February and May 2017 (prior to bleaching that year) in response to additional mortality observed following the 2015 surveys. The 2017 surveys employed a more quantitative approach in which coral condition was assessed at 16 points within replicate quadrats placed on staghorn thickets or colonies. A total of 23 staghorn sites were surveyed by the Raymundo Lab between 2020 and 2021. At each site the periphery of contiguous staghorn thickets was mapped with a GPS receiver and replicate point intercept transects were carried out within each thicket to determine the cover of live coral and other benthic cover types. Replicate 0.25 m^2 quadrats (n = 16 pts per quadrat) were also surveyed along each transect to assess species composition and the percentage of live and dead staghorn. The estimated mean extent of staghorn for each site was calculated by multiplying the total area of the thicket by the percentage of staghorn (live and dead) within that area.

Results and Discussions

Prior to the 2013 bleaching event, staghorn coral populations, which were primarily dominated by *Acropora* cf. *pulchra*, were estimated to have covered a total of approximately 0.33 km^2 of reef area around Guam (see Fig. 1.2 in Burdick et al. 2019). Most staghorn populations occurred on shallow reef flat platforms along the western coast, but some small thickets occurred in Apra Harbor, Cocos Lagoon, Pugua Patch Reef (Double Reef), and Togcha Bay. A comprehensive analysis of available historical data has not yet been conducted, but there are indications that *Acropora* cf. *pulchra* may not have been as dominant in past decades, and the presence of extensive staghorn rubble areas in Cocos Lagoon, Tepungan Bay, and Ga'an Pt. indicate staghorn was once more abundant in these areas.

Cumulative staghorn coral mortality from elevated SSTs and extreme low tides between 2013 and 2015 was estimated at 53 ± 10% by Raymundo et al. (2017). One large population in Agat, southwestern Guam, suffered 100% mortality, while eight other populations experienced 70% or greater mortality. Staghorn populations reassessed in 2017 exhibited reductions in live cover ranging from 29% to 100% compared to pre-2013 populations. Four of the surveyed thickets were devoid of any living tissue, three showed persistent mortality of more than 70% of the pre-2013 population, while seven populations had coral cover values of 50% or greater (Fig. 8). While the different methods employed during the 2020/2021 assessment prevent direct comparison to the results of prior assessments, both the quantitative data and qualitative observations generated by the 2020/2021 assessment indicated further loss in the

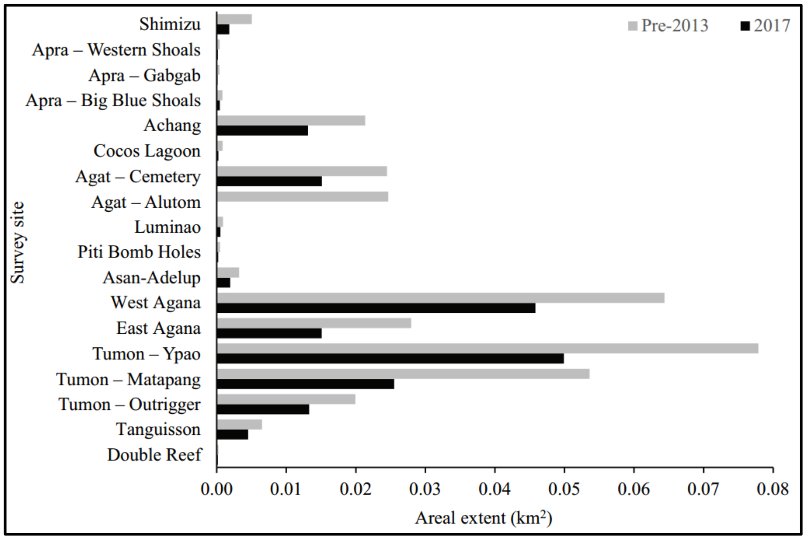


***Figure 8**. Change in estimated areal extent of 18 staghorn *Acropora* populations around Guam. The graph depicts the areal extent of the sites surveyed prior to 2014 compared to the estimated percent lost by 2017. Figure reproduced from Raymundo et al. (2019).*

extent, condition, and diversity of Guam's staghorn communities, likely as a result of the severe bleaching event in 2017, disease activity, and other stressors (Raymundo et al. 2022; Fig. 9). For example, no live staghorn coral was found in 2020/2021 at two sites (Pugua Patch Reef and Sharks Hole) at which live staghorn coral was reported in 2017. The 2020/2021 assessment also revealed that extensive dead staghorn thickets in Tumon, West Hagåtña, and Cocos Lagoon did not exhibit any signs of recovery. Changes in the extent and condition of staghorn communities in the vicinity of the HPRAs are discussed in the "Discussion" section for each of the respective HPRAs.

The staghorn mortality assessments and additional observations made by the Raymundo Lab and the MPC in recent years indicate that several of Guam's staghorn coral species are at the precipice of extirpation from the island's reef system, and the important habitat and function once provided by multiple staghorn coral species is now provided by a single species, *Acropora* cf. *pulchra*, that itself has seen significant impacts in recent years. Of the eight species recorded in Guam's waters, four are now found only at a single location, often reduced to a single thicket or scattered colonies. To complicate matters, the identity of many of Guam's staghorn coral species is currently in question, as recent research suggests a much greater diversity, and much smaller ranges, among members of the coral genus *Acropora* than has previously been recognized. Efforts to resolve the taxonomic uncertainty surrounding Guam's staghorn species by the UOG Biorepository, the Combosch Lab, and the Raymundo Lab are currently underway, while the Raymundo Lab and other members of the Guam Reef Restoration and Intervention Partnership (GRRIP) have established two in-water coral nurseries and are actively engaged in efforts to preserve and restore multiple staghorn species in Guam's waters. A recently awarded grant to the UOGML also supports the development of a pilot project involving the use of a small-scale aquarium facility for the preservation of rare coral species.

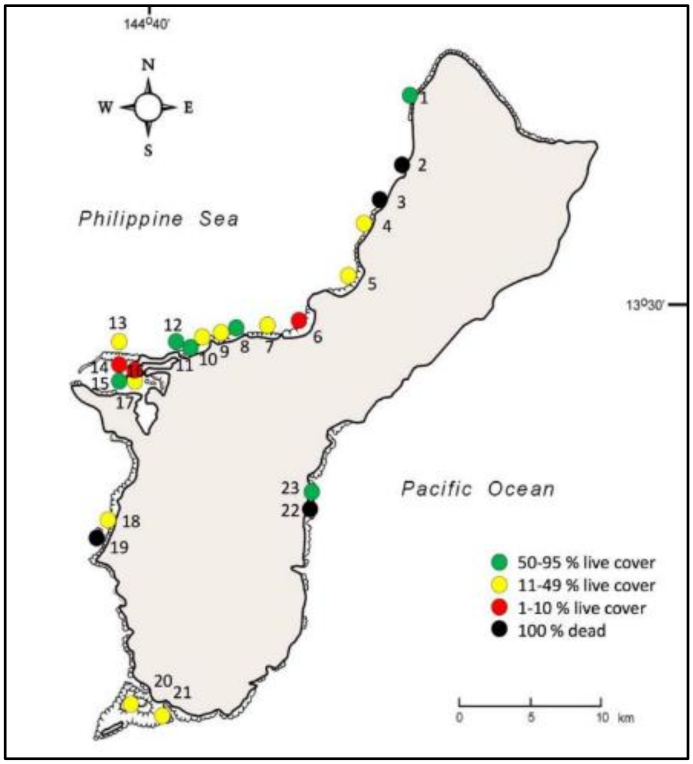


Figure 9. Map of Guam showing the location and condition of staghorn populations assessed in 2020/2021. Figure reproduced from Raymundo et al. (2022).

The Acropora aspera thicket on the reef flat in the eastern portion of Cocos Lagoon represents the last known remaining location of this species in Guam's waters (left). The largest known thicket of A. cf. vaughani in Guam's waters was located at Gabgab II, in Apra Harbor (middle), but this thicket was reduced to rubble after mortality caused by bleaching events in 2013 and 2014, shown here in 2016 (right).

CHANGES AT REEF FLAT MONITORING SITES: 2009–2022

Key findings

- Variability in the responses of live coral cover at the five reef flat monitoring sites to severe stress events highlights the heterogeneity among the coral communities, environmental conditions, and exposure to stress on Guam's reef flat platforms
- Sites comprised primarily of stress-susceptible staghorn corals generally exhibited more dynamic changes in coral cover, and exhibited overall declines of 10–50% between 2009 and 2022
- Coral cover was generally stable at the Piti site, which is dominated by stress-tolerant corals, but an overall decline of more than 20% was observed at the Luminao site by the end of 2022
- Coral disease outbreaks recorded at several reef flat sites have resulted in significant mortality of staghorn corals and other coral species

Introduction

The Reef Flat Monitoring Program (RFMP), led by Dr. Laurie Raymundo of the UOGML involves regular data collection at five sites along the northwestern coast. This project emerged from a 2009 study examining sewage nutrient eutrophication and its impact on coral disease along Guam's northwestern coast (see Redding et al. 2013). Upon completion of the study, local management agencies requested that the monitoring of a subset of these sites be continued as part of Guam's Long-term Coral Reef Monitoring Program. The RFMP has provided important, detailed information about trends in coral community health, and places particular attention on coral diseases, predators, bleaching and other coral health concerns and the relationship with water temperature and nutrients. The data generated through the RFMP provides a strong complement to the data collected at the seaward reef slope sites targeted by the GLTMP. The RFMP provides information to managers for a critical, dynamic, yet vulnerable reef zone, and is an essential component of a comprehensive coral reef monitoring strategy. The results of analyses of data collected by the RFMP were presented in previous reports, such as Burdick and Raymundo (2018). Coral health data collected by the RFMP also served as the basis for a study of the drivers of coral disease published in the scientific journal Frontiers in Marine Science in 2020 (see Green et al. 2020). The results of an updated analysis of changes in coral cover at the reef flat monitoring sites between 2009 and 2022 are presented below.

Methods

Monitoring occurs semi-annually (quarterly through 2020) at five sites, including Tumon, Tanguisson, West Hagåtña, Piti, and Luminao (Fig. 10). Three 20-meter permanent transects were established at each site in 2009. Transect markers have been lost during storms at certain sites, but new ones were established within the same coral community patch during the next census. A line intercept transect method is used to collect data on benthic composition and a one-meter-wide belt transect is used to survey coral health impacts. All colonies within the belt are identified to species and visually inspected for coral disease, predation, competitive overgrowths, bleaching, and partial mortality of unknown cause (after Raymundo et al. 2009). Maximum colony diameter estimates were incorporated beginning in 2011. Temperature loggers have been deployed at the Tumon, Piti and Luminao sites since 2009. Mean (± SD) quarterly percent live hard coral cover values were generated by averaging the values obtained at each of the transects surveyed each quarter, while mean annual percent cover values were generated by averaging the quarterly values for each year.

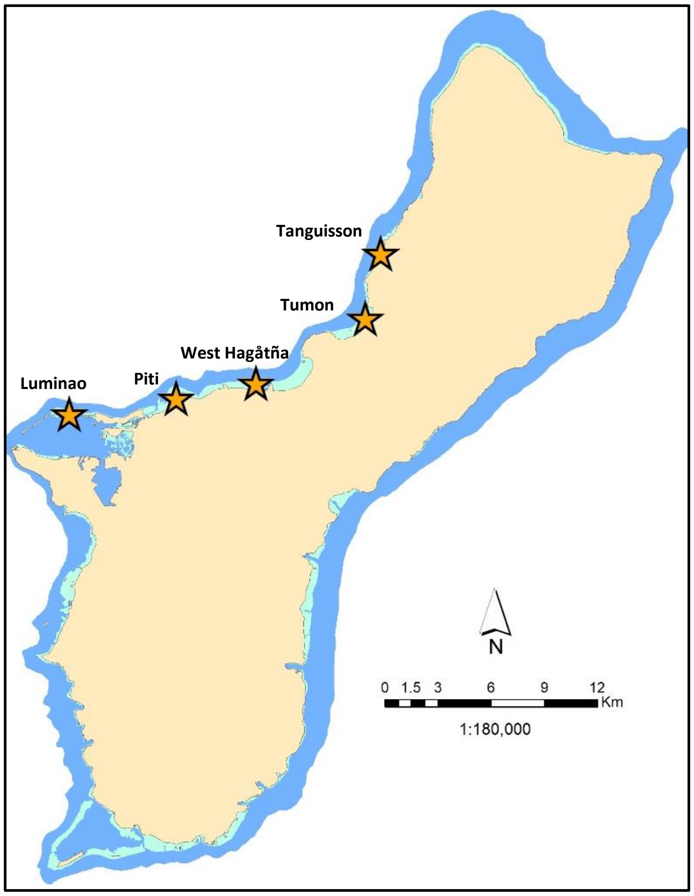


***Figure 10**. Location of Reef Flat Monitoring Program sites.*

Results and Discussion

The RFMP captured several significant events that impacted Guam's coral reefs since 2009. The RFMP previously documented coral loss averaging 27% across the five sites over a 10-year period (2009–2019) that encompassed repeated bleaching episodes and an ENSO-related year-long extreme tide event (Burdick and Raymundo 2018). The average loss in coral cover at reef flat monitoring sites for the period between 2009 and 2022 was slightly less, at 24%, owing to a notable increase in cover at the Tumon site in recent years. However, total coral cover decline within individual sites varied greatly, with sites supporting extensive staghorn *Acropora* populations generally showing the greatest losses. A detailed account of changes in coral cover at each of the five reef flat monitoring sites is presented below.

The West Hagåtña reef flat monitoring site is located on the relatively shallow outer portion of bay's reef flat, approximately 100 m northwest of the Hagåtña wastewater treatment plant (Fig. 13a). The coral community at the site is primarily comprised of the staghorn species, *Acropora* cf. *pulchra*, but other corals such as *Pavona* spp., mounding *Porites* spp., and *Psammocora* spp. are also common at the site. While statistical comparisons were not carried out for this preliminary analysis, coral cover at the original transects established at the West Hagåtña site was relatively stable (~15–17%) between 2009 and 2013 despite severe heat stress in 2013, with the exception of a possible short-term drop to 10% cover in 2010 (Fig. 12). Bleached corals were recorded during a Sept. 2013 visit to the site, but little recent mortality was recorded[2]. Coral cover at the new West Hagåtña transects, which were established in 2014 after the original transect locations had been lost, was similar before and after the 2014 heat stress event (27% and 30%, respectively), suggesting that the corals at the site were not significantly impacted by that event either. Exposure to a greater degree of water movement at the West Hagåtña site, which is located relatively close to the reef margin, and the dominance of a single, relatively stress-tolerant staghorn species (*A*. cf. *pulchra*) may have mitigated the impacts of the 2013 and 2014 bleaching events at the site[3]. A significant (~50%)

---

[2] The relatively limited impact of the 2013 event on some of Guam's staghorn coral species was noted by the MPC and others at the time, but within two-to-three weeks of the unusually early onset of heat stress in 2014 rapid, extensive mortality was observed at several sites

[3] This contrasts with the severe loss of live staghorn coral observed at a large thicket occurring closer to shore in West Hagåtña Bay. Benthic cover and bleaching condition data obtained at this extensive thicket, which is primarily comprised of *A*. cf. *pulchra,* by the GLTMP in Sept. 2013 showed a clear gradient in the prevalence of bleaching and bleaching-associated mortality, with corals occurring at the shoreward side of the thicket exhibiting significantly greater bleaching impacts compared to those at the seaward side (Raymundo et al. 2017).

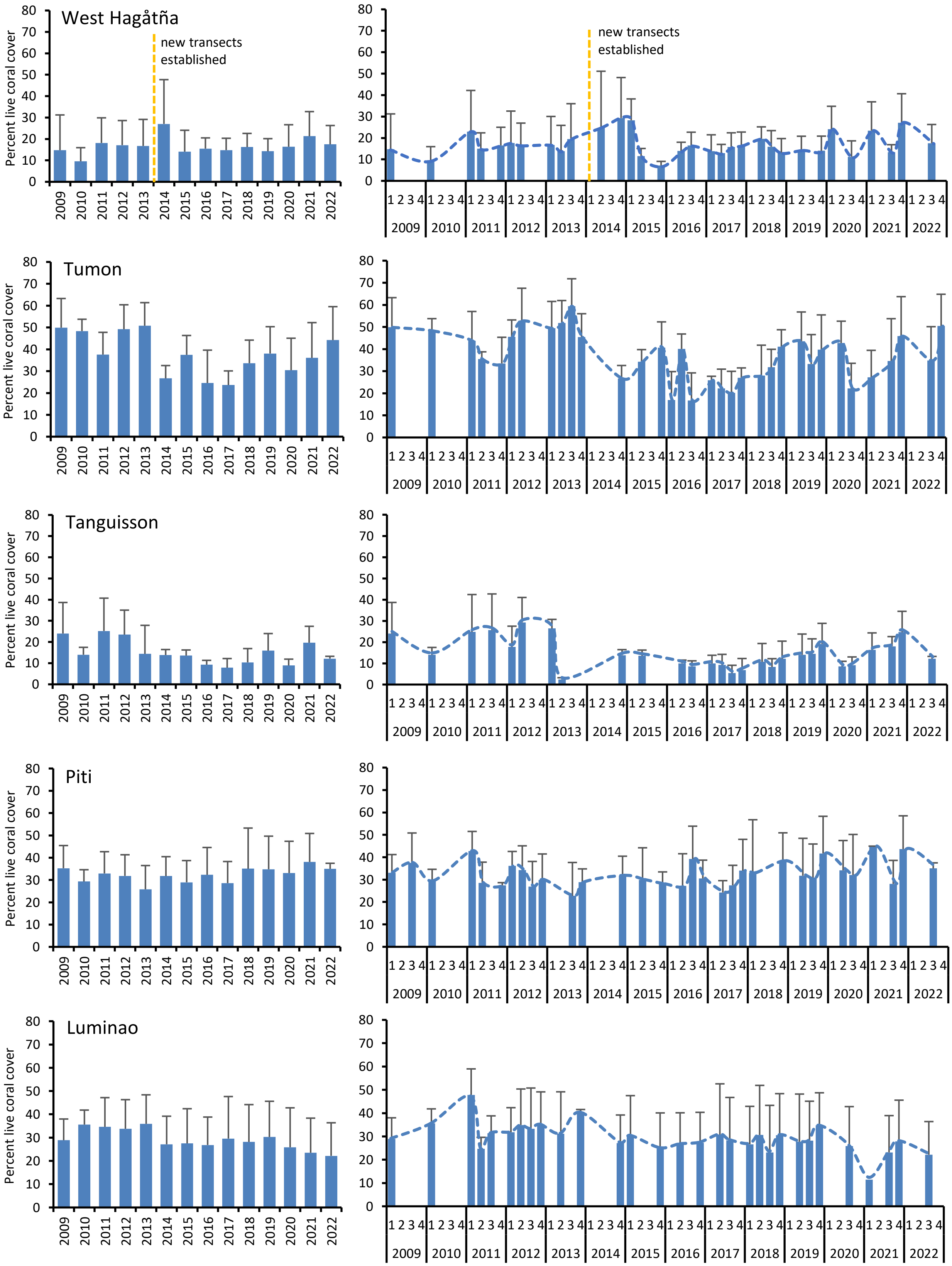


Figure 12. Mean annual (left) and mean quarterly (right) live coral cover at the reef flat monitoring sites between 2009 and 2022.

drop in coral cover at the West Hagåtña site in 2015 was likely associated with the extreme low tides that occurred throughout the year. The location of the West Hagåtña site on the generally shallower outer reef flat in the bay may have made the corals at the site more vulnerable to subaerial exposure than corals located in the slightly deeper inner reef flat zone. Coral cover remained relatively stable (~16–18%) at the West Hagåtña site between 2015 and 2022, despite the stress associated with the 2016 and 2017 thermal anomalies, with a possible brief increase to 21% in 2021. Mean annual coral cover in 2022 (18%) may represent an increase (20%) relative to cover following the 2017 heat stress event, but represents a 33% decrease relative to the 28% baseline recorded at the new transects in 2014.

The Tumon Bay reef flat monitoring site is located ~200 west of the Dusit Thani Guam Resort and occurs on the slightly deeper inner portion of the reef flat platform. The Tumon site is primarily dominated by *A*. cf. *pulchra,* yellow finger coral (*Porites cylindrica*), and other *Porites* spp., but also hosts *A. acuminata*, a staghorn species that appears to be more susceptible to bleaching than *A*. cf. *pulchra* (Fig. 13a). Like the West Hagåtña site, coral cover at the Tumon site also appeared relatively stable (~50–51%) between 2009 and 2013 despite any impacts that may have been associated with the 2013 bleaching event, although coral cover may have dropped to 38% in 2011 before quickly recovering by 2012 (Fig. 12). While the Tumon site was only visited during the fourth quarter in 2014, coral cover during that visit had dropped to 27%, representing a 40% decline from the 45% cover value recorded in the fourth quarter of 2013. The decline in 2014 was likely due to the impacts of the 2014 bleaching event, which were observed firsthand by the MPC at Ypao Beach, located at the southwestern end of Tumon Bay. Coral cover at the Tumon site actually appeared to increase through 2015, despite the severe impacts observed at other sites in response to the extreme low tides that year. The slightly greater depth of the Tumon site within the inner reef flat zone may have mitigated the impacts of the unusually low tide levels. Coral cover appeared to decline by ~34% between 2015 (38%) and 2016 (25%), and remained between 20% and 27% through 2017 before steadily increasing through mid-2020. The losses in live coral in 2016 and the lack of any detectable recovery in 2017 is likely a result of the heat stress events recorded both of those years, while the increase after 2017 is likely attributable to coral growth unimpeded by significant heat stress events or disease outbreaks. A significant decrease in coral cover between quarters 2 and 3 in 2020 was likely due to bleaching-associated mortality. Coral cover increased steadily through the end of 2021 following this significant decline. By the end of 2022, mean annual coral cover at the Tumon site was 44%, which represents a notable 83% increase from the low of 24% in 2017 following repeated stress events. However, cover in 2022 was still slightly lower than the ~50% cover recorded prior to the 2013–2017 bleaching events and the extreme low tides of 2015.

Bleaching-associated mortality in the staghorn coral species, Acropora cf. *acuminata, in the vicinity of the Tumon reef flat monitoring site in July 2014, at the peak of heat stress that year (top), and bleached A. muricata at Ypao Beach, also in July 2014. Greater than 95% of all A. muricata at Ypao Beach were killed in 2014, and the remaining living tissue has since perished, resulting in the extirpation of that species from Tumon Bay.*

The Tanguisson reef flat monitoring site is located ~450 m northeast of Tanguisson Pt. and straddles the shallow outer portion and the slightly deeper inner portion of the narrow reef flat platform. The site's occurrence on a narrow reef flat platform exposes it to significant water movement and flushing with open ocean water; however, the site is also relatively close to freshwater seeps and springs found closer to shore. The coral community at the site hosts significant *A*. cf. *pulchra* cover, but several *Porites* species, *Pavona decussata*, and *Psammocora contigua*

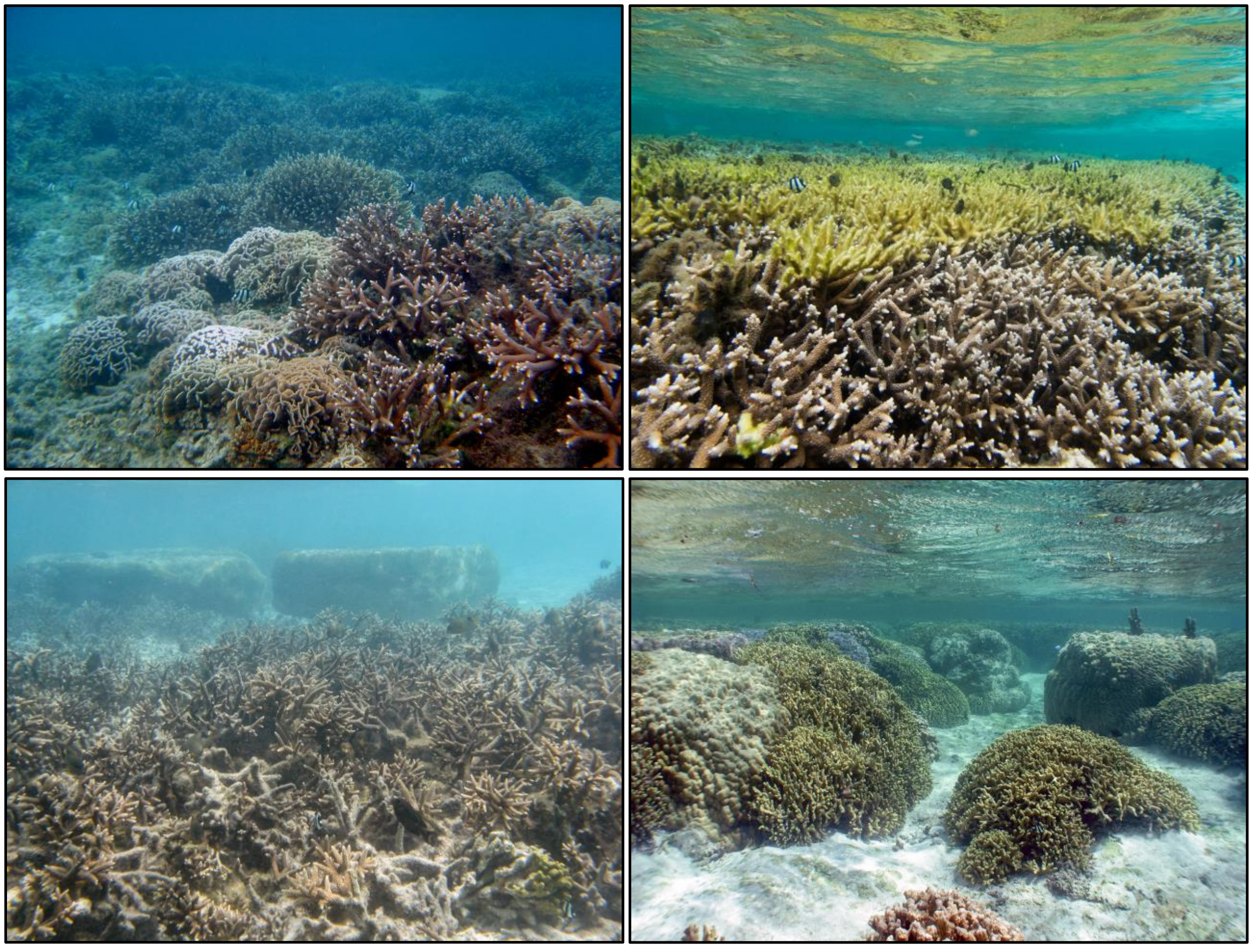
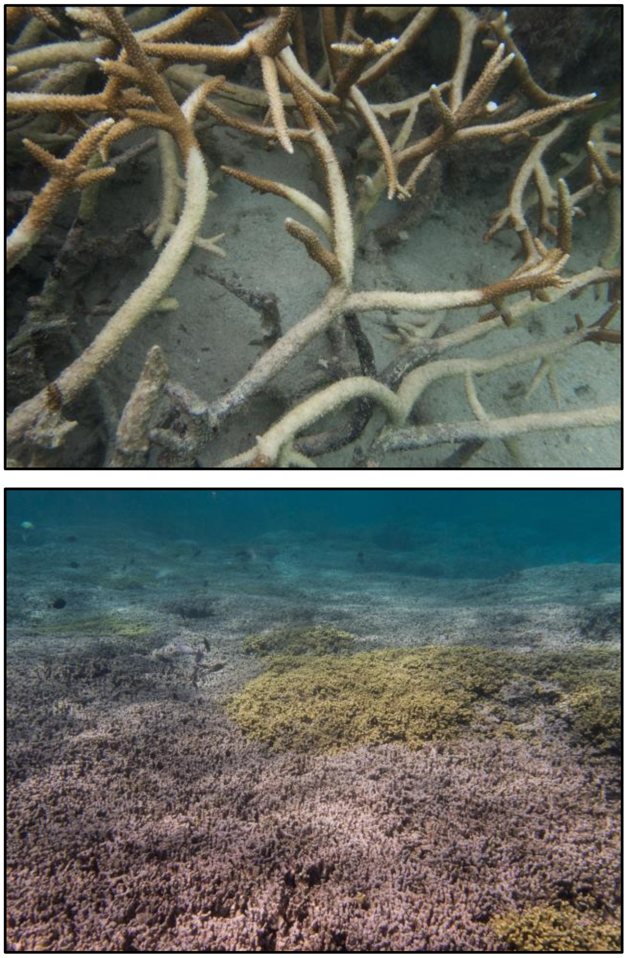


***Figure 13**.* An *Acropora* cf. *pulchra* and *Pavona*-dominated community in the vicinity of the West Hagåtña reef flat monitoring site in 2007, prior to the commencement of regular monitoring in Jan. 2009 (a); *Acropora* cf. *pulchra* with a moderate degree of subaerial exposure-associated mortality in the vicinity of the Tanguisson reef flat monitoring site in May of 2020 (b); a mix of *Acropora* cf. *pulchra*, *A*. *acuminata*, and mounding *Porites* in the vicinity of the Tumon reef flat monitoring site in Feb. 2014, after the severe heat stress in 2013 but prior to the onset of heat stress in 2014 (c); a coral community comprised primarily of mounding *Porites* and *P*. *cylindrica* in the vicinity of the Luminao reef flat monitoring site in May 2010 (d).

are also common at the site (Fig. 13c). Coral cover at the Tanguisson site exhibited a notable 38% decline in 2010, similar to the declines observed at the West Hagåtña and Piti reef flat monitoring sites, but it quickly recovered to baseline levels (24%) in 2011, where it remained through 2012 (Fig. 12). The cause of the 2010 decline is not clear from an analysis of the coral cover data alone, but a deeper analysis of cover data along each transect, in combination with the colony size and condition data and available environmental data and anecdotal reports, may provide some indication of the possible drivers of this decline. A precipitous (~90%) drop in coral cover at the Tanguisson site that occurred between quarters 1 and 2 in 2013, prior to the onset of heat stress in 2013, may have been caused by a disease outbreak at the site. While sampling during 2013 and 2014 was limited, coral cover appeared to have increased between the 2nd quarter of 2013 and the 4th quarter of 2014, despite the heat stress events in 2013 and 2014. However, mean annual coral cover remained relatively low (~14%) through 2015. Coral cover may have decreased between 2015 and 2016, and while sampling was limited those years the timing of the decline suggests that it may have been due, at least in part, to stress associated with the 2015 extreme low tides in 2015 rather than the 2016 heat stress event. A moderate loss in coral cover (from 9% to 5%) recorded during quarter 3 of 2017 may be related to the severe bleaching event that year, and possibly disease activity as well. The overall decline in cover between 2013 and 2017 is likely due to the multiple heat stress events, disease activity, and the 2015 extreme low tides. The post-2017 period was characterized by two cycles of steady coral cover increases followed by rapid losses. The increases in coral cover indicate conditions were favorable for coral growth during these ~2 year-long periods, but the two significant (~40%) losses in 2020 and 2022 suggest the coral community at the Tanguisson site is in a dynamic, vulnerable state. An examination of other quantitative and qualitative data obtained at the Tanguisson site may provide some indication of the possible drivers of these boom-and-bust cycles. Mean coral cover at the Tanguisson site in 2022 (12%) represented a 50% increase from the 2017 low of 8%, but actually represents at 50% decrease relative to the 24% cover value recorded in 2009, when the site was first established, and in 2012, the year immediately prior to the onset of multiple acute disturbance events.

The Piti reef flat monitoring site is located on the inner portion of the reef flat in Tepungan Bay, ~200 m to the northeast of the Fish Eye Marine Park observatory. In contrast to the significant declines detected at the sites with significant staghorn cover, and the generally more dynamic state of the coral community observed at those sites, coral cover at the *Porites*-dominated Piti site remained relatively stable between 2009 and 2022 (Fig. 12). Mean annual coral cover may have declined by ~20% between 2012 and 2013, a period that includes the 2013 heat stress event. However, data collection at the Piti site did not occur in the first half of 2013, so it is not entirely clear that the mortality that occurred during this period can be entirely attributed, directly or indirectly, to the 2013 heat stress event. An examination of the colony size and condition data and anecdotal observations may provide some indication of the probable driver(s) of this moderate decline. Coral cover at the Piti site appeared to remain stable, or possibly even increased somewhat, after 2013, suggesting that neither the 2014 heat stress event nor the extreme low tide events in 2015 significantly affected total coral cover at the site. Coral cover may have increased slightly in 2016, and then declined in 2017—possibly as a result of impacts associated with the 2017 heat stress event, but these relatively modest differences in cover may not be statistically significant. By 2018, coral cover at the Piti site recovered to the baseline reported when surveys began in 2009 (35%) and remained between 33% and 38% through 2022. While some periods of decline associated with heat stress, subaerial exposure, and possibly other stressors appear to have occurred at the Piti reef flat monitoring site between 2009 and 2022, total coral cover at the site remained relatively stable throughout this period. However, an examination of the colony size and condition data should be carried out to determine if there was a detectable shift in community composition, despite the apparent stability in total coral cover.

The Luminao reef flat monitoring site is located on the inner portion of the reef flat platform of Luminao Reef, a barrier system over which the Glass Breakwater was built shortly after World War II. Similar to the Piti site, the Luminao reef flat monitoring site hosts a coral community comprised primarily of *Porites* corals, such as *P. cylindrica*, *P. rus*, and mounding *Porites* spp., but unlike the Piti site *Acropora* cf. *pulchra* is also present (Fig. 13d). While coral cover was more stable at the Luminao site between 2009 and 2022 in comparison to the Tanguisson and West Hagåtña sites, an overall 24% loss in live coral cover was still detected during this period (Fig. 12). Despite a steep (48%) drop in cover recorded in quarter 2 of 2011, mean annual coral cover at the Luminao site remained between 34% and 36% from 2010 through 2013. Sampling between 2013 and 2015 was somewhat limited, but coral cover appeared to decline in 2014, from 41% in the 4th quarter of 2013 to 27% in the fourth quarter of 2014, possibly as a result of heat stress that year. Coral cover never returned to



An unidentified coral disease rapidly progressing from the base of branches in a small Acropora cf. pulchra thicket in the water intake channel next to Cabras Power Plant in August 2022. Significant mortality of the generally stress-tolerant yellow finger coral, Porites cylindrica, observed at Ypao Beach in mid-2022 was likely the result of the same, or highly similar, aggressive coral disease; several other coral species were also affected at the popular recreational site.

the ~35% level recorded in 2013, with recovery likely inhibited by the repeated heat stress events and the 2015 extreme low tides. After a modest potential increase through 2019, coral cover at the Lumina site steadily declined through 2022. The apparent steep drop in cover in the 1st quarter of 2021 is likely an artifact of the limited sampling that quarter, with data collected only along a single transect that had consistently much lower coral cover than the other two transects; mean cover values in quarters 3 and 4 of 2021, which were generated using data collected at all three transects, were similar (23% and 28%, respectively) to mean cover recorded during quarter 3 of 2020 (26%).

While an updated analysis of coral health impacts has not yet been carried out, a previous analysis found that white syndromes were the most prevalent diseases recorded from Guam, and were present at all sites during all census periods (Burdick and Raymundo 2018). However, prevalence was consistently much higher at the *Porites*-dominated sites (Luminao and Piti) than at the *Acropora*-dominated sites (Tumon, Tanguisson, and West Hagåtña). While this group of syndromes affects many species, and is currently thought to be caused by more than one infectious bacterium, *Porites* is known to be a dominant host genus on Guam (Myers and Raymundo 2009). In *Porites*, the disease can be highly prevalent but appears to be chronic and sublethal to the colony. However, it can also manifest as a highly infectious and rapidly progressing disease in an acute outbreak form, which is more lethal to whole colonies. Outbreaks of white syndrome have occurred in *Acropora* populations in Tanguisson and Tumon in 2017 and in *Pocillopora damicornis* in 2018. Outbreaks were observed at several sites during or immediately after bleaching events in 2014, 2016 and 2017, suggesting there may be a link between these two phenomena impacting coral health, and further investigation and monitoring are warranted. However, severe outbreaks of white syndrome or similar diseases have been detected even in the absence of large-scale thermal anomalies. For example, in mid-2022, the rapid mortality observed in multiple species, including *Acropora* cf. *pulchra* and *Porites cylindrica*, by the MPC at Ypao Beach in Tumon Bay, and of *A*. cf. *pulchra* in the cooling water intake channel next to the Cabras Power Plant in Piti, appears consistent with an aggressive white syndrome or similar disease (see images on previous page).

The variability in the responses to several stress events of live hard coral cover at the five reef flat monitoring sites between 2009 and 2022 illustrates the high degree of heterogeneity among coral communities and the reef flat environments in which they occur. Despite this variability, some commonalities in the timing and magnitude of observed changes in coral cover at several sites were evident. The dynamic changes observed at those sites with significant staghorn coral cover demonstrate the capacity of these communities to rapidly lose live coral in response to stress events, as well as their capacity for relatively rapid recovery when conditions are favorable. Coral cover at the two sites dominated by stress-tolerant corals, such as *Porites* spp., was generally more stable, although recent observations of significant, disease-associated losses in *Porites* corals and the relatively slow growth rates of many *Porites* species suggests that these communities are still vulnerable. A more comprehensive analysis of benthic cover, coral colony size and condition, and environmental data available for the reef flat monitoring sites may reveal additional information about changes in the health of benthic communities at these sites and provide further information about possible drivers of these changes so that managers and researchers can develop actions to mitigate stressors and, to the extent practicable, restore the lost functions, goods and services that these critical reef communities provide.

Next page: *A* *Porites*-*dominated reef along the submarine terrace in the Piti (Tepungan Bay) HPRA in 2020.*

CHANGE IN REEF CONDITION AT THE HIGH PRIORITY REEF AREAS

TUMON BAY MARINE PRESERVE

Key findings

- Coral cover was moderate (~30%) and remained relatively stable between 2012 and 2020, despite the multiple bleaching events that affected Guam's reefs during this period
- The stability in coral cover is likely attributed to the dominance of a small number of generally bleaching-resistant coral species, but the low diversity coral community may be vulnerable to disease outbreaks and more severe warming events in the future
- Data and staff capacity limitations currently prevent detailed analyses of reef fish survey data obtained between 2010 and 2021, but there are some indications that food fish biomass may have decreased after 2012 and increased between 2019 and 2021
- The biomass of all food fish families except groupers appeared to have increased between 2019 and 2021
- Total fish biomass in 2019 was moderate (21–41 g/m^2), at 39–77% of the potential total reef fish biomass estimated for an unimpaired Guam reef by Williams et al. (2015)
- The density of small food fishes may have declined between 2015 and 2019, but likely increased between 2019 and 2021
- Significant declines in the densities of sea cucumbers and edible shells (top shells and giant clams) after 2012 is likely directly or indirectly related to heat stress in 2013 and/or 2014

Previous page: *A Porites-dominated reef along the submarine terrace in the Piti (Tepungan Bay) HPRA in 2020.*
Above*: A benthic community, dominated by the plate-and-pillar coral, Porites rus, and mounding Porites species, typical of the submarine terrace reef zone within the Tumon Bay HPRA in 2020.*

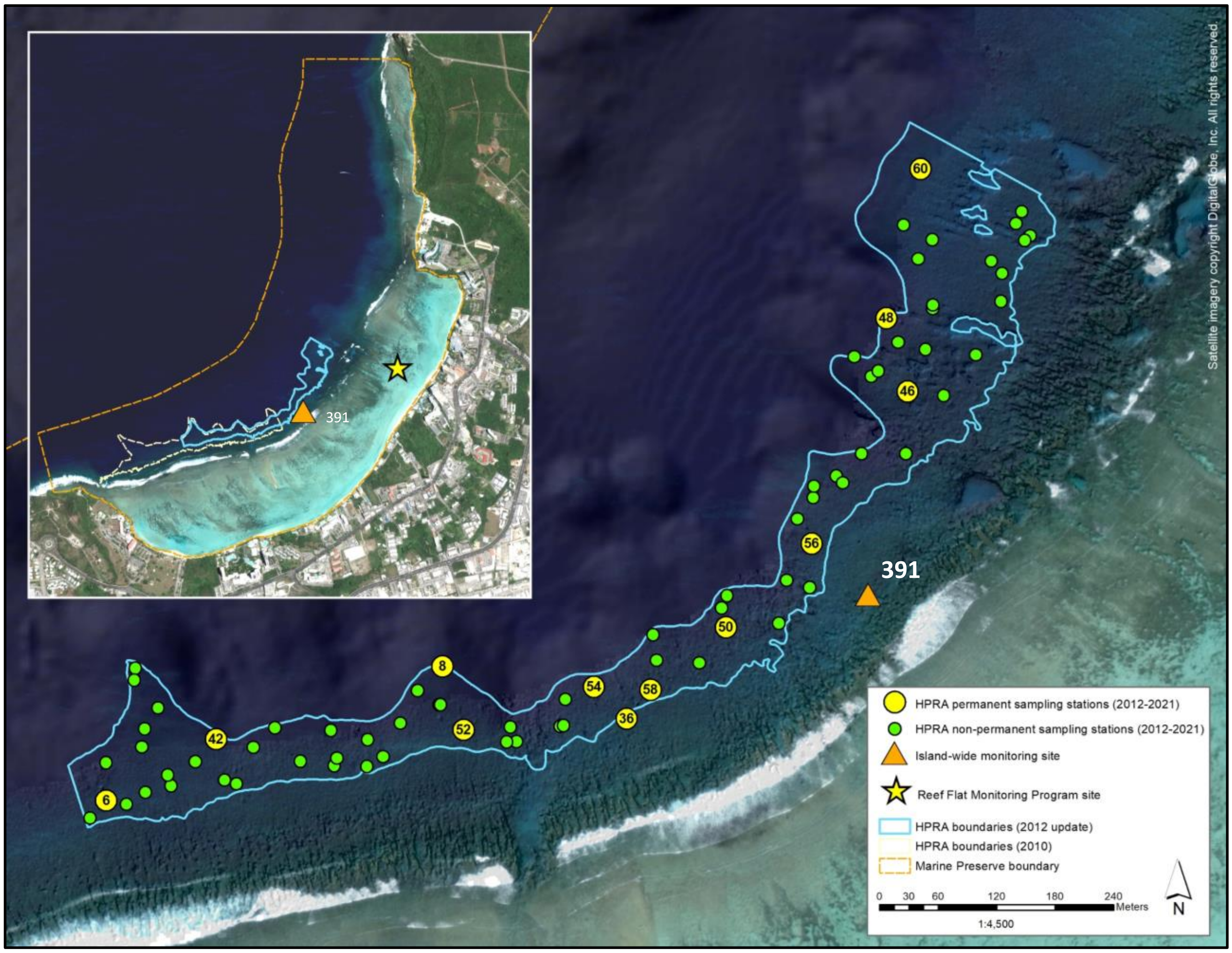


Figure 14. Map of the Tumon Bay HPRA site boundaries and sampling stations, as well as one of the permanently established island-wide bleaching response and recovery sites and the reef flat monitoring site within the vicinity of the HPRA.

Site overview

Tumon Bay hosts the Tumon Bay Marine Preserve and is a hub for tourism, recreational, and cultural activities on Guam. While the numerous impacts to the bay's reef ecosystem, such as coastal development and heavy recreational and commercial use, make discerning individual impacts difficult, the intensive, comprehensive monitoring of a carefully selected portion of the reef increases the likelihood that even relatively small changes in key ecosystem health parameters can be detected, and that indications of probable causes can be provided.

The regular data collection activities carried out within the Tumon Bay site by GLTMP staff are primarily focused on the submarine terrace of the seaward slope reef zone between depths of 7 and 15 m, an area of relatively gentle slope that extends from the base of the reef front to the edge of a steeper lower slope (Fig. 14). The benthic community prevalent within this seaward slope zone is distinct from that observed in the shallower, more wave exposed reef margin and reef front zones, and is dramatically different from the various benthic communities that occur across the shallow reef flat within the bay. A single permanently established island-wide bleaching response and recovery site (391) is located along the reef front near the center of the bay, just upslope of the HPRA, while a RFMP site is located on the shallow reef flat approximately 200 m west of the Dusit Thani Guam Resort.

The seaward slope in Tumon Bay was selected as the first HPRA targeted for long-term monitoring, with partial monitoring within the bay having occurred in June 2009 and a full data collection effort having occurred in 2010. The boundaries of the monitoring site were shifted northwestward after an analysis of baseline data collected in 2010 showed that the original boundaries included two distinct benthic communities and that the large variance in the data limited the ability to detect change in these communities (refer to Burdick and Brown, 2011, for more detail, links on p. 3 of this report). Data collection occurred within the new site boundaries in 2012, 2014–2015, and 2017–2022 (Appendix A). The re-prioritization of resources required to respond to the multiple bleaching events between 2013 and 2017 prevented data collection efforts in the Tumon Bay HPRA in 2013 and 2016, and also limited the data collection efforts for some other years. Baseline fish survey data for the modified Tumon Bay site were not collected until 2015, as surveys could not be completed in 2012 or 2014 because of poor weather conditions and staffing limitations.

In addition to data collected during regular visits to the HPRA site along the seaward slope, benthic cover and macroinvertebrate abundance data was also collected at four shallow (5 m depth) sites along the reef front in Tumon Bay, including two in close proximity to the HPRA site boundaries, in 2013 as part of a UOGML-led bleaching response effort to which GLTMP staff made significant contributions. One of these reef front sites (391) was re-visited in 2015 as part of an island-wide bleaching recovery assessment, in 2016 as part of an island-wide reef resilience assessment, in 2017 as part of the response to another major bleaching event and was surveyed and marked with rebar in 2021. The UOGML Reef Flat Monitoring Program led by Dr. Raymundo has collected benthic cover and coral health data at a reef flat site in Tumon Bay since 2009.

The results of analyses of available benthic cover, reef fish, and macroinvertebrate data obtained within the Tumon Bay HPRA between 2010 and 2021 are provided below. The results of an analysis of data obtained at the reef flat monitoring site in Tumon Bay is presented on p. 40 and summarized in the "Discussion" section below. While a detailed analysis of individual island-wide bleaching response and recovery sites has not yet been conducted, the results of a preliminary analysis of data obtained at individual island-wide sites are presented beginning on p. 26, and results pertinent to the reef front monitoring site in Tumon Bay (391) are referenced in the "Discussion" section below. The results of an analysis of data obtained at staghorn locations throughout the bay are summarized in the "Discussion" section and presented in more detail in Raymundo et al. (2022).

Benthic cover

Coral cover remained relatively stable across the Tumon Bay HPRA between 2012 and 2020, despite the multiple bleaching events affecting Guam's reefs during this period and the regular reports of crown-of-thorns feeding activity in the bay (Fig. 15). Mean coral cover may have declined slightly following the 2013 and 2014 bleaching

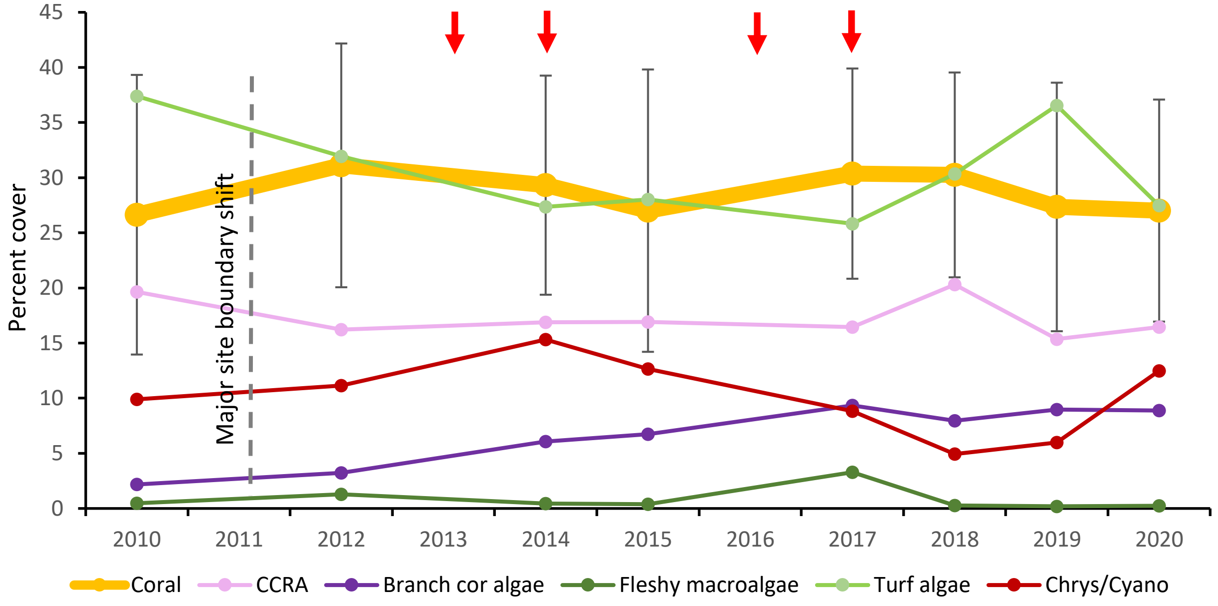


Figure 15. Mean (± SD) percent cover of major benthic cover classes for the Tumon Bay HPRA between 2010 and 2020. Mean cover values for 2010 were calculated using only those stations occurring within the new site boundaries established beginning in 2012. Red arrows indicate the timing of significant coral bleaching events, including the historically severe 2013 and 2017 events. Note that the elapsed period of time between each pair of neighboring data points varies from one to two years.

events (from 31% in 2012 to 27% in 2015; Partover.test, p = 0.1), but the p value of 0.1 suggests caution in assigning significance to this possible change. Mean coral cover remained at 30% between 2017 and 2018, despite a historically severe bleaching event occurring between data collection activities those years.

Total coral cover is an admittedly somewhat crude measure of coral community health, as the decline in the cover of some species can be masked by an increase in a small number of fast-growing “weedy” species, and is best used alongside an examination of changes in individual coral species (or groups of a few closely related species). The coral community within the Tumon Bay HPRA is comprised largely by the plate-and-pillar coral, *Porites rus*, and a small number of mound-forming *Porites* species that cannot be reliably identified to the species level in the field or in images (and thus these few species are all considered “mounding *Porites* spp.”). These species appear to be relatively resistant to bleaching, and while mounding *Porites* species grow slowly, plate-and-pillar coral can grow relatively fast. Benthic cover data indicate that *Porites rus* has comprised >50% of total coral cover in the Tumon Bay HPRA every surveyed year since 2012, while mounding *Porites* spp. comprised at least 25% of total cover and other *Porites* taxa (mainly *P. deformis* and *P*. cf. *lichen*) comprised <10% of total cover during this period; all other coral taxa combined contributed 5% or less to total coral cover (Figs. 16–17).

When data from all permanently marked and non-permanent sampling stations are considered, the percent cover of *Porites rus* in the Tumon Bay HPRA may have declined from 20% to 16% between 2012 and 2020, but this change was not statistically significant (Partover.test, p = 0.15). The cover of *P. rus* was more consistent at the permanent stations, remaining between 17% and 19% throughout this period and with no statistically significant difference between cover in 2012 and 2020 (Partover.test, p = 0.77). The cover of *P. rus* declined from 20% to 14% between 2012 and 2015 (all stations, Partover.test, p = 0.02), potentially as a result of impacts directly or indirectly associated with the back-to-back coral bleaching events in 2013 and 2014. However, *P. rus* at the permanent stations remained at 18% during this period (Partover.test, p = 0.91). Mean percent cover of *P. rus* across all stations did not significantly change after the 2017 bleaching event and may have even increased slightly (from 16% to 19%, Partover.test, p = 0.13). The cover of mounding *Porites* spp. increased from 8% to 10% in the Tumon Bay HPRA between 2012 and 2020 (Partover.test, p = 0.03). Mounding *Porites* spp. cover increased from 8% to 10% between 2012 and 2015 (Partover.test, p = 0.02), despite the coral bleaching events in 2013 and 2014, and remained stable through 2020 despite the 2017 coral bleaching event occurring during this period. The combined cover of other *Porites* taxa, namely *Porites deformis* and *P*. cf. *lichen*, was relatively low (<2.6%) throughout the study period, and

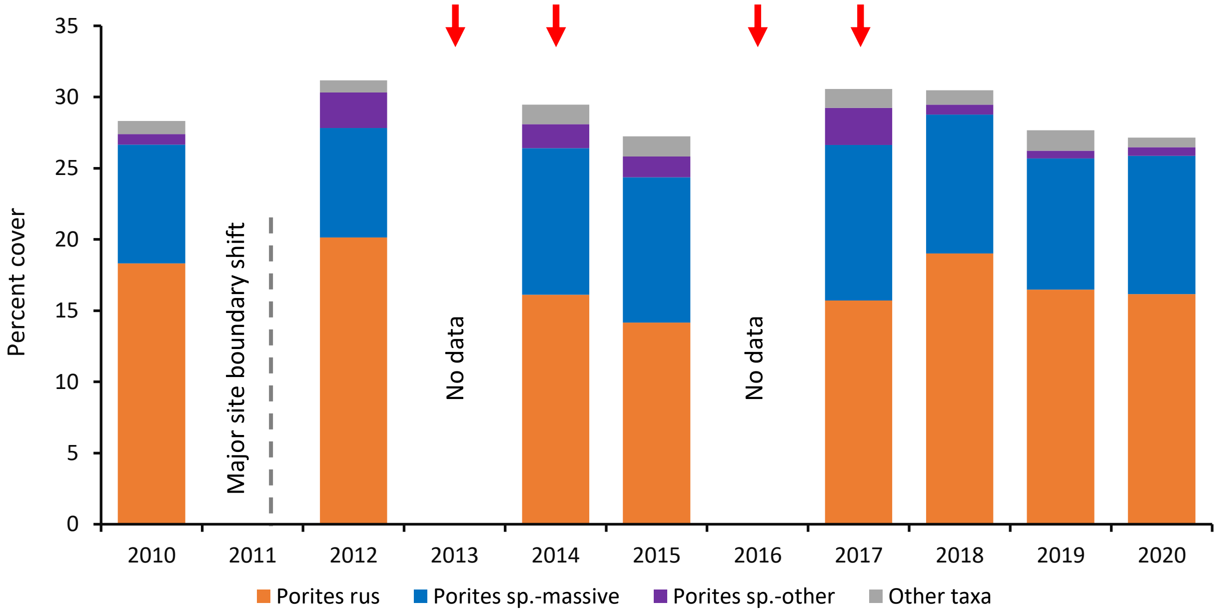


Figure 16. Percent cover of coral taxa for the Tumon Bay HPRA between 2010 and 2020. Mean cover values for 2010 were calculated using only those stations occurring within the new site boundaries established beginning in 2012. Red arrows indicate the timing of significant coral bleaching events, including the historically severe 2013 and 2017 events.

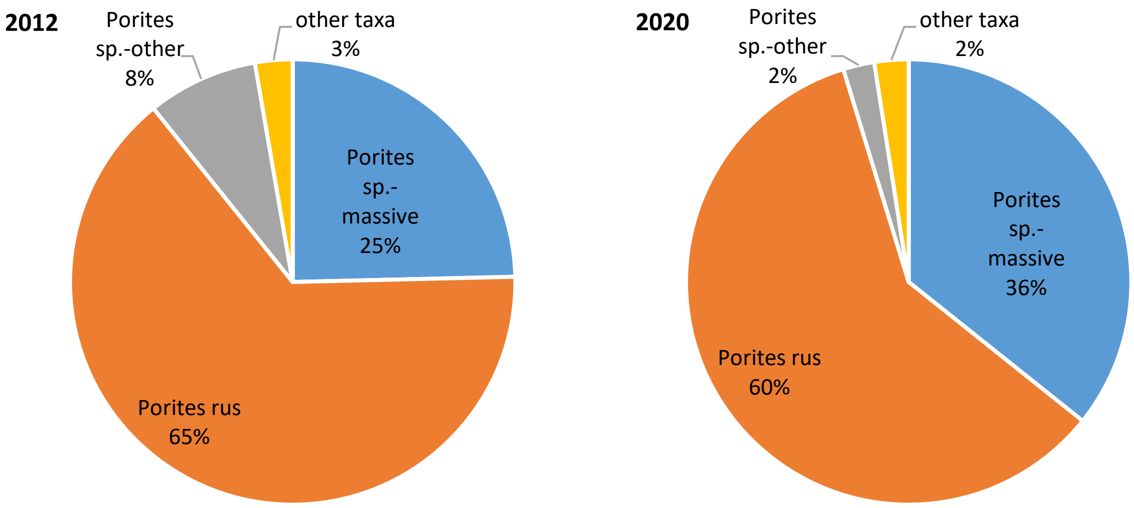


Figure 17. Percent contribution of individual coral taxa/taxa groupings to mean total coral cover for the Tumon Bay HPRA in 2012 and 2020.

decreased from 2.5% in 2012 to <1% in 2020 (Partover.test, $p < 0.001$). Cover first declined from 2.5% to 1.4% between 2012 and 2015 (Partover.test, $p = 0.005$), potentially as a direct or indirect result of the back-to-back 2013 and 2014 bleaching events. After a slight rebound (from 1.4% to 2.6%) between 2015 and 2017 (Partover.test, $p = 0.008$), cover of other *Porites* taxa decreased again (from (2.6% to 0.7%) following the 2017 event (Partover.test, $p < 0.001$) and remained <1% through 2020 (Partover.test, $p = 0.36$). The cover of all other coral taxa combined remained below 1.4% throughout the study period, and with no significant change between cover values in 2012 (0.8%) and 2020 (0.7%). The very low abundances of coral taxa known to be highly susceptible to bleaching impacts, such as *Acropora* and *Montipora,* prevents the detection of change in these taxa by the current GLTMP sampling design and survey and image analysis methodology.

While statistical comparisons were not made for non-coral benthic features through time for the purposes of this report, the cover of fleshy macroalgae—generally considered competitors with corals for space—was very low (<3%) throughout this period (Fig. 15). The cover of branching coralline algae, comprised mainly of the branching coralline algae, *Halimeda* spp., was also relatively low (<9%), but appeared to increase through time (from 3% to 9%). The cover of crustose coralline red algae, which include members of the orders Corallinales (mostly pink or red crusts) and Peyssonneliales (mostly deep red, brown, and rust-colored crusts), remained between 15% and 20% within the Tumon HPRA between 2012 and 2020. The cover of turf algae, which ranges from microturf ("bare rock") to algal growth up to 1 cm in height, remained between approximately 25% and 37%, while the cover of cyanobacteria and chrysophytes ranged from 5% to 15% in cover between 2012 and 2020.

Associated biological communities

Reef fishes

Biomass

Mean food fish biomass within the Tumon Bay HPRA appeared to decline (from ~42–45 $g \cdot m^{-2}$ in 2010/2012 to 23 $g \cdot m^{-2}$ in 2014), then remained stable at around 18–24 $g \cdot m^{-2}$ through 2020 before increasing to 37 ± 61 $g \cdot m^{-2}$ in 2021 (Fig. 18a). However, the very large standard deviation values for some years and the significant inter-observer bias evident in Fig. 18b suggests caution in interpreting potential trends in food fish biomass within the Tumon Bay HPRA during this period. Unfortunately, even if there was an actual decline in food fish biomass within the Tumon Bay HPRA between 2010/12 and 2014, the very limited amount of fish survey data obtained by Observer 1 during that period, and the high variability within the relatively limited number of observations, limit the detection of change with a high level of confidence. A more detailed examination of these data, as well as of observations obtained by a

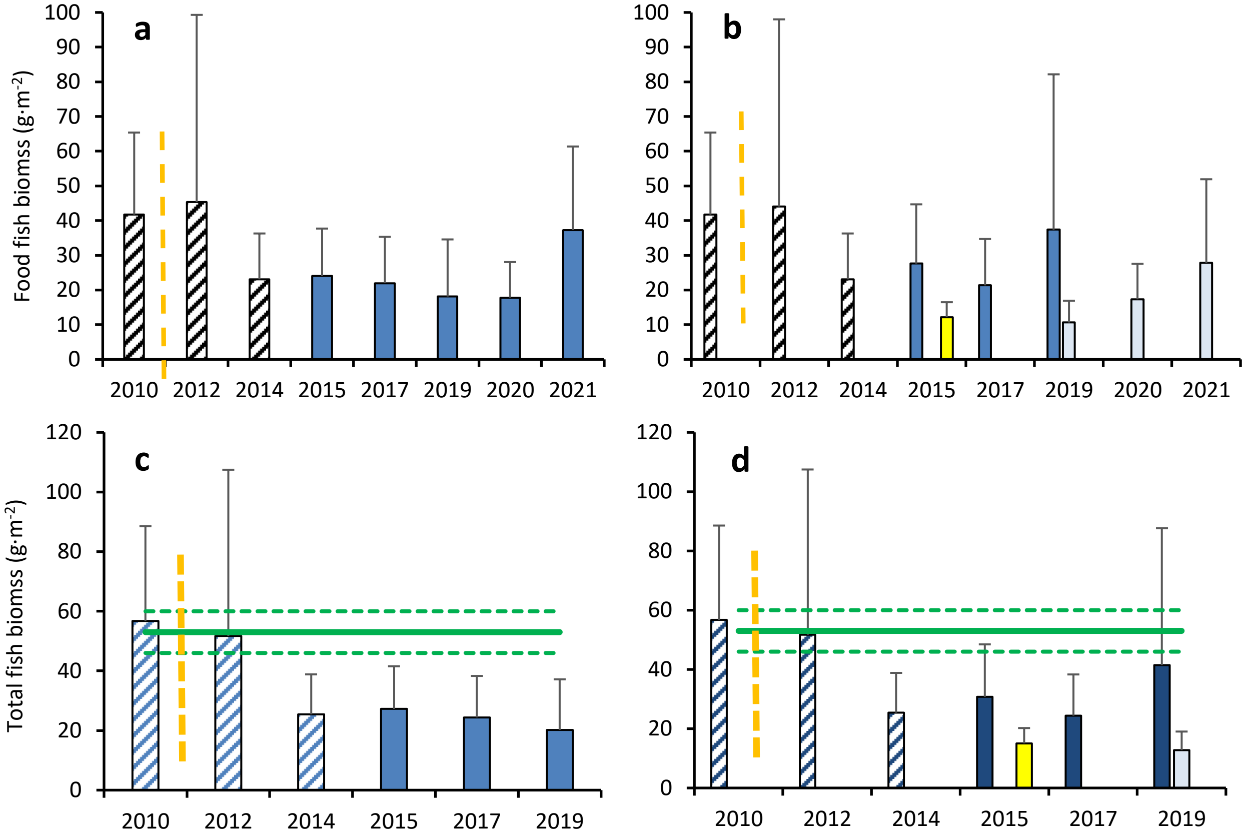


Figure 18. Food fish biomass (a, b) and total fish biomass (c, d) within the Tumon Bay HPRA between 2010 and 2021/2019, with data from all observers combined (a, c) and presented separately (b, d). Note that the time interval between survey years varied, and that the monitoring site boundary shifted substantially between 2010 and 2012 (indicated by orange dashed vertical line. Mean biomass values in 2010, 2012, and 2014 (represented by bars with the diagonal lines) should be considered cautiously, as these data include observations obtained by Observer 1 at only two, four, and five sampling stations, respectively, out of the usual 22 sampling stations. The green lines indicate the potential total reef fish biomass for an unimpaired Guam reef community (53 ± 7 g/m^2) estimated by Williams et al. (2015).

relatively inexperienced observer in 2010 that were excluded from the present analysis, may provide some useful insights into changes in food fish biomass within the Tumon Bay HPRA in the early years of monitoring. Despite the clear interobserver bias detected for data collected in 2015 and 2019, the relative stability of the robust fish survey data collected between 2015 and 2020 is likely an accurate reflection of the stability of total food fish biomass within the Tumon Bay HPRA during this period. When only the observations obtained by Observer 2 are examined, a clear increase in food fish biomass is evident between 2019 and 2021. Food fish biomass increased by ~60% (from 18 ± 17 g·m^{-2} to 29 ± 25 g·m^{-2}) during this period (Partover.test, p = 0.005). A more detailed analysis of GLTMP data and other datasets from this period may reveal possible drivers of this increase.

As with food fish biomass, total fish biomass, which includes the biomass of both food and non-food fishes, may have declined between 2012 and 2014, but the very limited amount of data collected during that period necessitates a great deal of caution in making such a comparison. Total fish biomass appeared to remain relatively stable between 2015 and 2019 when observations from all observers are combined (Fig. 18c). The apparent increase in total fish biomass evident when only the observations of Observer 1 are considered is not statistically significant (Partover.test, p = 0.5), owing to the relatively small number of surveys conducted by Observer 1 in 2019 (n = 10) and the high degree of variability in total biomass values across those sampling stations (Fig. 18d). The total fish biomass of 21 g·m^{-2} (all observers combined) and 41 g·m^{-2} (Observer 1 only) reported for the Tumon Bay HPRA in 2019 is approximately 39–77% of the potential total reef fish biomass value (53 ± 7 g·m^{-2}) estimated for an unimpaired Guam reef community by Williams et al. (2015). While Williams et al. (2015) relied on fish survey data collected using the same method used by the GLTMP prior to 2020, significant differences in geophysical conditions and the structure of benthic communities across the island necessitate caution in placing too much emphasis on comparisons of total fish biomass at a particular site to an island-wide target fish biomass value. Still, this target value provides at least some kind of ballpark figure against which these site-scale total reef fish biomass values can be compared.

Due to the significant interobserver bias discussed above, biomass values for selected food fish families within the Tumon Bay HPRA were calculated using only those observations recorded by Observer 2 between 2019 and 2021 (Fig. 19). While statistical comparisons were not carried out for this limited preliminary analysis, the biomass of all food fish families but groupers appeared to increase within the Tumon Bay HPRA during this period, although the high degree of variability in biomass values across sampling stations for some years may make it difficult to detect smaller changes.

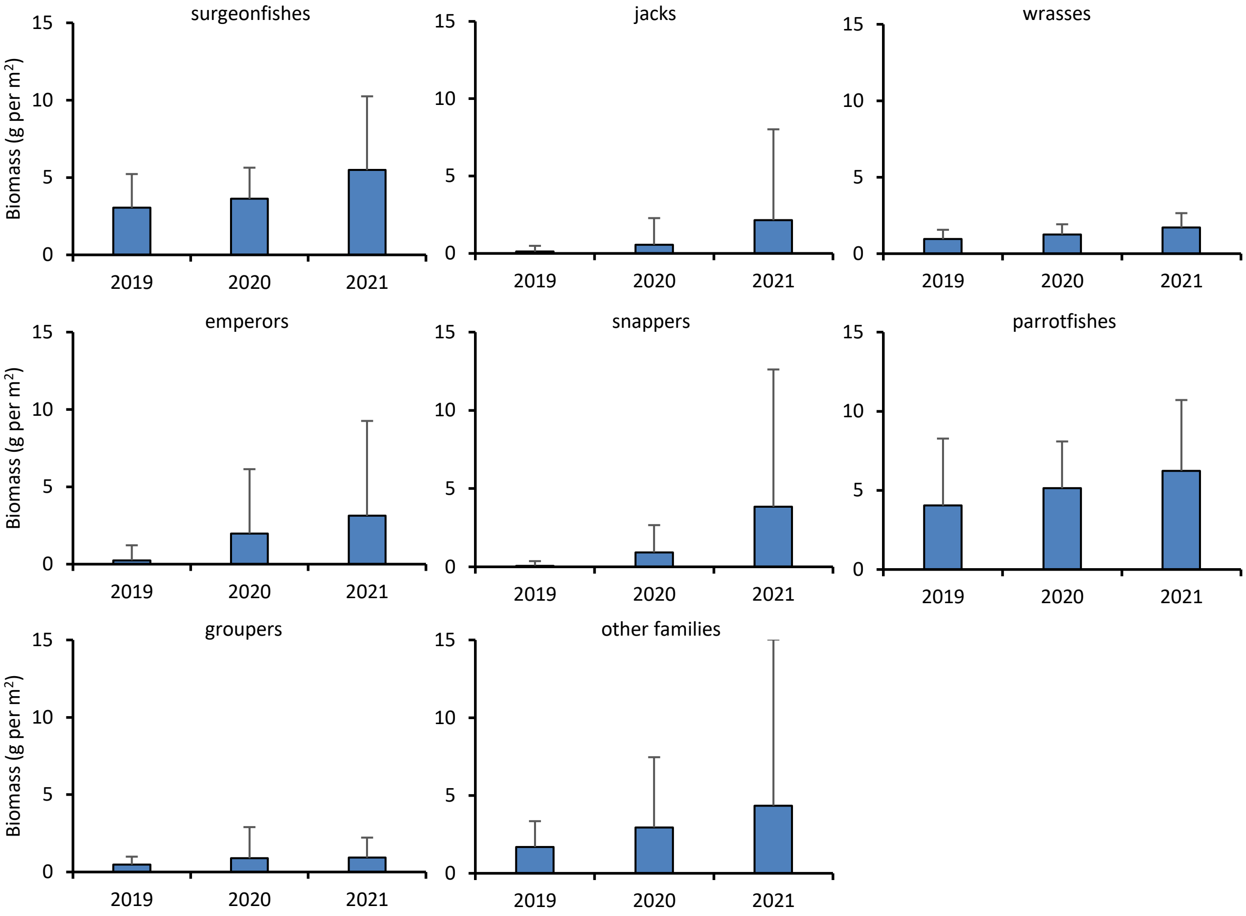


Figure 19. Biomass (g per m^2) of selected food fish families within the Tumon Bay HPRA between 2019 and 2021. Due to the significant interobserver bias detected early in the analysis, these values were calculated using only those observations collected by Observer 2, who collected data for the three most recent years.

Density

Food fish density within the Tumon Bay HPRA appears to have remained relatively stable for all three broad size classes between 2010 and 2021, with the exception of a spike in the smaller size class (≤ 20 cm) in 2017 (Fig. 20a). The apparent spike in the density of smaller food fishes in 2017 appears to have been driven by observations of large numbers of small (< 8 cm) unicornfishes recorded at two of the sampling stations, and the relatively large number of soldierfishes and small goatfishes at another. The influence of large, but infrequently observed, schools on broad metrics such as mean food fish density highlights the importance of carrying out a more detailed analysis of fish survey data to gain a better understanding of trends in individual groups. The low abundances of moderately sized (21 ≤ 40 cm) and large (> 40 cm) fish make detection of trends in these groups challenging, but the observations obtained by Observer 2 between 2019 and 2021 suggests that, along with food fish biomass, the density of moderately sized food fishes may have increased during that period (Fig. 20b).

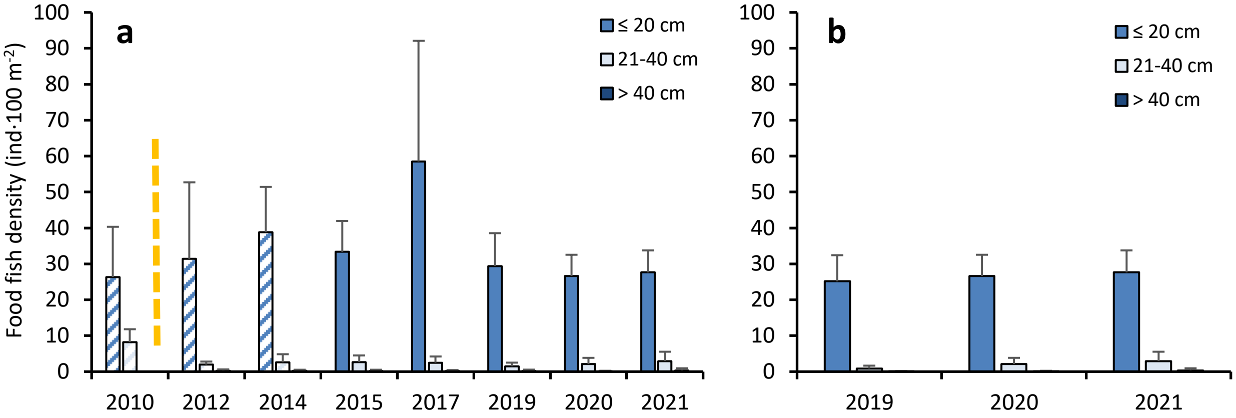


Figure 20. Food fish density (ind·100 m^{-2} ± SD) by size class within the Tumon Bay HPRA between 2010 and 2021, with data from all observers combined (a) and with only data from Observer 2 (b). Note that the time interval between survey years varied, and that the monitoring site boundary shifted substantially between 2010 and 2012 (indicated by orange dashed vertical line). Mean density values in 2010, 2012, and 2014 (represented by bars with the diagonal lines) should be considered cautiously, as these data include observations obtained by Observer 1 at only two, four, and five sampling stations, respectively, out of the usual 22 sampling stations.

Species Richness

Both the total and mean number of species recorded within the Tumon Bay HPRA fluctuated between 2010 and 2021 (Fig. 20). Total species richness ranged from a low of 130 species in 2010 (excluding 2014 due to very low sampling effort that year) to a high of 194 species in 2017, while mean species richness ranged from a low of 33 ± 10 species per sampling station in 2010 to 63 ± 10 species per sampling station in 2017. The fluctuations in total and mean species richness are likely primarily a result of interobserver bias and differing levels of sampling effort (e.g., number of sampling stations surveyed a given year), rather than a reflection of actual changes in the number of species present within the Tumon Bay HPRA. Mean species richness should be less sensitive to differing levels of effort, but comparisons across years may be limited when too few sampling stations were surveyed a given year, or if the number of recorded species varied significantly between sampling stations. When considering only those observations obtained by Observer 2 between 2019 and 2021, total species richness was relatively stable at around 130–150 species. A more detailed analysis of species counts obtained by the different observers is likely to reveal species that are consistently detected by one observer but not another, or other biases that could be minimized with targeted training and calibration. However, a deeper analysis, especially one that accounts for rugosity and benthic community structure, may also reveal that the fluctuations in the survey data may reflect actual differences in the species diversity in the Tumon Bay HPRA across space and time, and could indicate possible drivers of the fluctuations in total species richness, such as recruitment pulses or changes in benthic community structure or condition.

Macroinvertebrates

The mean density of sea cucumbers decreased significantly (by 81%) between 2010 to 2021 (Partover.test, p = <0.001), which is consistent with the preliminary findings presented by Burdick, Brown and Miller (2019) for the period between 2010 and 2018 (Figs. 22–23). This decline in sea cucumbers was largely driven by the large decline in the spiky sea cucumber, *Stichopus chloronotus,* which is one of the most commonly observed sea cucumber species along the seaward slope in Tumon Bay. The most significant drop in sea cucumber density was a 69% decline that occurred between 2012 and 2014 (Partover.test, p <0.001), with densities continuing to gradually decline through 2019. Sea cucumber densities remained low through 2021. Mean edible shell (mainly top shells and giant clams) density decreased by 63% between 2012 and 2015 (Partover.test, $p < 0.001$), and was variable (between 2.5 and 4.3 ind·100 m^{-2}) between 2015 and 2021 (Figs. 22–23). Mean density of larger sea stars (mainly *Linckia* spp.) remained relatively stable at around 1 to 2 ind·100 m^{-2} between 2010 and 2021, while the mean urchin density increased between 2012 and 2019 (Partover.test, $p < 0.01$).

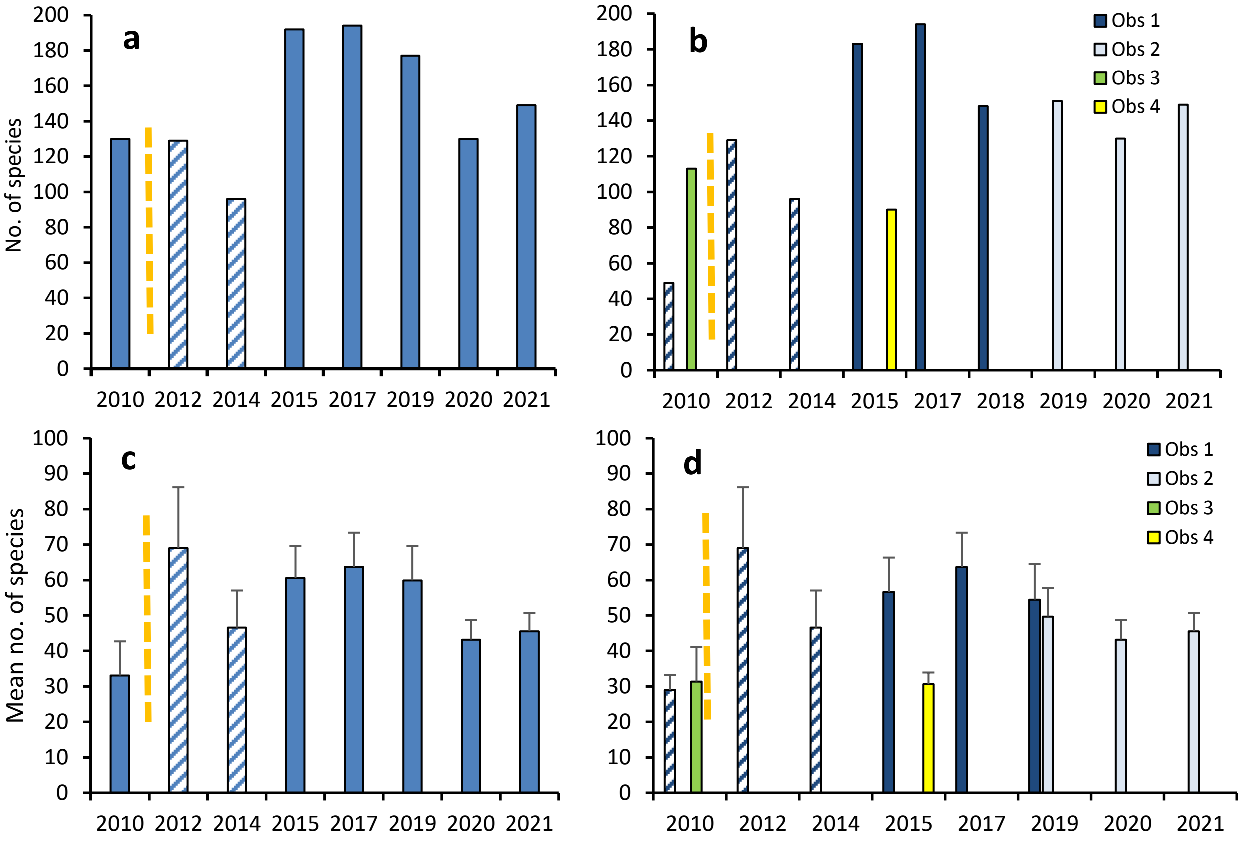


Figure 21. Total (a, b) and mean (c, d) fish species richness within the Tumon Bay HPRA between 2010 and 2021, with data from all observers combined (a, c) and presented separately (b, d). Note that the time interval between survey years varied, and that the monitoring site boundary shifted substantially between 2010 and 2012 (indicated by orange dashed vertical line). Species richness values in 2010, 2012, and 2014 (represented by bars with the diagonal lines) should be considered cautiously, as these data include observations obtained by Observer 1 at only two, four, and five sampling stations, respectively, out of the usual 22 sampling stations.

While it's not entirely clear what caused the rapid decline in *Stichopus chloronotus*, a species generally not targeted for harvesting, observations by Val Brown of lesions and dead individuals of this species in the Piti Bomb Holes Marine Preserve in June 2014 suggests that a disease may have affected this usually abundant species, and the coincidence of Ms. Brown's observations of disease in *S. chloronotus* with a thermal stress event that followed the previous year's historically severe thermal stress event, suggests that the unusually warm water temperatures may have also had a role in the apparent mass mortality of *S. chloronotus*. Other species may have been impacted, but disease was not reported for other taxa and the generally low densities of other sea cucumber taxa makes detection of changes with rapid assessments difficult.

A diseased spiky sea cucumber (Stichopus chloronotus) observed in the Piti Bomb Holes Marine Preserve in 2014. Photo provided by Marybelle Quinata.

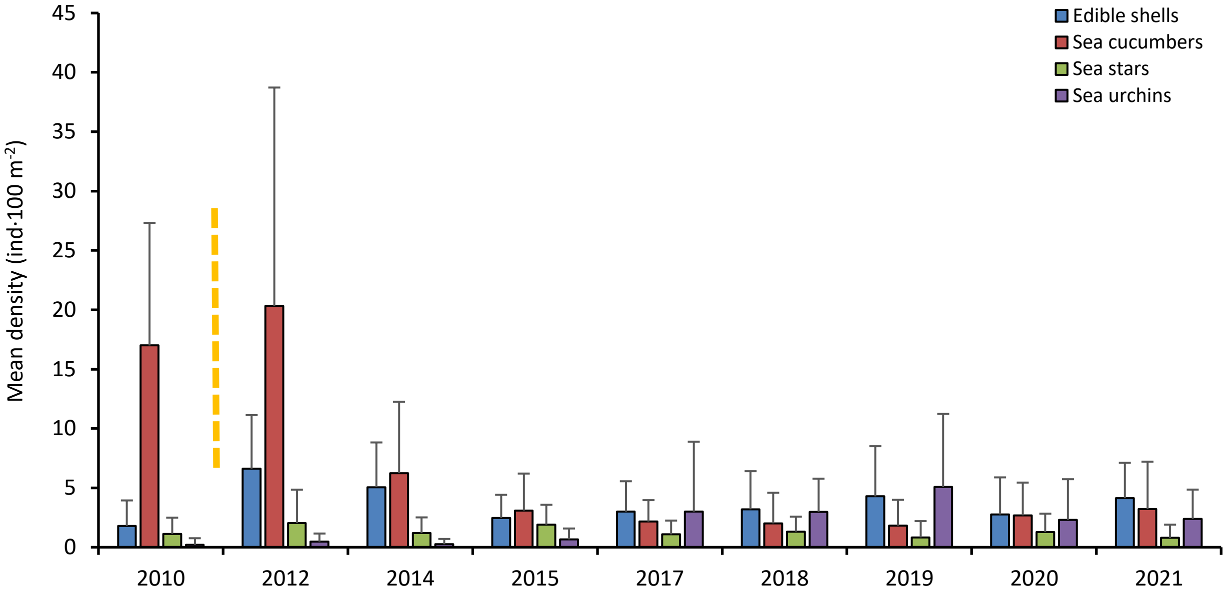


Figure 22. Mean density (ind·100 m^{-2} ± SD) for macroinvertebrate groups of interest within the Tumon Bay HPRA between 2010 and 2021. Note that the time interval between survey years varied in duration, and that the monitoring site boundary shifted substantially between 2010 and 2012 (indicated by orange dashed vertical line).

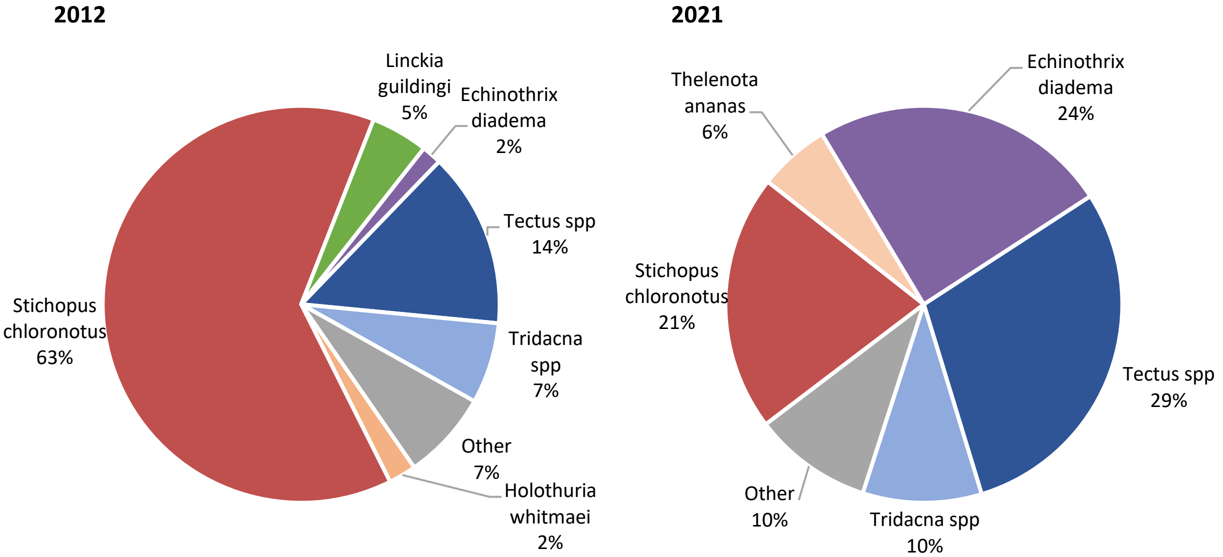


Figure 23. Relative abundance of individual macroinvertebrate taxa within the Tumon Bay HPRA in 2012 and 2021.

Discussion

The relative stability of total coral cover along the submarine terrace in Tumon Bay over the last decade, despite the impacts of multiple thermal events during this period, suggests that the coral community along this part of the reef is relatively robust to the levels of stress experienced during these events. This apparent resilience is likely a result of the dominance of the generally stress-tolerant *Porites* corals. The cover of other coral taxa, especially stress-susceptible taxa such as *Acropora* spp. and *Montipora* spp., was already very low prior to onset of the 2013 bleaching event. Historical data is limited for Tumon Bay and nearby areas, but a qualitative assessment of the coral community along the submarine terrace in Tumon Bay carried out by Richard Randall in the late 1960s suggests that there were more stress-susceptible taxa present within this part of the reef prior to the first recorded COTS outbreak that occurred shorty after the surveys (Randall 1971). Subsequent COTS outbreaks and chronically elevated COTS densities have likely suppressed populations of corals that are the preferred prey species of COTS, and limited recruitment, water quality impacts and reduced levels of herbivory may have further inhibited recovery of these species. While COTS densities have remained relatively low within the Tumon HPRA since surveys began in 2010, their continued impact within the HPRA has been evident by the numerous observations of recently killed colonies of preferred prey, such as *Acropora* spp. and *Montipora* spp. It is likely that even low COTS densities can hinder recovery of these coral taxa through selective predation. The stability of coral cover and the overall resilience of the coral community may not persist as sea surface temperatures continue to increase, and the frequency and severity of coral bleaching events increases in response. The almost total dominance of the coral community along the submarine terrace in Tumon Bay by *Porites rus* and a small number of massive *Porites* species may make it more vulnerable to devastation by diseases, the virulence of which may be enhanced by warming ocean temperatures.

The stability of coral communities within this part of the reef is in stark contrast to the bleaching- and disease-associated decline in the health of coral communities observed on the extensive shallow reef flat platform within the bay. Coral communities on the reef flat platform are comprised of different sets of dominant coral species that overlap little with the set of species that dominate the submarine terrace zone within the HPRA. The staghorn coral mortality assessment in 2015 estimated mortality rates at 40–70% for major staghorn *Acropora* thickets across the bay's reef flat platform as a cumulative result of thermal stress events in 2013 and 2014 (Raymundo et al. 2017), while more recent staghorn mortality assessments in 2017 and 2020/2021 found no remaining *A. muricata* in the bay, and expansive dead areas of large thickets (mainly *A.* cf. *pulchra*) have failed to recover. As reported above, coral cover at the shallow *Acropora* cf. *pulchra*- and *A. acuminata*-dominated coral community surveyed regularly by the RFMP in Tumon Bay declined by ~11% between 2009 and 2022. However, a notable 87% increase in coral cover was observed at the site between the end of 2017 and 2022, a period with limited heat stress. Such a rapid increase in coral cover suggests that recovery can occur quickly at this and similar reef flat coral communities, particularly those hosting the fast-growing, generally resilient staghorn species, *Acropora* cf. *pulchra*, when conditions are favorable. But the projected increase in the severity and frequency of heat stress events, the increasing reliance on a smaller number of coral species, combined with the potential increase in disease activity associated with ocean warming and the array of other anthropogenic are expected to challenge even the most resilient reef systems on Guam and around the world.

Coral cover at a single permanently established island-wide bleaching response and recovery survey site located within the reef front zone adjacent to the Tumon Bay HPRA was similar in 2013 (16%) and 2021 (18%). Although statistical comparisons have not yet been made for individual island-wide survey sites, coral cover appears to have been little affected by the 2013 bleaching event, remaining at around 16% between 2013 and 2015, but may have been impacted by the more severe 2017 event, with coral cover dropping from a peak of 21% near the

The coral community along the 5 m depth contour surveyed in Tumon Bay as part of the island-wide bleaching monitoring effort is primarily comprised of relatively bleaching resistant coral species, such as Goniastrea retiformis and Leptoria phrygia, as seen in the image taken at the site in 2021 (left). However, these surveys did not capture significant losses of the more bleaching-susceptible species, such as several Acropora and Pocillopora species, that were much more abundant at shallower (< 4 m) depths in the calmer waters of the western leeward coast, as seen in the image at the right, taken at a nearby reef front area in 2010.

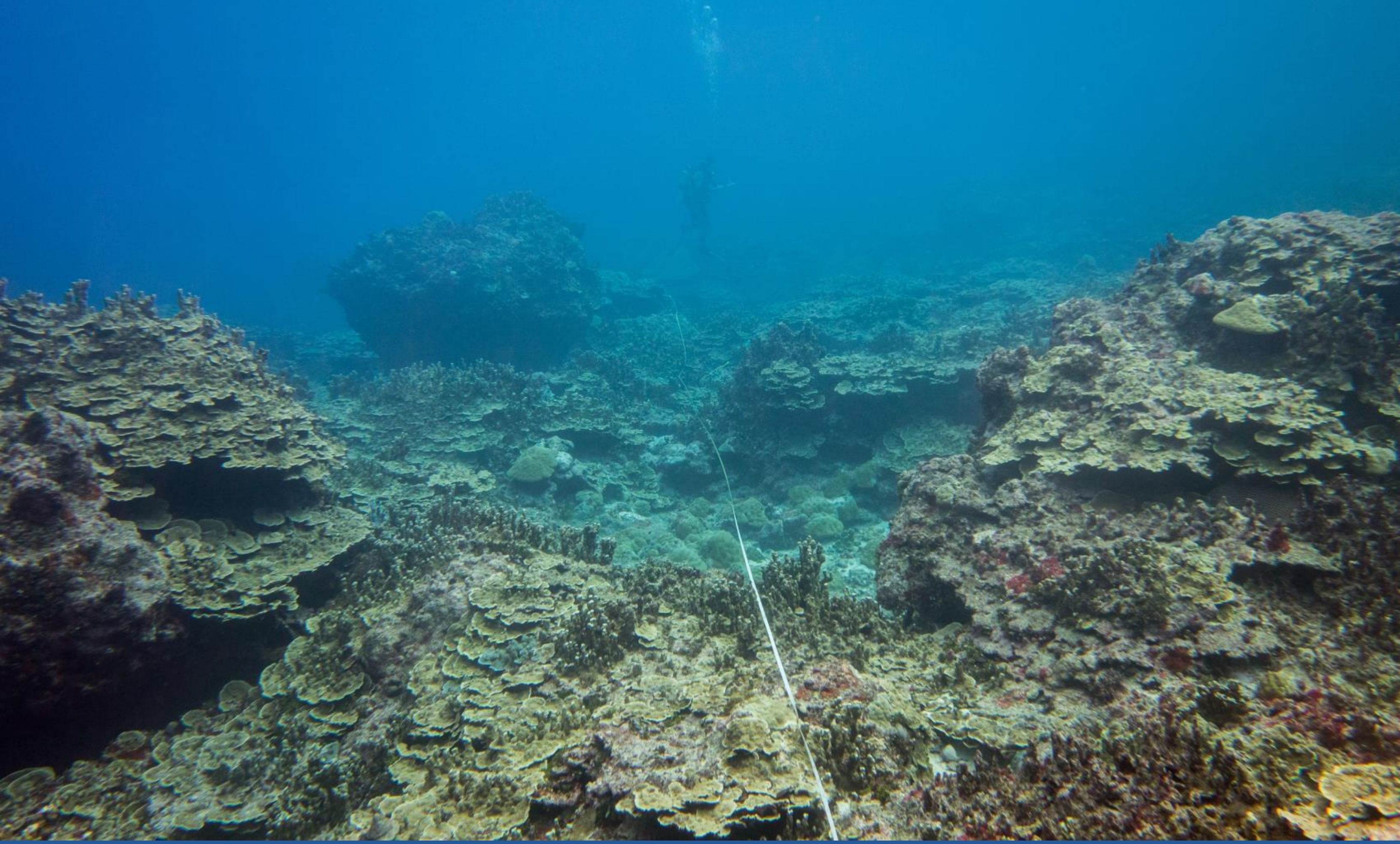
beginning of the 2017 event to 18% in 2021. The relative stability in coral cover at the reef front survey site in Tumon Bay is consistent with the stability observed across several other reef front survey sites located along the west coast, and of coral cover in the submarine terrace zone of Tumon Bay, in recent years. This stability is in contrast to major losses of live coral cover at other reef front sites around the island between 2013 and 2017, estimated at 34% island-wide and 59% for the eastern windward reef front sites (as discussed on p. 26 of this report).

The reef front zone in Tumon Bay, as with the submarine terrace and other reef areas along the west coast, generally hosted lower cover of stress-susceptible coral taxa than reefs along the eastern windward coast prior to the onset of severe coral bleaching in 2013, likely lessening the overall impact of heat stress events on these communities. The dominance of stress-tolerant taxa along the west coast may be due in part to historical impacts of predation by COTS, which preferentially prey on many of the same coral taxa that are highly susceptible to heat stress. COTS can more regularly feed at shallower depths on the west coast due to the lower wave energy on this side of the island, while the zone of impact by COTS on the eastern windward coast typically begins at a greater depth. However, further analysis of historical data is required to better understand patterns and drivers of historical changes in the reef communities on the island’s west coast.

While many seaward slope communities along the west coast are dominated by stress-tolerant coral taxa, it is important to note that through at least 2013 stress-susceptible coral taxa were abundant along the west coast at depths shallower than the 5 m depth contour targeted at the island-wide survey sites, and thus the impacts of multiple heat stress events on these shallower communities have not been quantitively assessed. Anecdotal observations made by the MPC along the reef front in Tumon Bay and other areas along the west coast between 2004 and the present suggest there were catastrophic losses of *Acropora* spp., some *Pocillopora* spp. and other coral taxa in these shallower (< 5 m) reef front coral communities as a result of the 2013–2017 bleaching events (see images on p. 58 and p. 71 for examples).

The data and staff capacity limitations mentioned above preclude confident statements about the status of reef fish populations within the Tumon Bay HPRA at this time. However, there are some indications that food fish biomass may have decreased along the submarine terrace after 2012, and that the density of small food fishes may have declined between 2015 and 2019. The reef fish survey data obtained by a single observer since 2019 allows for more confidence in the assertion that total food fish biomass, the biomass of all food fish families (except groupers), and the density of small fishes increased steadily between 2019 and 2021. Total fish biomass in the Tumon Bay HPRA in 2019 was highest among all of the HPRAs, but was still between 39% and 77% of the potential total reef biomass estimated for an impaired Guam reef by Williams et al. (2015). Further study of the reef fish data obtained by the GLTMP from the Tumon Bay HPRA, possibly in combination with other datasets may better resolve any trends in the reef fish community that may have occurred within the Tumon Bay HPRA and other HPRAs over the last decade. Further investigation may also provide an indication of the likely causes of any declines (e.g., poaching or habitat degradation) or improvements (e.g., fisheries policy or improvement in habitat condition) in reef fish community condition, and may facilitate robust comparisons to reef fish communities within the East Agana HPRA, which was intended to serve as a control site against which the effectiveness of the Tumon Bay HPRA could be evaluated..

The precipitous decline in sea cucumber densities within the Tumon Bay HPRA between 2012 and 2014, which was comprised primarily by the decline in the spiky sea cucumber (*Stichopus chloronotus*), was a notable event that demands additional attention. The possible link between the apparent mass mortality of this sea cucumber species and ocean warming should be investigated further, and evidence for possible impacts to other reef invertebrates should be gathered. Ocean warming is projected to increase over the coming decades and is expected to increase the virulence and prevalence of marine disease, potentially resulting in the mass mortality of sea cucumbers and other marine organisms that may be important sources of protein for coastal communities, or which may play key roles in maintaining coral reef function. The spiky sea cucumber is not typically harvested (B. Tibbatts, personal communication), so its decline is not likely to directly impact people on Guam dependent on sea cucumbers for part of their diet or income. It is not entirely clear what the impact of the decline in spiky sea cucumber means for the benthic community or for reef function more broadly, but given the large proportion that this one species contributes to total sea cucumber densities along the bay’s submarine terrace, nutrient recycling and the control of thin cyanobacterial growth over the substrate may be compromised, possibly resulting in lower productivity and increased impacts to reef calcifiers and habitat-providers (e.g., corals and coralline algae) by cyanobacteria. It’s also unclear why recovery in *Stichopus chloronotus* hasn’t yet been observed in the Tumon HPRA (or in the East Agana HPRA, see p. 68), or is occurring very slowly, especially when recovery of the species was evident in the Piti HPRA.



EAST AGANA BAY

Key findings

- Coral cover was high and remained relatively stable, at around 45%, between 2010 and 2020, despite the multiple bleaching events that affected Guam's reefs during this period
- The stability in coral cover is likely attributed to the dominance of a single bleaching-resistant coral species, *Porites rus*, but the very low diversity coral community may be vulnerable to disease outbreaks, the aggressive encrusting sponge *Terpios*, and more severe warming events in the future
- Data and staff capacity limitations currently prevent detailed analyses of reef fish survey data obtained between 2010 and 2021, but there are some indications that food fish biomass may have decreased after 2010 and increased steadily after 2019
- The biomass of jacks and "other families" may have increased between 2019 and 2021, while all other food fish families remained relatively stable
- Total fish biomass in 2019 was relatively low (12 g/m^2), at 23% of the potential total reef fish biomass estimated for an unimpaired Guam reef by Williams et al. (2015)
- The density of small food fishes may have declined between 2015 and 2020, but likely increased between 2020 and 2021
- Significant declines in the densities of sea cucumbers and edible shells (mainly top shells) between 2014 and 2016 is likely directly or indirectly related to heat stress; sea cucumber densities remained low through 2021, while edible shells increased steadily during this period

***Above**: High coral cover and high rugosity at a Porites rus-dominated reef along the submarine terrace in the East Agana Bay monitoring site in 2020.*

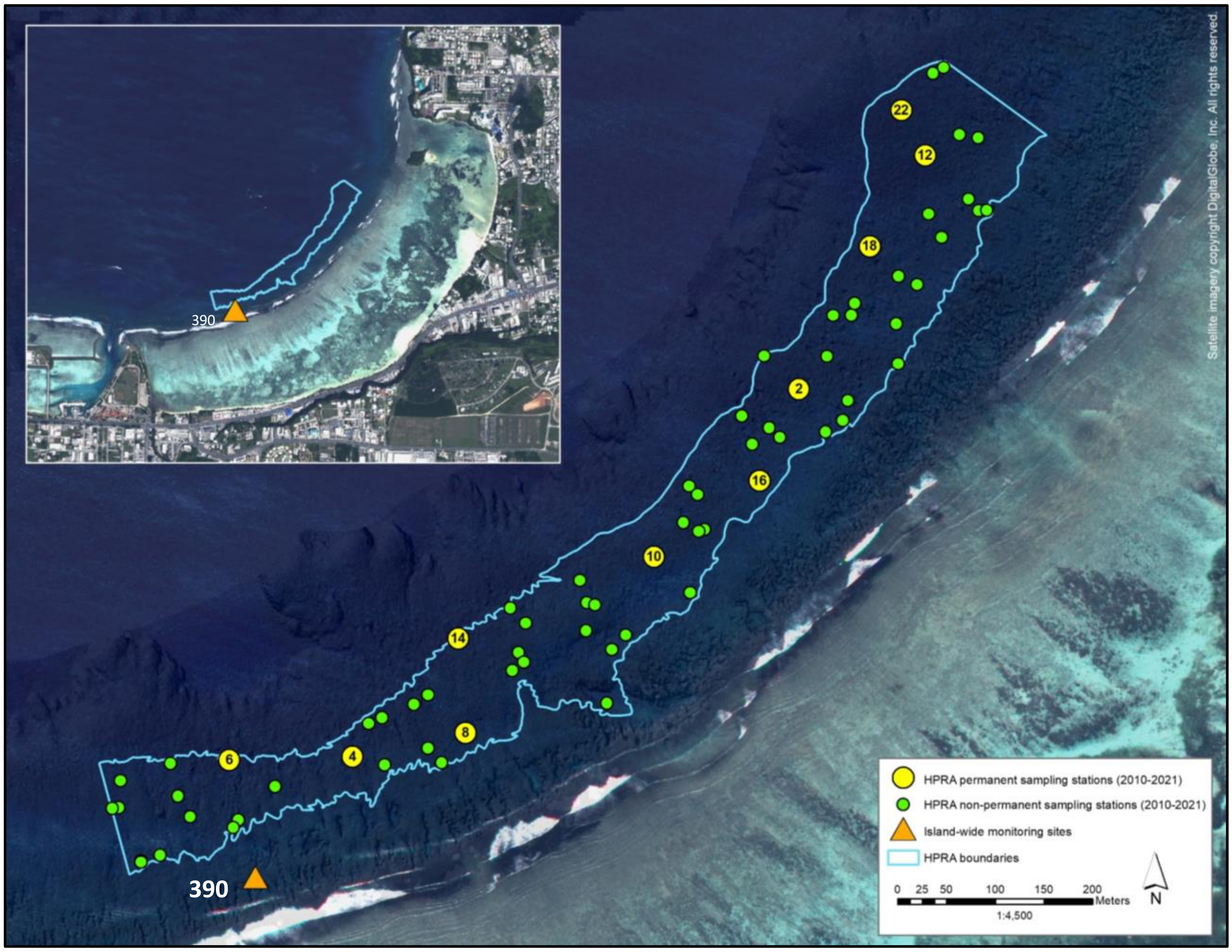


***Figure 24**.* Map of the East Agana Bay HPRA site boundaries and sampling stations, as well as the location of one of the island-wide bleaching response and recovery sites within the vicinity of the HPRA.

Site overview

East Agana Bay was the second HPRA to be established for long-term monitoring, with the completion of baseline data collection in 2010. East Agana Bay, which does not host a locally managed Marine Preserve, was chosen as a comparison site to Tumon Bay, thus monitoring in East Agana occurs along the reef slope terrace within the same depth range as that targeted within the Tumon Bay site (Fig. 24). The reef structure in each of the bays is generally similar and both bays are impacted by non-point source pollution, so the intent was that the pairing of the bays will allow an examination of the relative effects of protection status on the reef communities in the bays. A direct comparison of reef ecosystem condition between the Tumon and East Agana HPRAs will not be addressed here, as a more comprehensive analysis that accounts for differences in geology, wave exposure, productivity and other factors is required before such comparisons can be made.

As with the Tumon HPRA, and other reef tracts around the island, the benthic community prevalent within the seaward slope zone in East Agana Bay is distinct from that observed in the shallower, more wave exposed reef margin and reef front zones, and is dramatically different from the various benthic communities that occur across the shallow reef flat within the bay. A single permanently established island-wide bleaching response and recovery site (390) is located along the reef front just upslope of the western side of the HPRA. East Agana Bay does not host an RFMP site.

Baseline data collection occurred within the East Agana HPRA in 2010, with subsequent data collection efforts occurring in 2012, 2014, and 2016–2022. Staffing limitations and the re-allocation of resources required to respond to the multiple bleaching events between 2013 and 2017 prevented the collection of data in the East Agana site in

2013 and limited the collection of data for the other sampling years (Appendix A). Benthic cover and macroinvertebrate abundance data was also collected at two shallow (5 m depth) sites along the reef front in East Agana Bay, including one close to the HPRA site boundaries, in 2013 as part of a UOGML-led bleaching response effort. The reef front site (390) located close to the HPRA was re-visited in 2015 as part of an island-wide bleaching recovery assessment, in 2016 as part of an island-wide reef resilience assessment, in 2017 as part of the response to another major bleaching event and was surveyed and marked with rebar in 2021.

The results of analyses of available benthic cover, reef fish, and macroinvertebrate data obtained within the East Agana HPRA between 2010 and 2021 are provided below. Data obtained at the island-wide bleaching response and recovery site, and at staghorn locations within the bay are presented in other sections of this report (pages 26 and 36, respectively), but are referenced in the "Discussion" section below.

Benthic cover

Mean cover of living hard coral was 45% when surveys began in the East Agana Bay HPRA in 2010 and 43% in 2020, the last year for which data is available (Fig. 25). While the 2010 and 2020 cover values were statistically indistinguishable, some notable increases and decreases were detected at various points throughout the decade. For example, within the two years between the 2010 and 2012 survey efforts total coral cover in the East Agana HPRA increased from 45% to 56% (Partover.test, p = 0.02) but then declined to 47% in 2016 (Partover.test, p = 0.005), potentially a result of the 2013 and 2014 bleaching events. In contrast, the greatest increase in coral cover was observed between 2017 and 2018, when cover increased from 41% to 53% (Partover.test, p = 0.001), despite the historically severe 2017 coral bleaching event, but then fell to 43% in 2020 (Partover.test, p = 0.005).

Porites rus comprised greater than 80% of total cover in the East Agana Bay HPRA across all survey years between 2010 and 2020, reaching as high as 89% of total coral cover in 2012 (Figs. 26–27). Mounding *Porites* spp. comprised between 8% and 15% of coral cover during this period, while the contribution to total coral cover by other *Porites* taxa and other coral genera contributed less than 3%. As expected, trends in the cover of *Porites rus* mirrored that of total coral cover between 2010 and 2020. *Porites rus* cover was 36% in 2010 and 35% in 2020 (Partover.test, p = 0.91), but significant fluctuations were detected at various points during that 10-year period. For example, the cover of *P. rus* rapidly increased from 36% to 50% between 2010 and 2012 (Partover.test, p = 0.02) before decreasing

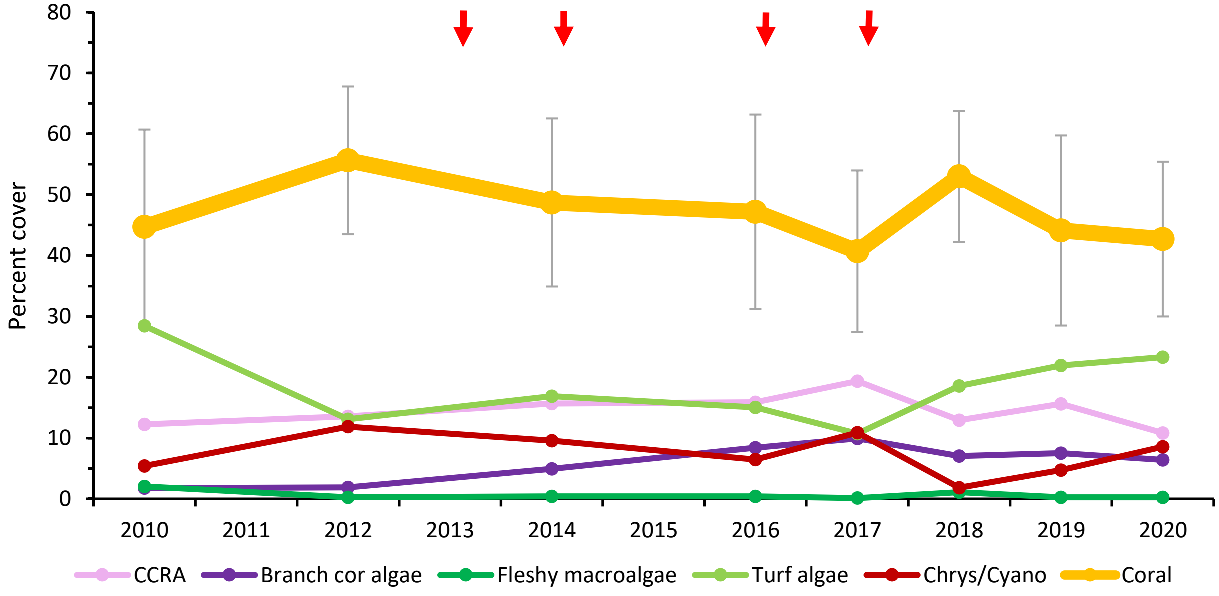


Figure 25. Mean (± SD) percent cover of major benthic cover classes for the East Agana Bay monitoring site between 2010 and 2020. Red arrows indicate the timing of significant coral bleaching events, including the historically severe 2013 and 2017 events. Note that survey years are not presented in regular annual increments, and thus the elapsed period of time between each pair of neighboring data varies from one to two years.

to 42% in 2014 (Partover.test, p = 0.03). The decrease could potentially be a result of impacts associated with the 2013 and 2014 bleaching events, both of which occurred between the 2012 and 2014 data collection efforts. However, *P. rus* cover further declined to 34% in 2017 (Partover.test, p = 0.04), prior to the 2017 bleaching event, but then increased to 46% in 2018, after the 2017 bleaching event and fell to 35% in 2020 (Partover.test, p = 0.007) despite the lack of significant thermal stress events occurring since 2017. Increases in the cover of the aggressive encrusting sponge, *Terpios hoshinota*, appear to be generally associated with declines in *P. rus* cover, suggesting the sponge may have played at least some role in these decreases. The cover of mounding *Porites* spp. and the cover of non-*Porites* coral taxa each remained between 5% and 7%, with no statistically significant changes detected at various intervals during the study period.

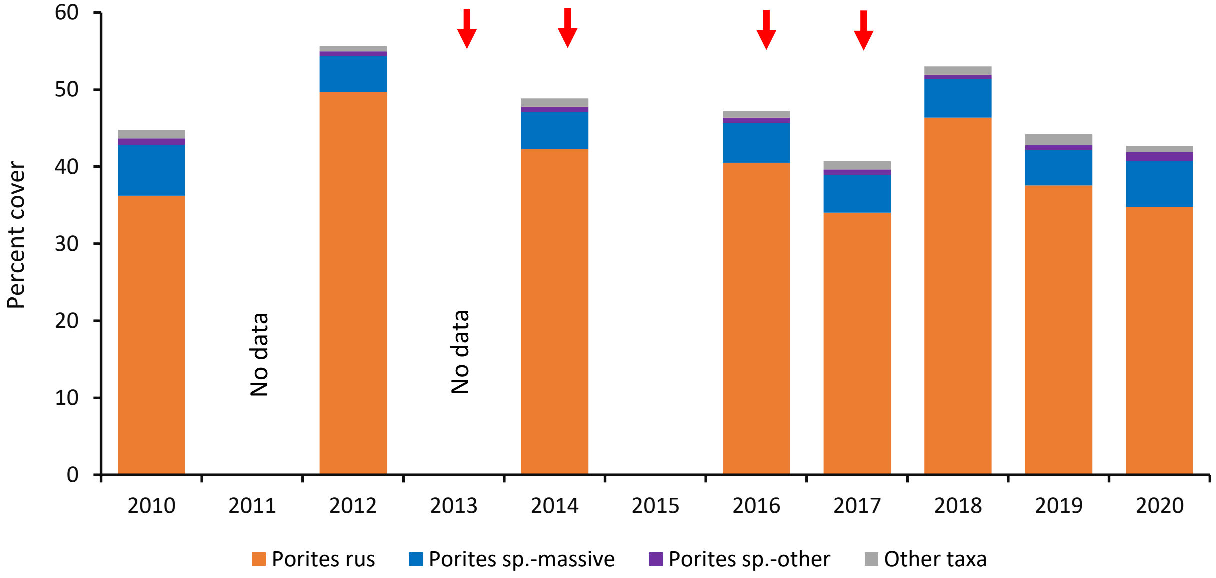


Figure 26. Percent cover of coral taxa for the East Agana Bay HPRA between 2010 and 2020. Red arrows indicate the timing of significant coral bleaching events, including the historically severe 2013 and 2017 events. Note that the elapsed period between each pair of neighboring data varies from one to two years.

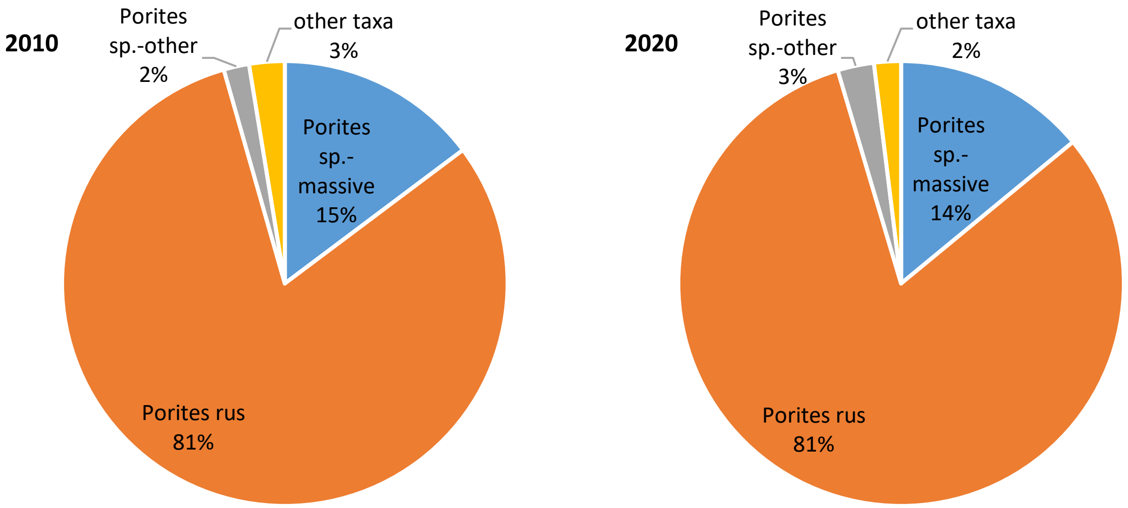


Figure 27. Percent contribution of individual coral taxa/taxa groupings to mean total coral cover for the East Agana Bay HPRA in 2010 and 2020.

Statistical comparisons were not made for non-coral benthic features through time, but as with the Tumon Bay HPRA the cover of fleshy macroalgae—generally considered competitors with corals for space—was very low within the East Agana HPRA throughout this period, with the highest cover (2%) reported in 2020 and cover at <1% for all other years (Fig. 25). The cover of branching coralline algae, comprised mainly of the macrophyte *Halimeda* spp., appeared to increase from 2% in 2010 to 6% in 2020, peaking at 10% in 2017. The cover of crustose coralline red algae in 2010 and 2020 was similar (12% and 9%, respectively), but peaked at 19% in 2017. The cover of turf algae was also similar in 2010 and 2020 (28% and 23%, respectively), but reached a low in 2017 (11%) and was highest in 2010 (28%). The cover of cyanobacteria and chrysophytes, which are generally considered more ephemeral than macrophytes, was also similar in 2010 and 2020 (5% and 9%, respectively), with a low of 2% in 2018 and a high of 12% observed in 2020.

Associated biological communities

Reef fishes

Biomass

Mean Food fish biomass within the East Agana Bay HPRA appeared to increase by 25% (from 12 g·m^{-2} to 15 g·m^{-2}, Partover.test, p = 0.02) between 2015 and 2021 (Fig. 28a). However, as discussed in regard to the apparent changes in food fish biomass in the Tumon Bay HPRA, interobserver bias must be considered when interpreting trends using data collected by the two different observers. In this case, the generally higher biomass values reported by Observer

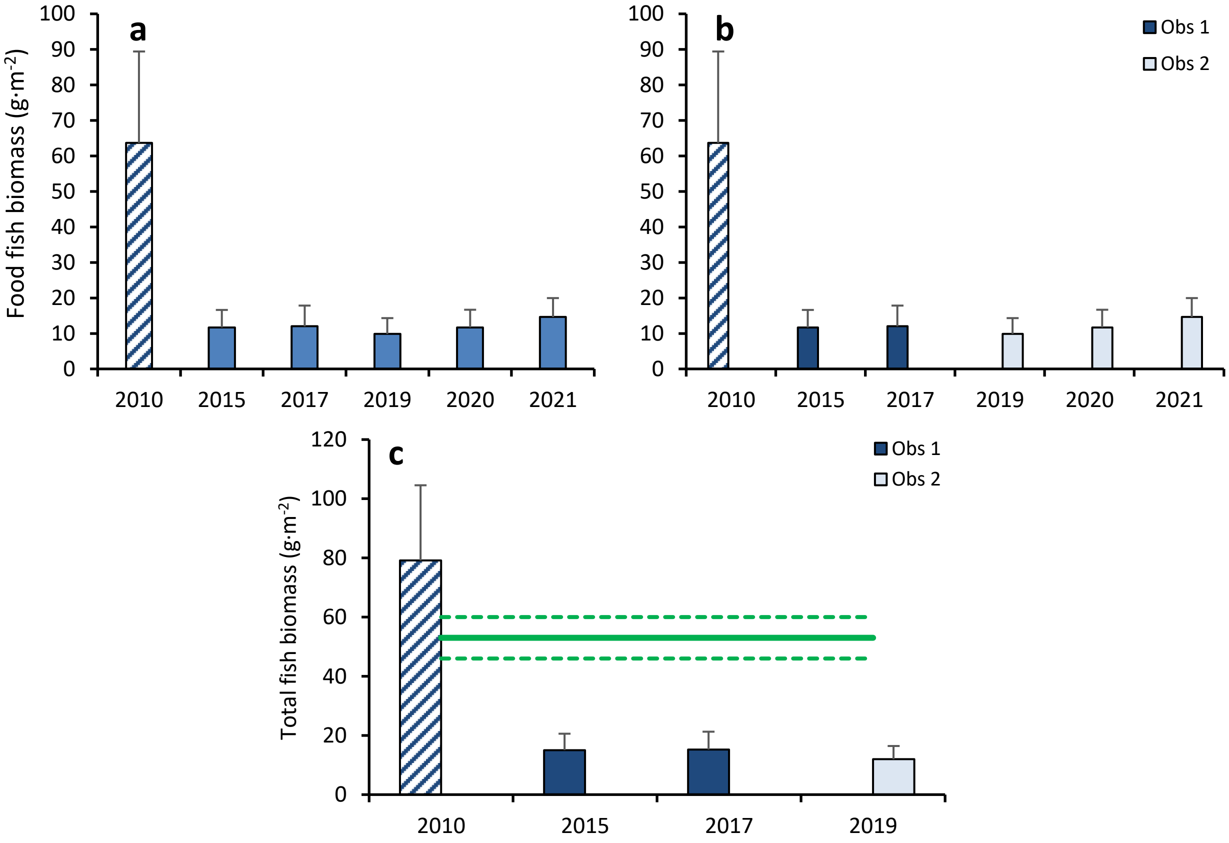


Figure 28. Food fish biomass (a, b) and total fish biomass (c) within the East Agana Bay HPRA between 2010 and 2021/2019, with data from all observers combined (a) and presented separately (b, c). Note that the time interval between survey years varied. Mean biomass values for 2010 (represented by bars with the diagonal lines) should be considered cautiously, as these data include observations obtained by Observer 1 at only three out of the usual 20 sampling stations. Data collected by Observer 4 at the 17 other sampling stations in 2010 were excluded from the analysis for reasons discussed previously. The green lines indicate the potential total reef fish biomass for an unimpaired Guam reef community (53 ± 7 g/m^2) estimated by Williams et al. (2015).

1, who carried out most surveys prior to 2019, in comparison to Observer 2, who carried out all surveys during and after 2019, may mean that the increase reported between 2015 and 2021 may be an underestimate of the actual increase in food fish biomass within the East Agana HPRA during this period (Fig. 28b). The extraordinarily high mean food fish biomass value reported for 2010 should be considered only with a high degree of caution, as it relies on survey data from only three out of 20 sampling stations and is included here for comparison to values provided in previous GLTMP reports. While it is unlikely that food fish biomass within the East Agana HPRA declined so precipitously after 2010, the recording by Observer 1 of high food fish biomass values at all three of the surveyed stations suggests that food fish biomass may have declined to some degree during this period. In fact, the food fish biomass values recorded for those stations represent the three highest values for any sampling stations surveyed within the East Agana HPRA between 2010 and 2021.

As with food fish biomass, total fish biomass, which includes the biomass of both food and non-food fishes, likely declined within the East Agana Bay HPRA between 2010 and 2015, but the very limited amount of data collected in 2010 necessitates a great deal of caution in making such a comparison (Fig. 28c). Total fish biomass may have declined by around 20% (Partover.test, $p = 0.05$) between 2015 and 2019, a period during which much more robust data collection occurred, but, again, it is not clear what role interobserver biomass may have played in this apparent decline. Due to the modification of the survey method to generate count and length data for only food fishes beginning in 2020, total fish biomass cannot be reported beyond 2019. The total fish biomass values of ~12–15 $g \cdot m^{-2}$ reported for the East Agana Bay HPRA between 2015 and 2019 are quite low (23–28%) in comparison to

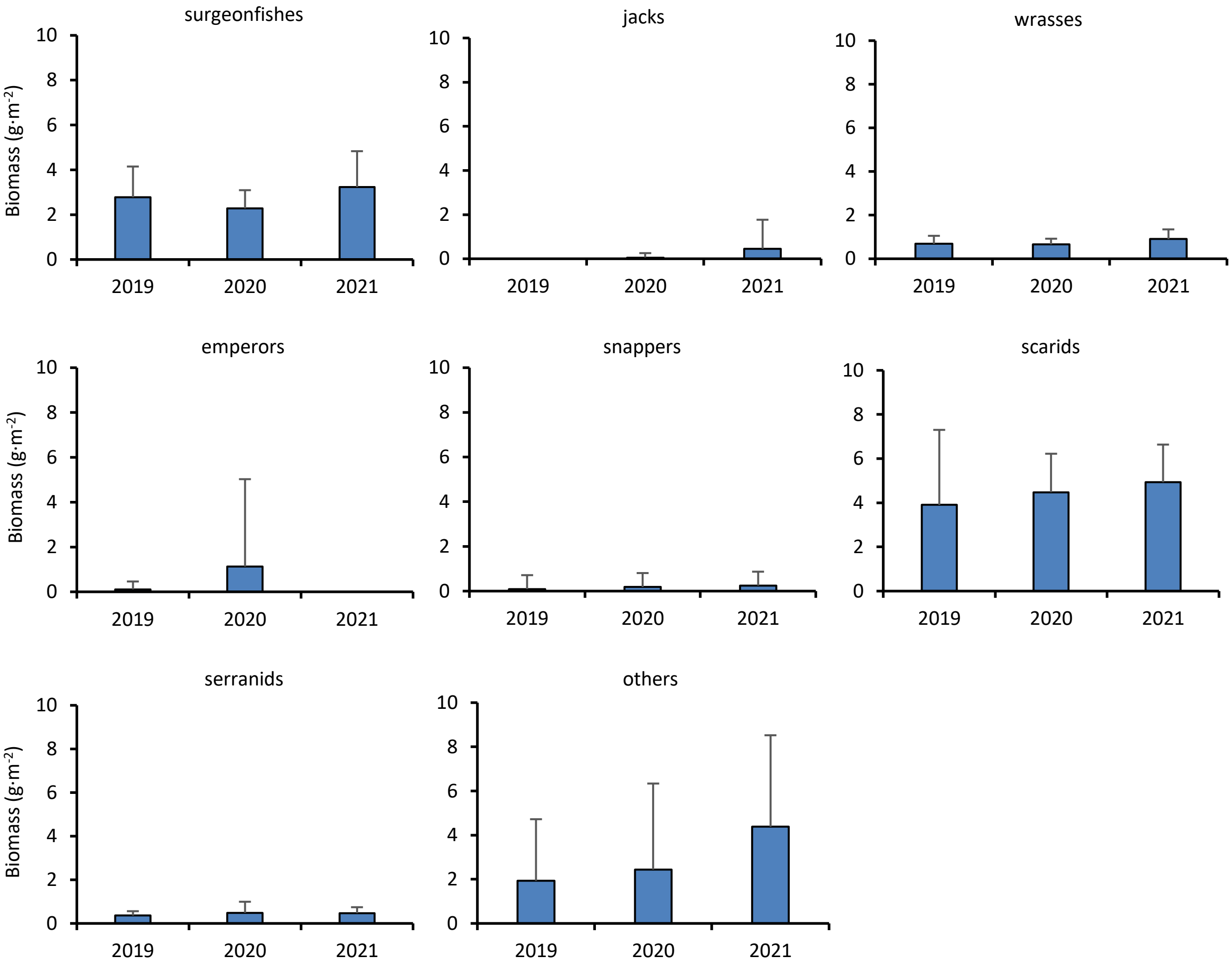


Figure 29. Biomass ($g \cdot m^{-2}$) of selected food fish families within the East Agana HPRA between 2019 and 2021. Due to the significant interobserver bias detected early in the analysis, these values were calculated using only those observations collected by Observer 2, who collected data for the three most recent years.

the target value (53 ± 7 g·m^{-2}) for an unimpaired Guam reef system estimated by Williams et al. (2015), notwithstanding the caveats mentioned earlier in regards to making such a comparison.

Due to the significant interobserver bias discussed above, biomass values for selected food fish families within the East Agana Bay HPRA were calculated using only those observations recorded by Observer 2, who carried out all fish surveys between 2019 and 2021 (Fig. 29). While statistical comparisons were not carried out for this limited preliminary analysis, the biomass of most food fish families appeared to remain relatively stable during this period, with the possible exception of an increase in jacks and "other families," which includes fusiliers, triggerfishes, goatfishes, rudderfishes, and various other food fish families.

Density

The density of small (≤ 20 cm) food fishes within the East Agana Bay HPRA may have declined between 2015 and 2020, before increasing significantly in 2021 (Fig. 30a). While the apparent decline in the density of small food fishes between 2015 and 2020 could potentially be attributed, at least in part, to interobserver bias, the slight decrease between 2019 and 2020, and the increase between 2020 and 2021 likely reflects actual changes in density, as all surveys were conducted by a single observer during this period (Fig. 30b). The low densities of moderately sized (21 ≤ 40 cm) and large (> 40 cm) fish make detection of trends in these groups challenging, but a more detailed analysis may be able to detect trends in the density of these larger size classes.

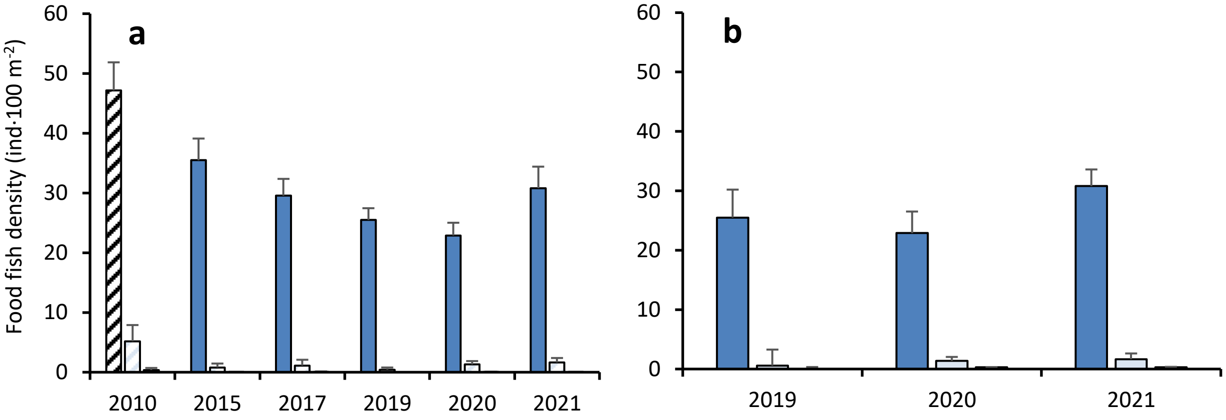


Figure 30. Food fish density (ind·100 m^{-2} ± SD) by size class within the East Agana Bay HPRA between 2010 and 2021, with data from all observers combined (a) and with only data from Observer 2 (b). Note that the time interval between survey years varied. Mean density values in 2010 (represented by bars with the diagonal lines) should be considered cautiously, as these data include observations obtained by Observer 1 at only three out of the usual 20 sampling stations. Data collected by Observer 4 at the 17 other sampling stations in 2010 were excluded from the analysis for reasons discussed previously.

Species Richness

Both the total and mean number of species recorded within the East Agana Bay HPRA appeared to fluctuate between 2010 and 2021 (Fig. 31). Total species richness ranged from a low of 122 species in 2020 (excluding 2010 due to very low sampling effort that year) to a high of 168 species in 2017, while mean species richness ranged from a low of 39 ± 5 species per sampling station in 2020 to 61 ± 10 species per sampling station in 2017. The fluctuations in total and mean species richness are likely primarily a result of interobserver bias and differing levels of sampling effort (e.g., number of sampling stations surveyed a given year), rather than a reflection of actual changes in the number of species present within the Tumon Bay HPRA. When considering only those observations obtained by Observer 2 between 2019 and 2021, total species richness was relatively stable at between 113 and 129 species. A more detailed analysis of species counts obtained by the different observers is likely to reveal species that are consistently detected by one observer but not another, or other biases that could be minimized with targeted training and calibration. Additionally, a deeper analysis that accounts for rugosity and benthic community structure may also reveal that the fluctuations in the survey data may reflect real-world change in the species diversity in the East Agana Bay HPRA.

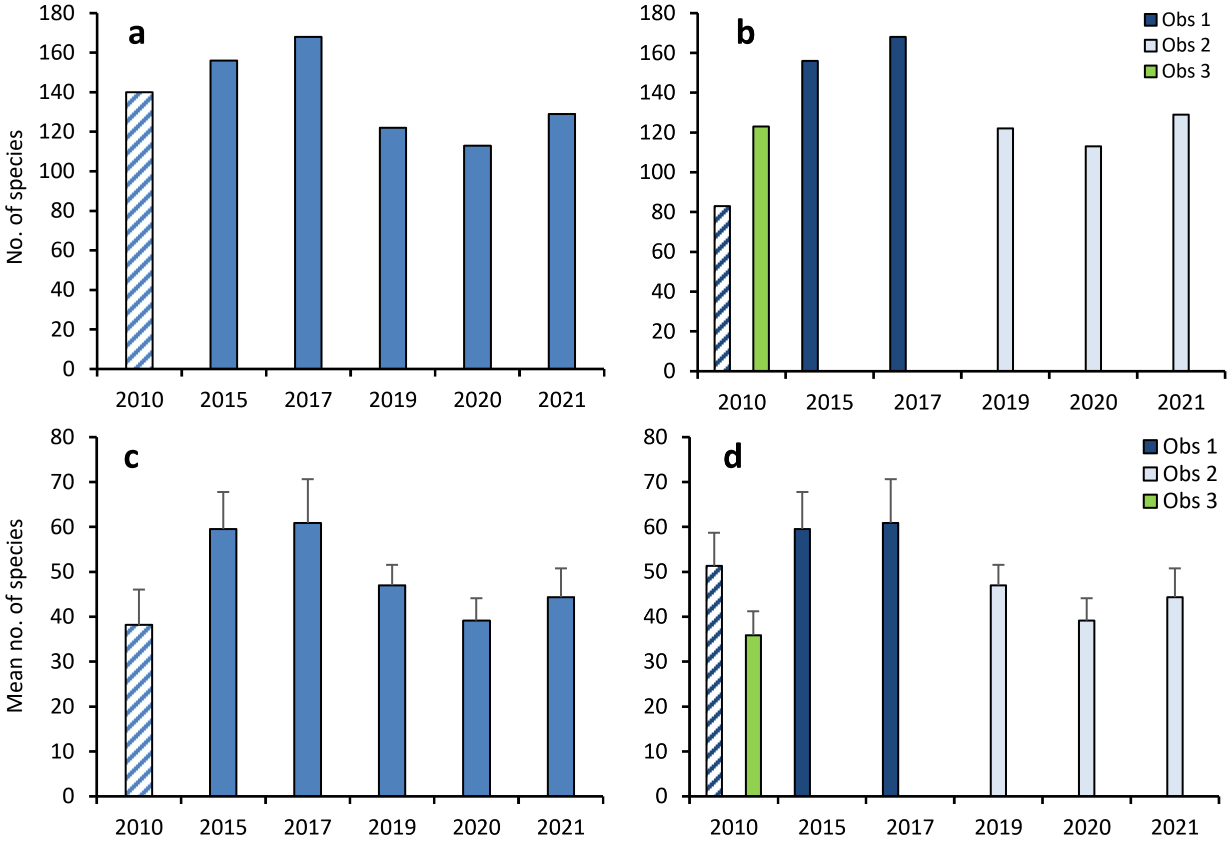


Figure 31. Total (a, b) and mean (c, d) species richness within the East Agana Bay HPRA between 2010 and 2021, with data from all observers combined (a, c) and presented separately (b, d). Note that the time interval between survey years varied. Species richness values in 2010 (represented by bars with the diagonal lines) should be considered cautiously, as these data include observations obtained by Observer 1 at only three out of the usual 20 sampling stations. Data collected by Observer 4 at the 17 other sampling stations in 2010 were excluded from the analysis for reasons discussed previously.

Macroinvertebrates

While sea cucumber densities within the East Agana HPRA were consistently low, likely due in large part to the high coral cover (which leaves limited amounts of preferred habitat and may obscure some individuals from the observer), sea cucumber density declined by 92% between 2014 and 2016 (Partover.test, $p < 0.01$) and remained very low through 2021 (Figs. 32–33). The high degree of variability in sea cucumber densities across sampling stations in 2010 obscured any changes that may have occurred between 2010 and 2012. The decline in sea cucumber densities within the East Agana HPRA appeared to occur later than the decline in sea cucumber densities detected in the Tumon Bay HPRA, but the back-to-back thermal stress events in 2013 and 2014, and the suspected marine disease first observed in Tepungan Bay in 2014, are likely to have played major roles in the decline.

The mean density of edible shells, comprised primarily of top shells (*Tectus* spp.), was similar in 2010 and 2021, but appeared to have fluctuated in the intervening years (Figs. 32–33). Edible shell density reached a peak of 6.1 ind·100 m^{-2} in 2012 but then declined (by 69%) to around 2 ind·100 m^{-2} in 2016 (Partover.test, $p = 0.01$). The density of edible shells steadily increased after 2018, representing a doubling between 2018 and 2021 (Partover.test, $p = 0.04$. The density of large sea stars may have declined by 80% between 2012 and 2014 (Partover.test, $p = 0.09$), but this change is only significant at the 0.1 level. Edible shell density remained low, at around 0.1 to 0.2 ind·100 m^{-2}, through 2021. No large sea urchins were detected within the East Agana Bay HPRA in 2010 or 2012, but after a single *Echinothrix diadema* was detected in 2014 urchin density appeared to increase each year through 2020 before dropping in 2021.

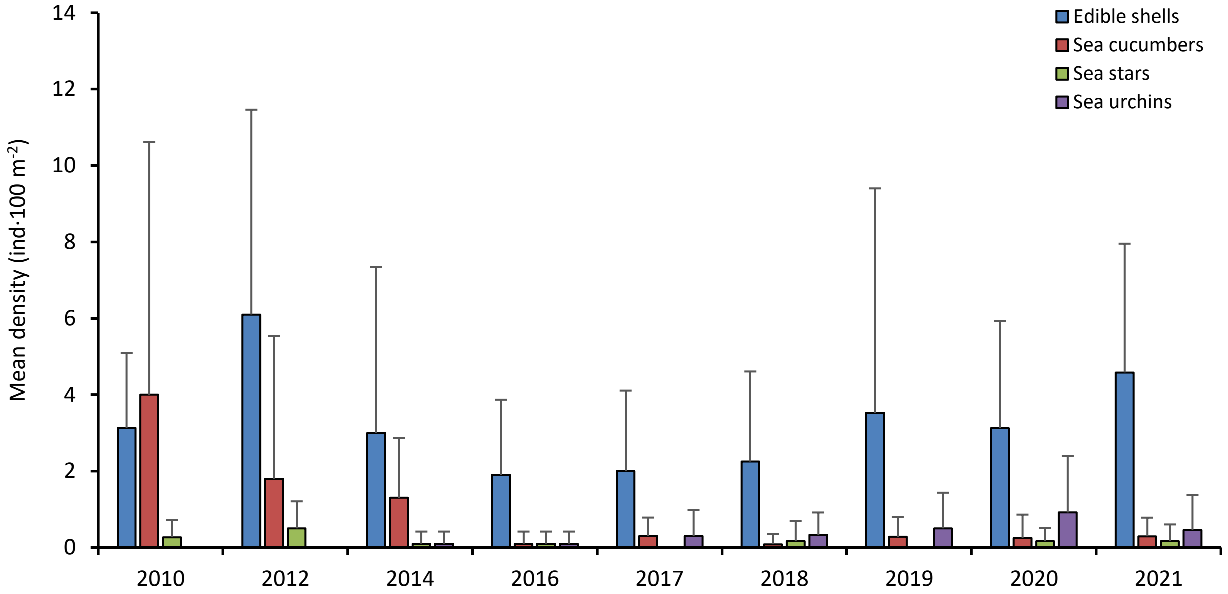


Figure 32. Mean density (ind·100 m^{-2} ± SD) for broad macroinvertebrate groups of interest for the East Agana Bay site between 2010 and 2021. Note that the time interval between survey years varied.

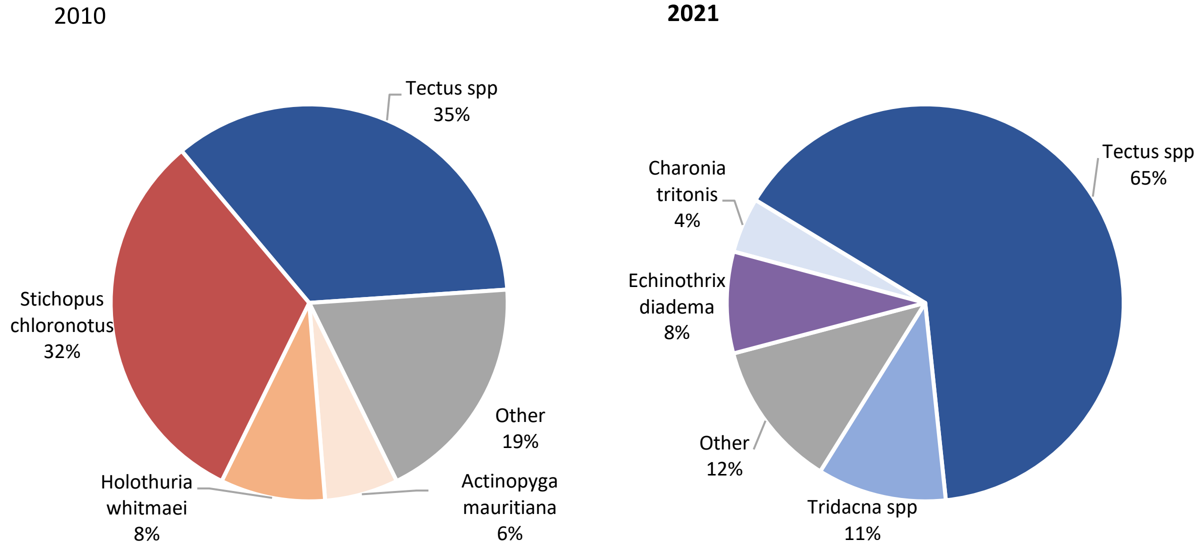


Figure 33. Relative abundance of individual macroinvertebrate taxa within the East Agana Bay HPRA in 2010 and 2021.

Discussion

The relative stability in total coral cover along the submarine terrace within the East Agana Bay site suggests that, like the coral community within the Tumon Bay HPRA, the coral community is relatively robust to the levels of stress experienced during recent coral bleaching events. Although a reef flat site within East Agana Bay has not been surveyed as part of the Reef Flat Monitoring Program, staghorn coral communities within the bay assessed during the 2015 staghorn coral mortality assessment were estimated to have experienced a 20% mortality rate as a cumulative result of thermal stress evens in 2013 and 2014. While this mortality rate was the lowest among all major staghorn thickets assessed in that study, the decline in staghorn coral on the reef flat in East Agana Bay between

2013 and 2015 is still significant and is in contrast to the stable coral cover values within the submarine terrace zone during this period.

The stability in coral cover on the submarine terrace targeted within the East Agana Bay HPRA is also in contrast to major losses of live coral cover at shallow (5 m) reef front sites around the island discussed previously, and to the loss of coral cover observed at the single, shallow (5 m) bleaching response and recovery site (390) occurring within the bay's reef front zone. Total coral cover at the site, which occurs adjacent to the submarine terrace zone targeted within the East Agana Bay HPRA, declined from 35% to 18% between 2013 and 2017; this represents a 49% decline relative to the 2013 mean (two-sided t-test, p = 0.01). Coral cover appeared to rebound significantly at the reef front site, reaching 32% by 2021. A more in-depth analysis will reveal how the coral community may have shifted during this period, providing an indication of coral bleaching "winners and losers."

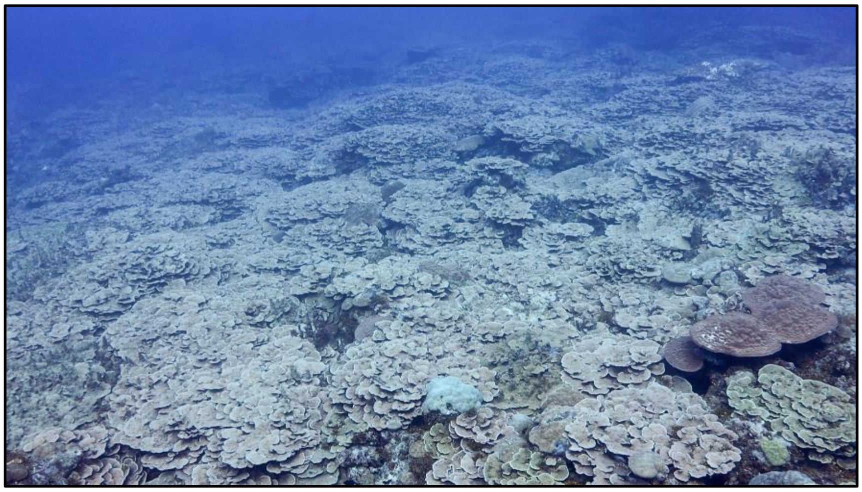
As discussed previously in regard to the reef front in Tumon Bay, it's important to consider that many of the coral taxa most susceptible to bleaching are uncommon along the west coast, or occur at depths shallower than the 5 m depth contour targeted at the island-wide survey sites. Anecdotal observations made by the MPC along the reef front in the eastern and western portions of Agana Bay between 2004 and the present suggest significant losses of *Acropora* spp., some *Pocillopora* spp. and other coral taxa in these shallower (< 5 m) reef front coral communities (see image at right and on p. 71).

While the large Acropora cf. pulchra thickets on the outer reef flat in the center of East Agana Bay suffered some losses as a result of the 2013 and 2014 event, significant areas with live coral were still observed in this area in 2015 (top), and more recent (2022) observations indicate these thickets continue to persist. The bottom image depicts numerous recently killed Acropora scattered among the mostly living Pocillopora setchelli colonies in the shallow (~3 m) reef front near the bleaching monitoring site in East Agana Bay in November 2013. Like many other reef tracts along the northwest coast of Guam, the upper few meters of the wave-exposed reef front in East Agana Bay hosted large numbers of Acropora, but most of these corals perished as a result of heat stress between 2013–2015.

As discussed for the coral community of the Tumon Bay HPRA, the relatively high resilience of the coral community in the East Agana Bay HPRA to recent levels of thermal/light stress is likely a result of the dominance of the generally stress-tolerant *Porites* corals and the very low abundance of stress-susceptible taxa. The already-low cover of stress-susceptible taxa present at the site prior to the onset of severe coral bleaching events in 2013 appeared to remained relatively stable over the last decade, but a more in-depth investigation of changes in the cover or colony density of stress-susceptible taxa is required to better understand which species may have been most impacted by the repeated bleaching events.

The apparent resilience of the coral community along the submarine terrace in East Agana Bay monitoring site may not persist as sea surface temperatures continue to increase, and the frequency and severity of bleaching events increases in response. The dominance of the coral community by a single coral species (*Porites rus*) may make it more vulnerable to devastation by diseases, the virulence of which may be enhanced by warming ocean temperatures.

With the previously discussed data and staff capacity limitations noted, there are some indications that food fish biomass may have decreased significantly along the submarine terrace in East Agana Bay after 2010, and that

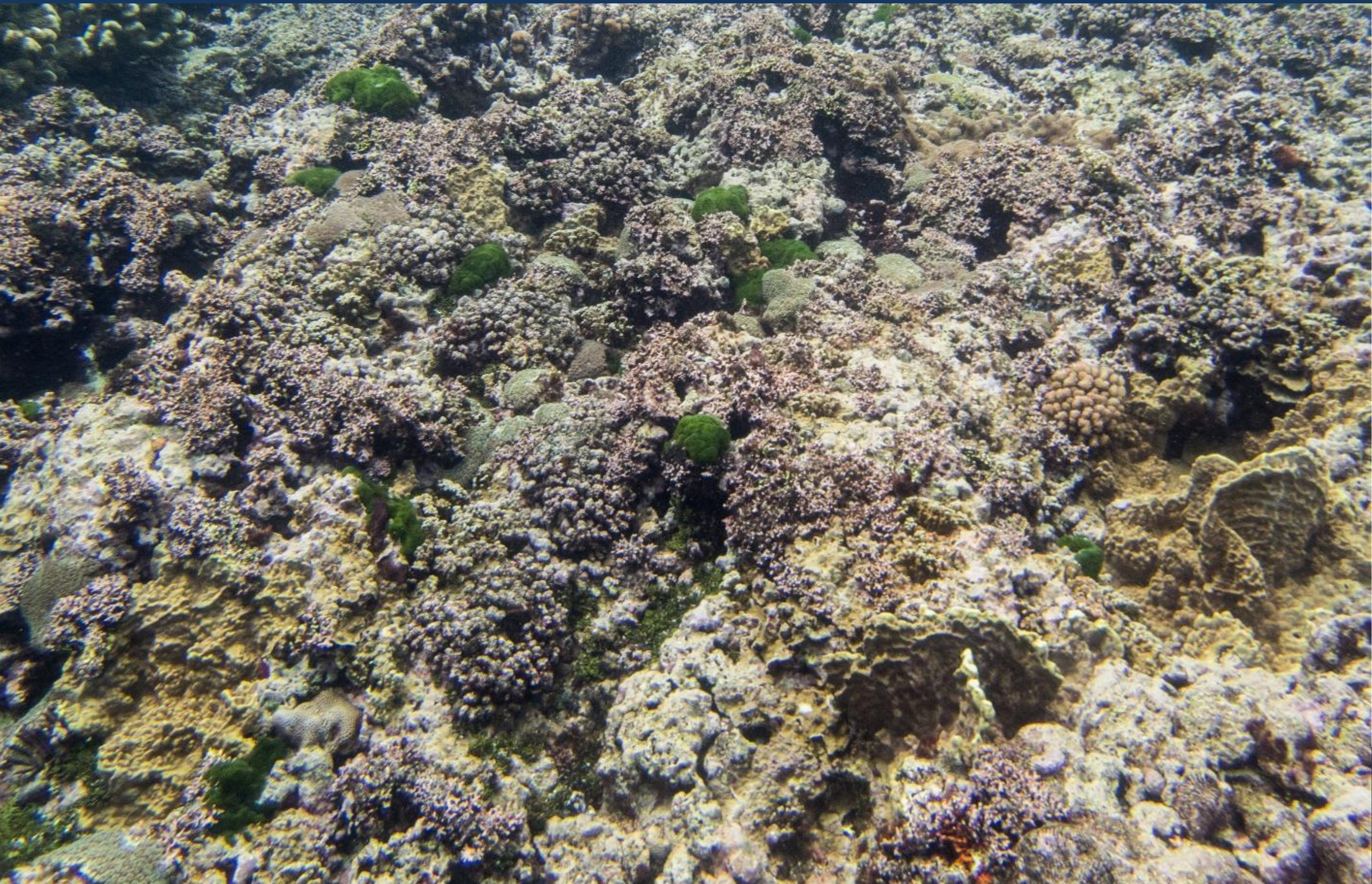


Some areas of East Agana Bay host significantly higher coral cover than the ~45% average cover detected in the bay between 2010 and 2021, such as this deeper (~15-18 m) area in the southwest of the bay (photographed in 2020), but this cover is mostly comprised of a single species (Porites rus).

the density of small food fishes may have declined between 2015 and 2020. The reef fish survey data obtained by a single observer since 2019 allows for more confidence in the assertion that total food fish biomass likely increased steadily after 2019, and that the density of small food fishes increased between 2020 and 2021. While statistical comparisons have not yet been made for individual food fish families through time, the increase in food fish biomass between 2019 and 2021 appears to have been primarily a result of an increase in the biomass of jacks and the “other families” group, which includes triggerfishes, fusiliers, goatfishes and other food fish families; the biomass of all other food fish families appears to have remained relatively stable. Total fish biomass in the East Agana Bay HPRA, data for which was last obtained in 2019, was relatively low, at 23% of the potential total reef biomass estimated for an impaired Guam reef by Williams et al. (2015). Further study of the reef fish data obtained by the GLTMP within the East Agana Bay HPRA, possibly in combination with similar datasets (e.g., NOAA PIFSC RAMP cruise fish surveys) may better resolve any trends in the reef fish community that may have occurred within the East Agana Bay HPRA and other HPRAs over the last decade, especially in the earlier years when the GLTMP data for the East Agana HPRA was particularly sparse and affected by significant interobserver bias issues. Further investigation may also provide an indication of the likely causes of any declines (e.g., poaching or habitat degradation) or improvements (e.g., fisheries policy or improvement in habitat condition) in reef fish community condition, and may facilitate robust comparisons to reef fish communities within the Tumon Bay HPRA, as well as to the reef fish communities at other HPRAs and other reef areas around Guam.

The striking decline in sea cucumber densities, particularly of the spiky sea cucumber (*Stichopus chloronotus*), within the East Agana Bay HPRA between 2014 and 2016 is similar in magnitude to the decline observed within the Tumon Bay HPRA. Sea cucumber densities remained low through 2021, as they did within the Tumon HPRA. The decline in sea cucumber densities within the East Agana HPRA occurred later than the decline detected in the Tumon HPRA, but could still have been driven by the 2014 heat stress event. The significant decline in edible shells (primarily top shells) between 2012 and 2016 may also have been directly or indirectly due to heat stress, but the possible role of harvesting should also be considered. Edible shell density did not appear to be significantly affected by the 2017 heat stress event, and steadily increased through 2021. As suggested previously, the possible link between ocean warming, marine diseases, and mass mortality of reef invertebrate species—especially commercially and ecologically important taxa, should be investigated further. The consequences of significant losses of reef invertebrate taxa to the overall reef function of Guam’s reefs should also be investigated, and possible mitigation actions explored.

***Next page**: Coral reef communities along the same stretch of the shallow (2.5 m), wave-exposed reaches of the seaward slope in West Hagåtña Bay in 2007 (top) and 2018 (bottom). Prior to the onset of multiple severe coral bleaching events beginning in 2013 this reef area, as with other shallow reef areas around the island exposed to a moderate amount of wave energy, were lush with coral, and hosted species rarely found outside of this relatively narrow band of reef. The reef communities in this “goldilocks” reef zone appeared to be resilient to local stressors, such as crown of thorns outbreaks, impacted herbivore populations, and degraded water quality, while their deeper counterparts lost significant amounts of coral over the last few decades or were dominated by just a handful of stress-tolerant coral species. But the abundance of stress-susceptible corals, especially Acropora spp., made these shallow reef front communities especially vulnerable to heat and light stress during the multiple, severe coral bleaching events that affected Guam’s reefs between 2013 and 2017. The data collected at slightly greater depth (5 m) along the reef front during these events showed significant declines around the island, and the loss of almost 60% of all corals along the east coast. Anecdotal observations and data collected at shallow transects in Pago Bay suggest that the loss of coral was even greater (>90%) at these shallower depths, with some corals that were once locally abundant almost completely disappearing.*



PITI BOMB HOLES MARINE PRESERVE

Key findings

- Coral cover was relatively low (16%) and was relatively stable between 2010 and 2020, despite the multiple bleaching events affecting Guam's reefs during this period
- The overall stability in coral cover can likely be attributed to the dominance of bleaching-resistant coral species (mainly mounding *Porites* spp.), but losses in other coral taxa were detected
- Data and staff capacity limitations currently prevent detailed analyses of reef fish survey data obtained between 2012 and 2020, but there are some indications that food fish biomass remained low between 2012 and 2020
- The biomass of parrotfishes may have increased between 2018 and 2020, while all others remained relatively stable
- Total fish biomass in 2018 was relatively low (12–21 $g{\cdot}m^{-2}$), at 23–40% of the potential total reef fish biomass estimated for an unimpaired Guam reef by Williams et al. (2015)
- The density of small food fishes likely increased between 2018 and 2020
- The significant declines in the densities of sea cucumbers and edible shells (mainly top shells) between 2012 and 2014 was likely directly or indirectly related to heat stress events; edible shell densities remained low through 2021, while sea cucumber densities increased significantly after 2017

Above*: A reef community to the west of Tepungan Channel in the Piti HPRA in 2020. This area hosts a greater proportion of stress-susceptible coral genera compared to the reef area east of the channel (see photo on p. 47).*

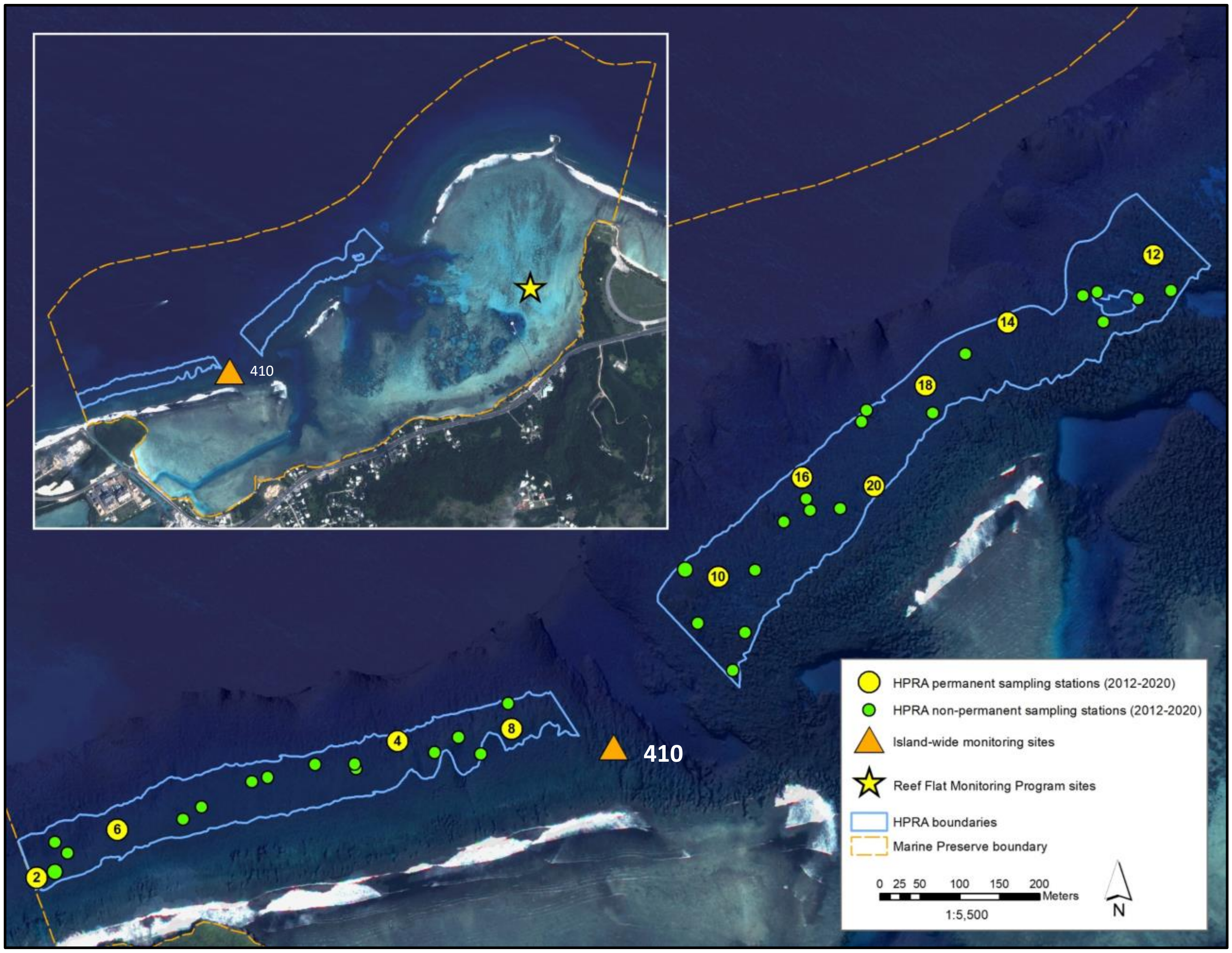


Figure 34. Map of the Piti (Tepungan) Bay HPRA site boundaries and sampling stations, as well as one of the island-wide bleaching response and recovery sites and the reef flat monitoring site within the bay.

Site overview

The Piti Bomb Holes Marine Preserve was selected for long-term monitoring upon completion of baseline monitoring at Western Shoals in 2011, with baseline data collection occurring in 2012. Piti (Tepungan) Bay is considered a high priority reef area on account of its designation as a locally managed Marine Preserve, its location downstream from watershed restoration activities, its heavy use by commercial tourism operators and resident recreational users, and its location adjacent to a non-preserve site (Asan) monitored on a regular basis by the National Park Service. Since its establishment as an HPRA, a coral nursery, coral mitigation sites, and a community-based coral reef monitoring program site have been established in the bay.

To maintain consistency with the East Agana and Tumon Bay monitoring sites, monitoring in Piti (Tepungan) Bay occurs within the same reef zone and depth range. However, in order to contribute to an understanding of the impacts of land-based sources of pollution on coral reef ecosystems within the bay, and to assist in the evaluation of the effectiveness of watershed restoration activities in improving water quality and coral reef health, the HPRA includes two separate sections of submarine terrace zone (7–15 m depth): one extending to the west of Tepungan Channel and one extending to the east of the channel (Fig. 34). A single permanently established island-wide bleaching response and recovery site (410) is located along the reef front on the western side of Tepungan Channel, while a RFMP site is located on the shallow reef flat approximately 200 m to the northeast of the Fish Eye Marine Park observatory.

Baseline data collection occurred at the Piti HPRA in 2012, with subsequent data collection efforts occurring in 2014, 2018, and 2020 (Appendix A). Staffing limitations and the re-prioritization of resources required to respond to the multiple bleaching events between 2013 and 2017 prevented the collection of data the Piti site on an annual basis and limited the collection of data for multiple sampling years. In addition to data collected during regular visits to the HPRA site along the seaward slope, benthic cover and macroinvertebrate abundance data was also collected at a single shallow (5 m depth) site along the reef front in Piti (Tepungan) Bay. This reef front site (410) was re-visited in 2015 as part of an island-wide bleaching recovery assessment, in 2016 as part of an island-wide reef resilience assessment, in 2017 as part of the response to another major bleaching event and was surveyed and marked with rebar in 2021. The UOGML Reef Flat Monitoring Program led by Dr. Raymundo has collected benthic cover and coral health data at a reef flat site in the bay since 2009.

The results of analyses of available benthic cover, reef fish, and macroinvertebrate data obtained within the Piti HPRA between 2012 and 2020 are provided below. The results of an analysis of data obtained at the reef flat monitoring site are presented on p. 40 and summarized in the "Discussion" section below. While a detailed analysis of individual the island-wide bleaching response and recovery sites has not yet been conducted, the results of a preliminary analysis of data obtained at the single island-wide site located near the Piti HPRA is presented in the "Discussion" section below. The results of an analysis of data obtained at staghorn locations within the bay are summarized in the "Discussion" section below and presented in more detail in Raymundo et al. (2022).

Benthic cover

There was no detectable change in mean total coral cover across the Piti HPRA between 2012 and 2020, remaining at 16% for both years (Fig. 35). However, coral cover increased by 25% (from 16% to 20%) between 2012 and 2014 (Partover.test, p = 0.04) before falling back to 16% in 2018 (Partover.test, p = 0.05), where it remained through 2020. In contrast to the *Porites rus*-dominated coral communities in the Tumon Bay and East Agana Bay HPRAs, the coral community within the Piti HPRA is dominated by mounding *Porites* spp., and is comprised of a larger percentage of non-*Porites* taxa, such as *Leptoria phrygia*, *Dipsastraea* spp., and other merulinids (Figs. 36–37). Mounding *Porites* spp. consistently comprised greater than 50% of total cover in the Piti HPRA between 2012 and 2020, while *P. rus* comprised between 6% and 9%, merulinids comprised between 11% and 16%, and other non-*Porites* coral taxa comprised between 9% and 18%.

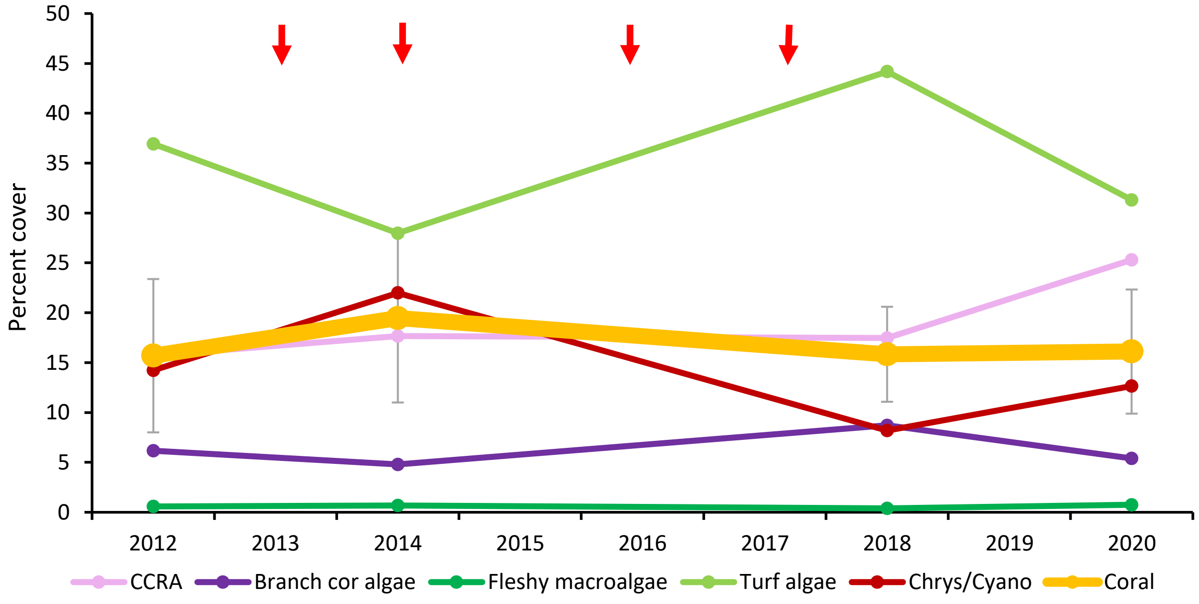


Figure 35. Mean (± SD) percent cover of major benthic cover classes for the Piti HPRA between 2012 and 2020. Red arrows indicate the timing of significant coral bleaching events, including the historically severe 2013 and 2017 events. Note that the elapsed period of time between each pair of neighboring data varies from two to four years.

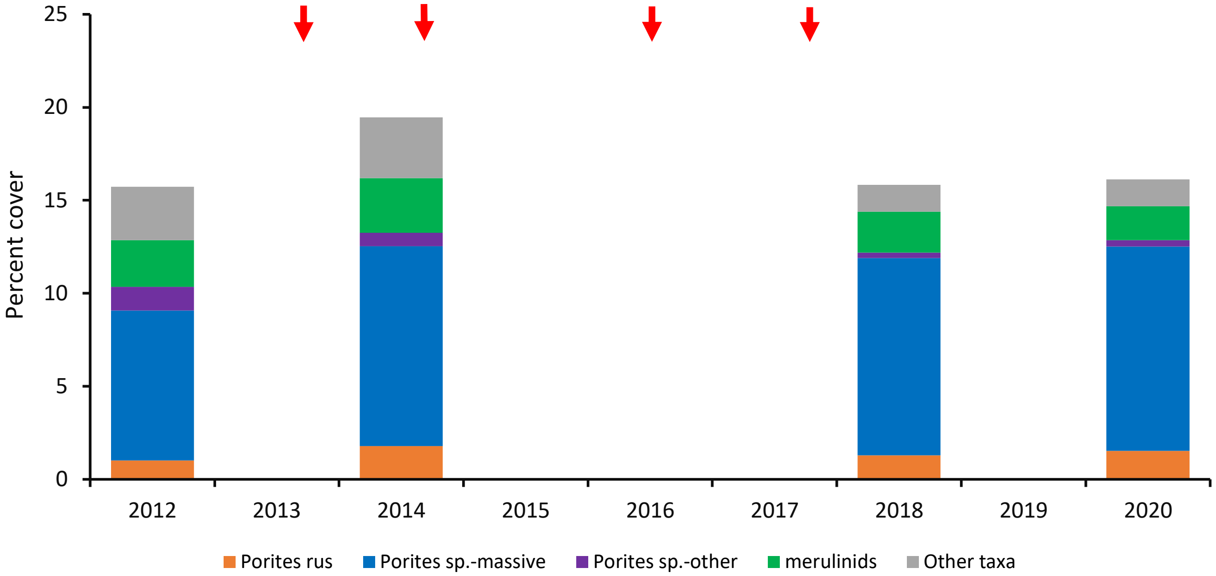


Figure 36. Percent cover of coral taxa for the Piti HPRA between 2012 and 2020. Red arrows indicate the timing of significant coral bleaching events, including the historically severe 2013 and 2017 events. Note that survey years are not presented in regular annual increments, and thus the elapsed period of time between each pair of neighboring data varies from two to four years.

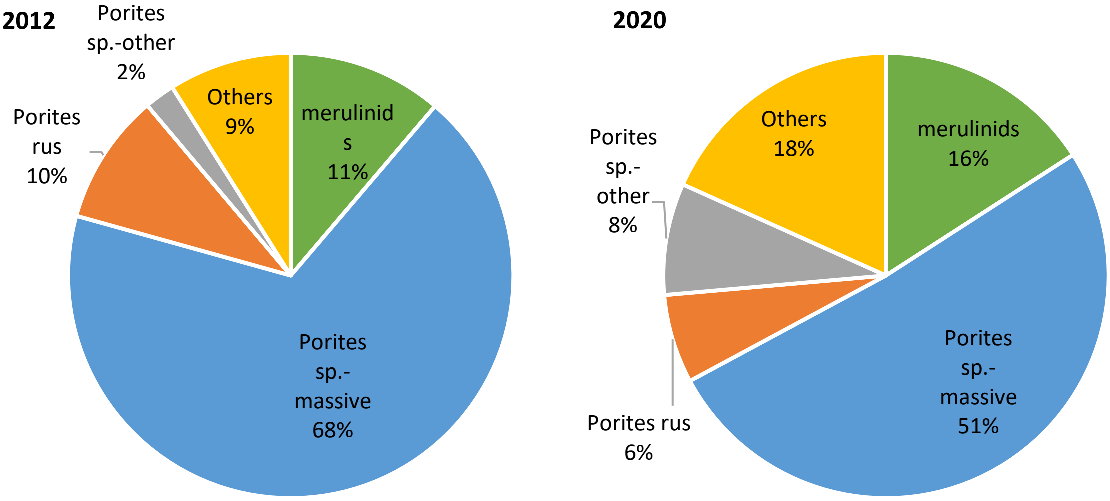


Figure 37. Percent contribution of individual coral taxa/taxa groupings to mean total coral cover for the Piti HPRA in 2010 and 2020.

The increase in the cover of mounding *Porites* spp. from 8% to 11% between 2012 and 2014 comprised the majority of the change in total coral cover during this two-year period, while a decrease in other *Porites* spp. (not including *P. rus*) from 1.3% to 0.7% was the only other statistically significant change in the cover of other coral taxa detected during this period. The increase in total coral cover and the cover of massive *Porites* species observed between 2012 and 2014 occurred despite the impact of the 2013 coral bleaching event, likely owing to the high tolerance to thermal stress of mounding *Porites* spp. relative to many other coral taxa. The cover of mounding *Porites* remained at 11% in 2018 and 2020, despite the impacts of the 2017 bleaching event, but a decline in cover of merulinids and other non-*Porites* taxa was still detectable between 2014 and 2020 despite the already low cover values for these groups.

Merulinids, which, on average, appear to be moderately susceptible to bleaching impacts and predation by COTS, declined from 2.9% to 1.8% between 2014 and 2020 (Partover.test, p = 0.04), while all other non-*Porites* taxa (combined into the "Others" group) declined from 3.3% to 1.4% (Partover.test, p = 0.002) during this period. The relatively long time intervals between data collection activities at the Piti HPRA make it difficult to pin-point exact causes for the declines in the non-*Porites* coral taxa between 2014 and 2020, but it is likely that thermal stress, disease, and predation by COTS all played significant roles in this decline.

Statistical comparisons were not made for non-coral benthic features through time, but as with both the Tumon HPRA and East Agana HPRA, the cover of fleshy macroalgae was very low (<1%) throughout the study period (Fig. 35). The cover of branching coralline algae, comprised mainly of the macrophyte *Halimeda* spp., remained between 5% and 6%, while the cover of crustose coralline red algae appeared to increase from 16% to 25%. Most of the increase in CCRA was comprised of taxa within the Order Corallinales (the pink and red crusts), but other encrusting algae also increased during this period. The cover of the turf algae fluctuated between 37% and 44%, but consistently occupied the largest proportion of the benthic community, and the generally ephemeral cyanobacteria/chrysophytes ranged between 8 and 22% during the study period.

Associated biological communities

Reef fishes

Biomass

Mean food fish biomass within the Piti HPRA appeared to decrease by 45% (from 21.4 g·m^{-2} to 11.8 g·m^{-2}, Partover.test, P = 0.03) between 2012 and 2020 (Fig. 38a). However, as discussed above, interobserver bias must be considered when interpreting trends using data collected by the two different observers. In this case, the generally

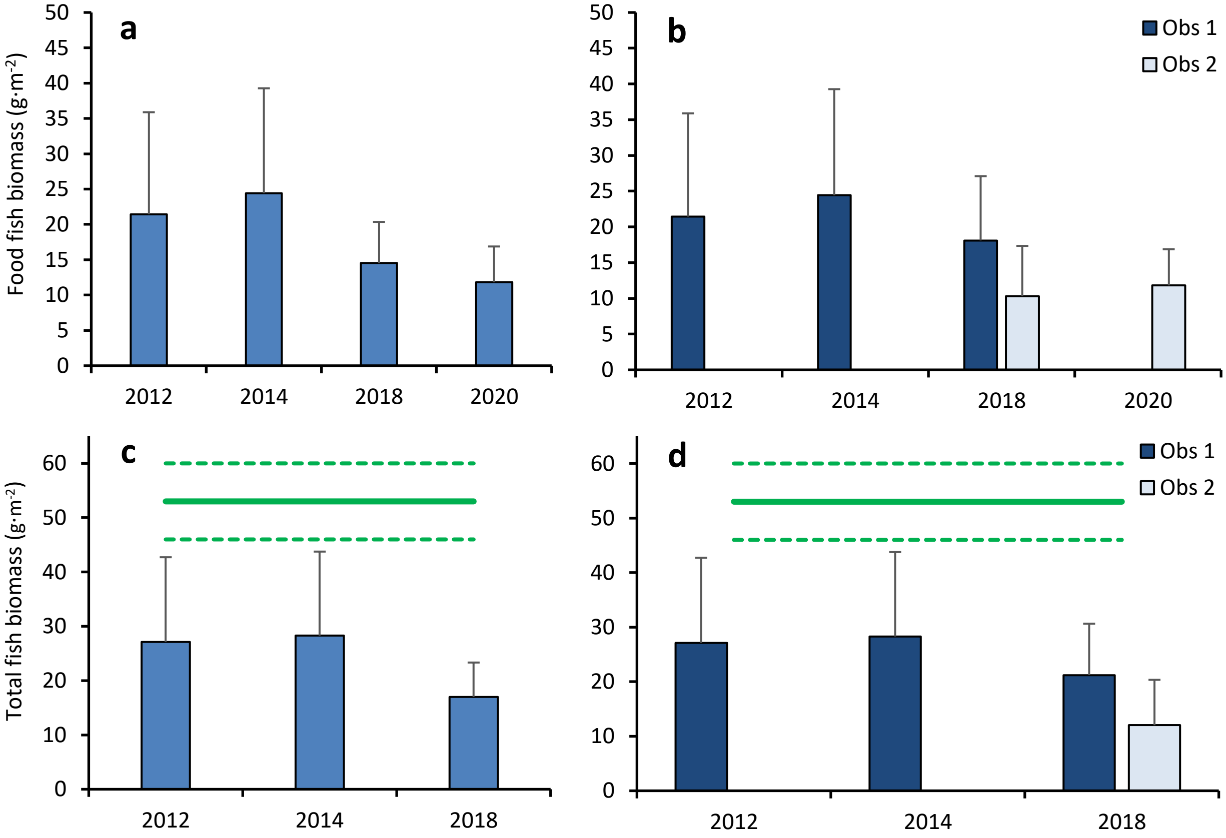


Figure 38. Food fish biomass (a, b) and total fish biomass (c) within the Piti HPRA between 2012 and 2020, with data from all observers combined (a, c) and presented separately (b, d). Note that the time interval between survey years varied. The green lines indicate the potential total reef fish biomass for an unimpaired Guam reef community (53 ± 7 g·m^{-2}) estimated by Williams et al. (2015).

higher biomass values reported by Observer 1, who carried out most surveys prior to 2019, in comparison to Observer 2, who carried out all surveys during and after 2019, likely means that the decrease reported between 2015 and 2021 may not reflect an actual decrease in food fish biomass within the Piti HPRA during this period (Fig. 38b). Indeed, when using only those observations obtained by Observer 1, no significant difference in food fish biomass was detected between 2012 and 2018 (Partover.test, p = 0.4). The slight difference in food fish biomass recorded by Observer 2 in 2018 and 2020 was not statistically significant, suggesting food fish biomass remained stable during this period.

As with food fish biomass, total fish biomass appeared to decline within the Piti HPRA between 2012 and 2018, but some or all of this apparent decline can be attributed to interobserver bias (Fig. 38c). When only data obtained by Observer 1 was examined, the slight difference in mean total fish biomass between 2012 and 2018 was not statistically significant (Partover.test, p = 0.17; Fig. 38d). Due to the modification of the survey method to generate count and length data for only food fishes beginning in 2020, total fish biomass cannot be reported beyond 2018 for the Piti HPRA. The total fish biomass values of between 12 and 28 g·m^{-2} reported for the Piti HPRA in 2018 are relatively low (23–58%) in comparison to the target value (53 ± 7 g·m^{-2}) for an unimpaired Guam reef system estimated by Williams et al. (2015), notwithstanding the caveats mentioned above.

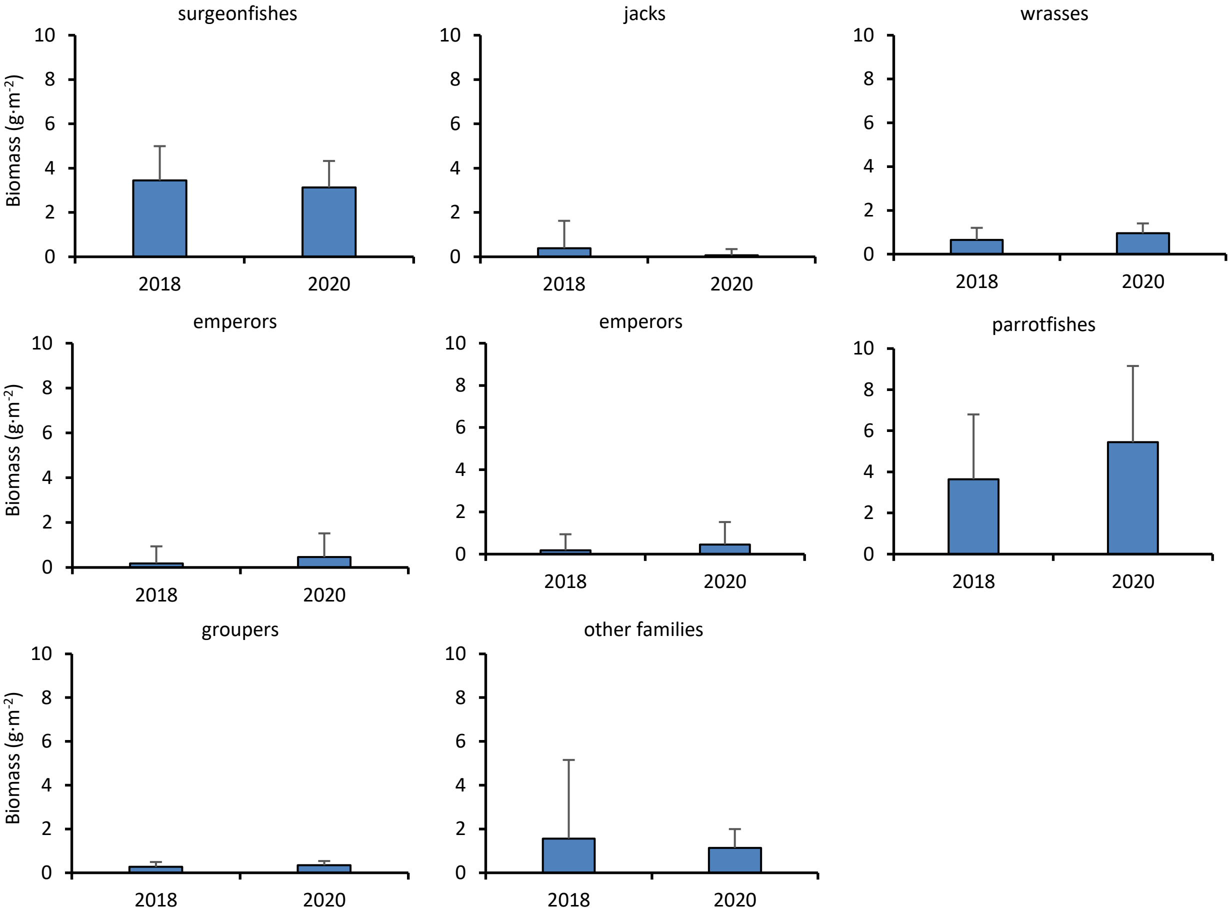


Figure 39. Biomass (g·m^{-2}) of selected food fish families within the Piti HPRA in 2018 and 2020. Due to the significant interobserver bias detected early in the analysis, these values were calculated using only those observations collected by Observer 2, who collected data for the two most recent years (2018 and 2020).

Due to the significant interobserver bias discussed above, biomass values for selected food fish families within the Piti HPRA were calculated using only those observations recorded by Observer 2, who carried out all fish surveys in 2018 and 2020 (Fig. 39). While statistical comparisons were not carried out for this limited preliminary analysis, the biomass of most food fish families appeared to remain relatively stable during this period, with the exception of a possible increase in the biomass of parrotfishes (Partover.test, p = 0.09).

Density

The density of small (≤ 20 cm) food fishes within the Piti HPRA likely increased between 2012 and 2020 (Fig. 40), but the apparent decrease in the density of moderately sized (21 ≤ 40 cm) food fishes is likely a result of interobserver bias, rather than a reflection of an actual decrease. When only those observations obtained by Observer 2 are examined, a clear increase in the density of small food fishes is detected between 2018 and 2020, while the density of moderately sized fishes remained low (but stable) during this period. A very small number of individuals > 40 cm were detected in 2018, while none were observed in 2020. The low densities of moderately sized and large food fishes make the detection of trends in these groups challenging, but a more detailed analysis may be able to detect trends in the abundances of these larger size classes.

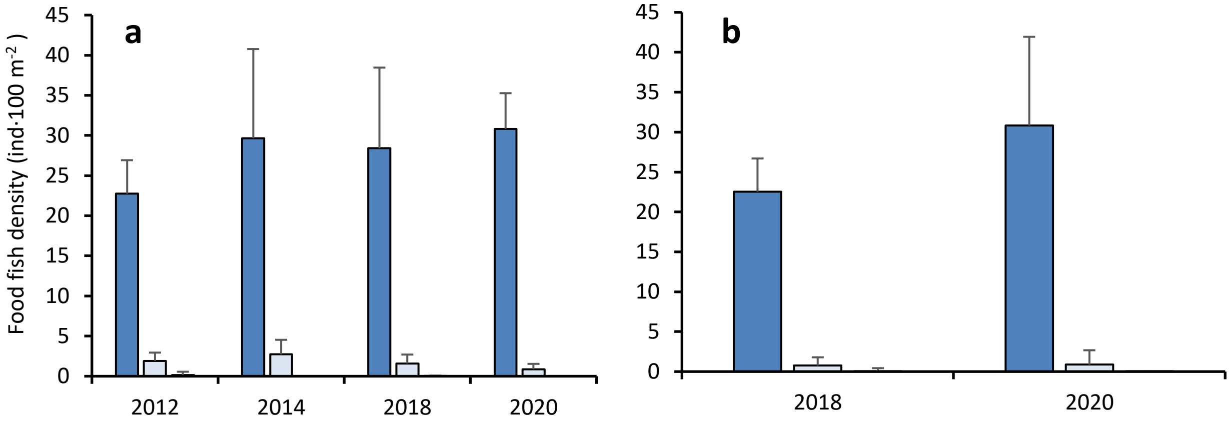


Figure 40. Food fish density (ind·100 m^{-2} ± SD) by size class within the Piti HPRA between 2012 and 2020, with data from two observers combined (a) and with only data from Observer 2 (b).

Species Richness

Both the total and mean number of species recorded within the Piti HPRA appeared to decline between 2012 and 2020, but this decline is very likely a result of interobserver bias rather than a reflection of an actual decline in species richness during this period (Fig. 41). Total species richness ranged from a low of 108 species in 2020 to a high of 164 species in 2018, while mean species richness ranged from a low of 37 ± 5 species per sampling station in 2020 to 56 ± 13 species per sampling station in 2012. When considering only those observations obtained by Observer 2 in 2018 and 2020, total species richness was relatively stable at 111 and 108 species, respectively. As mentioned above, a more detailed analysis of species counts obtained by the different observers is likely to reveal species that are consistently detected by one observer but not another, or other biases that could be minimized with targeted training and calibration. Additionally, a deeper analysis that accounts for rugosity and benthic community structure may also reveal that the fluctuations in the survey data may reflect real-world change in the species diversity in the Piti HPRA.

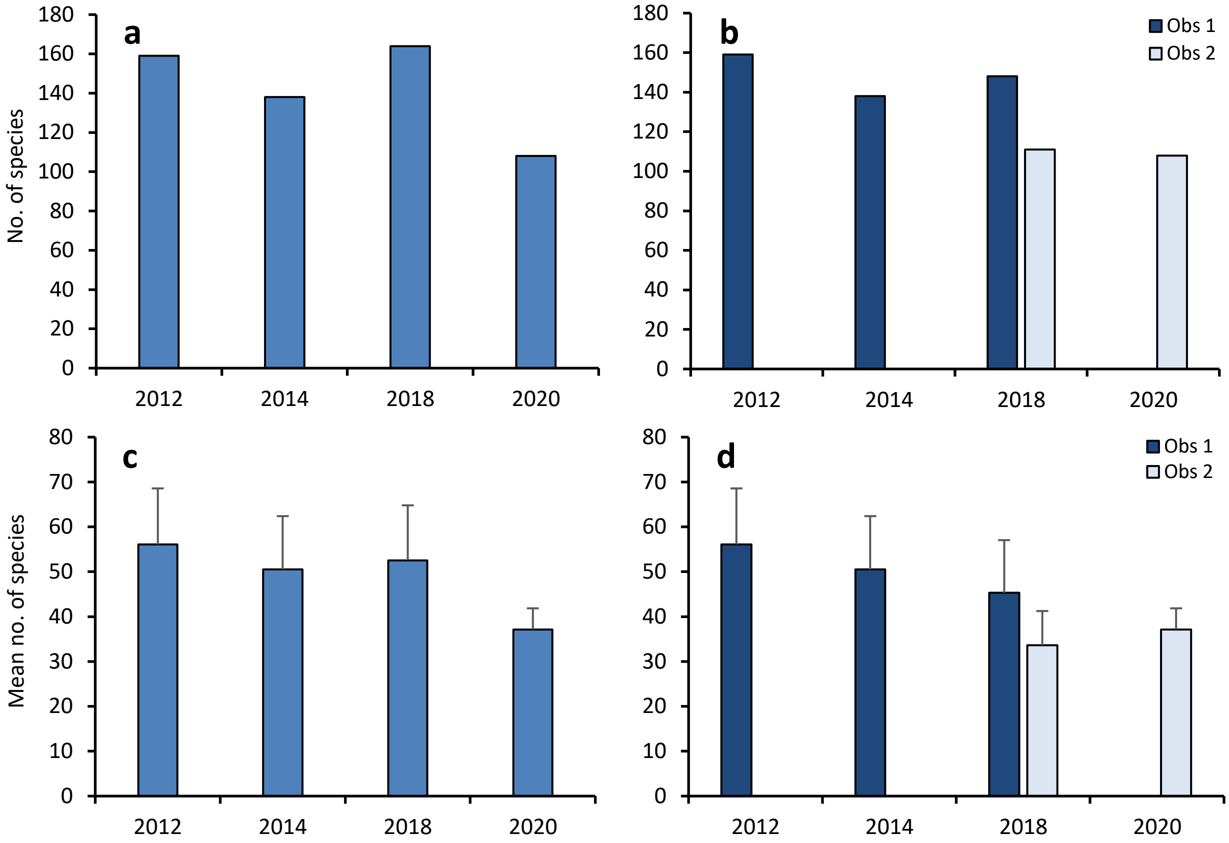


Figure 41. Total (a, b) and mean (c, d) species richness within the Piti HPRA between 2012 and 2020, with data from all observers combined (a, c) and presented separately (b, d). Note that the time interval between survey years varied.

Macroinvertebrates

Mean sea cucumber density within the Piti HPRA declined by 69% (from 6.0 ± 3.7 to 1.8 ± 1.4 ind·100 m^{-2}) between 2012 and 2017 (Partover.test, p = <0.01), before increasing over 300% by 2020 (Partover.test, p = <0.01; Fig. 42–43). Sea cucumber density dropped 38% between 2012 and 2014 alone (Partover.test, p = 0.04). The decline in sea cucumber densities within the Piti HPRA occurred during a period marked by multiple thermal stress events, suggesting a possible link between these events and the decline in sea cucumber densities within the site. The absence of severe thermal stress events after 2017 may have allowed sea cucumber densities to rebound in recent years.

The mean density of edible shells decreased by 67% (from 4.8 to 1.6 ind·100 m^{-2}) from 2012 to 2014 (Partover.test, p < 0.001) and remained relatively stable (at 1.2–2 ind·100 m^{-2}) through 2020 (Fig. 42–43). This decline in edible shells was comprised primarily by a large decline in the density of top shells (mainly *Tectus niloticus*), although the density of giant clams (*Tridacna* spp.) also appeared to decline during this period. The density of large sea stars and sea urchins appeared to have remained relatively stable, at around 0.4 to 1.4 ind·100 m^{-2}), although densities of both groups appeared to have spiked in 2018 before declining again in 2020.

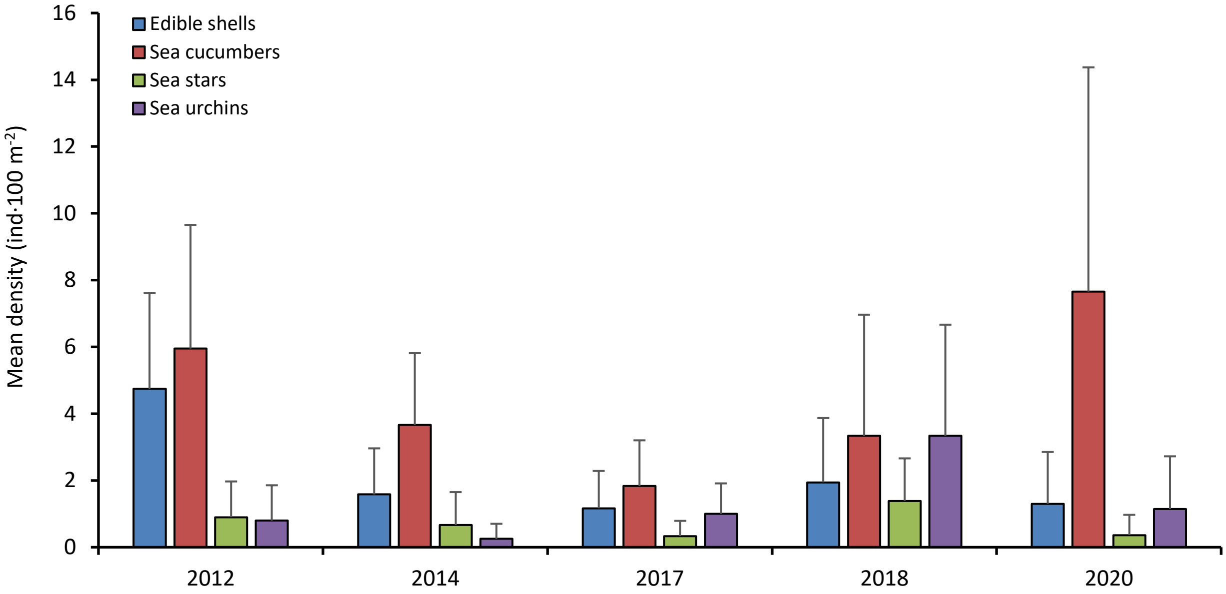


Figure 42. Mean density (ind·100 m^{-2} ± SD) for broad macroinvertebrate groups within the Piti HPRA between 2012 and 2020.

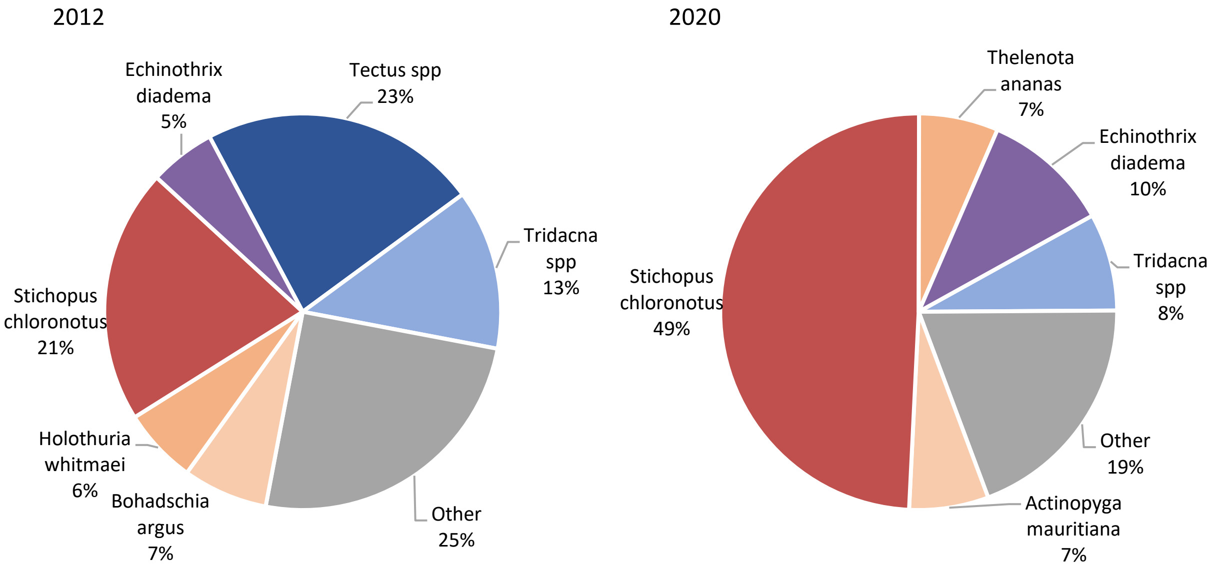


Figure 43. Relative abundance of individual macroinvertebrate taxa within the Piti HPRA in 2015 and 2021.

Discussion

The stability of total coral cover along the submarine terrace within the Piti HPRA during the study period, despite the impact of multiple heat stress events, suggests that the coral community within submarine terrace zone of Tepungan Bay is relatively robust to the levels of stress experienced during these events. It's important to note, however, that while the generally stress-tolerant *Porites* corals that comprise most of the coral cover in the Piti HPRA appear to have fared relatively well over the last decade, the already low cover of merulinids and other non-*Porites* taxa suggests a continued shift towards lower diversity, and an increasing reliance on a handful of coral species to provide essential reef functions.

Relative stability in total coral cover was also observed at a *Porites*-dominated reef flat site surveyed regularly as part of the RFMP. Total cover at the reef flat monitoring site was 35% at the start of monitoring (2009) and at the last year for which data is available (2022). While some periods of decline associated with heat stress, subaerial exposure, and possibly other stressors occurred at the Piti reef flat monitoring site during this period, the relative stability in total coral cover suggests that the coral community at the site is generally resilient to the levels of stress experienced in recent decades.

In contrast to the overall stability of coral cover for the *Porites*-dominated communities of the submarine terrace and reef flat monitoring site, the small staghorn coral thickets that occur on the bay's shallow reef flat platform experienced mortality rates of between 80% and 95% as a cumulative result of thermal stress events in 2013 and 2014[4] (Raymundo et al. 2017). Staghorn corals and other bleaching-susceptible corals comprise a small proportion of the coral communities that occur across the bay's reef flat, however, with stress-tolerant taxa such as *Porites cylindrica*, *Porites rus,* and *Pocillopora damicornis* comprising the majority of hard coral communities across much of the reef flat platform. The presence of staghorn rubble far from any currently living staghorn suggests that staghorn corals were more common in the bay in past decades. The rapid expansion and coalescence of numerous small *A*. cf. *pulchra* thickets within Tepungan Channel and on the reef flat adjacent to the channel, in the far west of the bay, over the last decade or more suggests the conditions in that part of the bay have been more favorable for growth over the last decade, with the near constant current within and near the channel likely conferring some degree of resilience to this population. However, extensive mortality of much of the *A*. cf. *pulchra* on the reef flat next to the channel caused by prolonged subaerial exposure in 2023 dramatically illustrates the vulnerability of these very shallow communities to extreme low tide events.

Staghorn (mainly Acropora cf. pulchra) appears to be thriving in the western side of Tepungan Bay, especially along the bottom of the man-made portion of Tepungan Channel (top), although extreme low tide events in the first half of 2023 have caused significant mortality in colonies on the shallow reef flat adjacent to the channel. Old skeletons of numerous plating and corymbose Acropora colonies along the extensive outer reef flat and shallow reef front in the center of Tepungan Bay in 2008 (bottom, suggest substantial (and likely COTS-associated) mortality of stress-susceptible coral taxa occurred in this area long before the onset of severe bleaching in 2013

Interestingly, total coral cover at the island-wide bleaching response and recovery site, which occurs within the reef front zone adjacent to the submarine terrace targeted within the HPRA, appeared to increase from 21 ± 4% in 2013 to 33 ± 5% in 2017, representing an increase of 57%

---

[4] Data collected in 2015 for the staghorn coral mortality assessment did not include data for numerous small *Acropora pulchra* thickets that occur at the bottom of Tepungan Channel. These thickets, which appear to have been unaffected by the multiple recent bleaching events, were not discovered until after the study was completed, and have since grown rapidly. Staghorn coral species that suffered significant losses, and which may no longer occur in the Tepungan Bay, include *Acropora acuminata*, *A. teres*, and *A. muricata.*

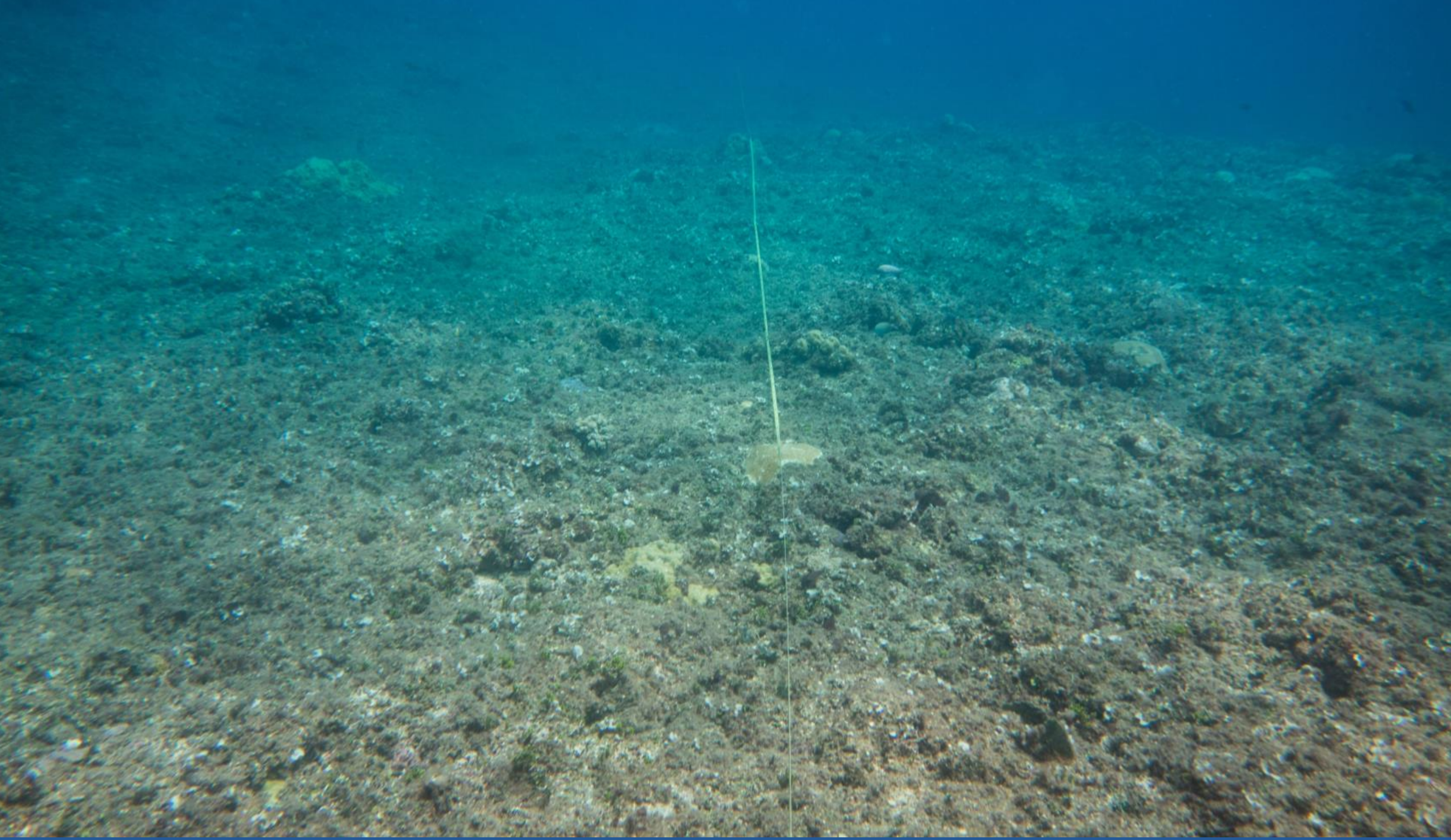
relative to the 2013 mean (Unpaired t-test, p = 0.03). While a statistical comparison has not yet been carried out using more recent data obtained at the reef front site, coral cover in 2021 was 32%, indicating relatively stability after 2017. A more in-depth analysis will reveal how the coral community may have shifted during this period, providing an indication of coral bleaching "winners and losers." As discussed previously, many of the coral taxa most susceptible to bleaching are uncommon along the west coast, or occur at depths shallower than the 5 m depth contour targeted at the island-wide survey sites. Anecdotal observations made by the MPC along the reef front in the eastern and western portions of Piti (Tepungan) Bay between 2004 and the present suggest some losses of *Acropora* spp., some *Pocillopora* spp. and some other coral taxa in these shallower (< 5 m) reef front coral communities (see photos on the next page).

While the overall stability of coral cover along the submarine terrace and at the reef flat monitoring site within Tepungan Bay in recent years is consistent with the lack of detection of change along the western leeward coast in response to recent bleaching events, it is in contrast to the major losses of live coral cover detected at shallow reef front sites island-wide discussed previously. While historical data for Piti Bay is limited, the dominance of stress-tolerant corals and observations of skeletons of long-dead *Acropora* and other stress-susceptible taxa suggest that current benthic assemblages in the bay are the result of selection pressure from decades of local stressors, and that the proportion of stress-susceptible coral taxa—many of which contribute substantially to reef growth and complexity—likely disappeared before they could be devastated by the historically severe bleaching events of recent years. The apparent resilience of the *Porites*-dominated coral community along the submarine terrace and reef flat monitoring site in Tepungan Bay, like the similar communities in the Tumon and East Agana sites, will likely struggle as sea surface temperatures continue to increase, and the frequency and severity of bleaching events increases in response.

While a more robust analysis is required to adequately account for the data and staff capacity limitations, there are indications that food fish biomass remained relatively low within the Piti HPRA between 2012 and 2020 and that the density of small food fishes may have increased between 2018 and 2020. The reef fish survey data obtained by a single observer since 2018 allows for more confidence in the assertion that total food fish biomass remained low between 2018 and 2020. While statistical comparisons have not yet been made for individual food fish families through time, parrotfish biomass may have increased between 2018 and 2020, while the biomass of all other food fish families appears to have remained relatively stable. Total fish biomass in the Piti HPRA, data for which was last obtained in 2018, was relatively low, at 23–40% of the potential total reef biomass estimated for an impaired Guam reef by Williams et al. (2015). Further study may better resolve any trends in the reef fish community that may have occurred within the Piti HPRA and other HPRAs over the last decade and may provide an indication of the likely causes of any declines or improvements in reef fish community condition. While the GLTMP and the National Park Service's (NPS) Pacific Islands Inventory and Monitoring Program use different reef fish survey methods, careful integration of fish and benthic data obtained in the Piti HPRA and the Asan reef area monitored regularly by NPS may allow comparisons between these Marine Preserve (Piti) and non-preserve (Asan) reef areas.

The decline in sea cucumber (mainly the spiky sea cucumber) and edible shell (mainly top shell) densities within the Piti HPRA between 2012 and 2014 is similar in magnitude and timing to the decline observed within the Tumon Bay and East Agana Bay HPRAs. It is not clear why sea cucumber densities within the Piti HPRA increased significantly after 2017, in contrast to the low densities reported from the other two HPRAs through 2021, or why edible shell density within the Piti HPRA remained low through 2021 but increased within the Tumon and East Agana HPRA. The decline in sea cucumber and edible shell densities within the Piti HPRA also appears to have been a direct or indirect (e.g., through a marine disease) result of the heat stress events in 2013 and/or 2014, and may also have been affected by the 2017 event. As suggested previously, the possible link between ocean warming, marine diseases, and mass mortality of reef invertebrate species should be investigated further, and a better understanding of the possible consequences of significant losses of reef invertebrate taxa to the overall reef function of Guam's reefs be sought.

Numerous branching corals (mostly Acropora spp.) at ~2–3 m depth in the upper reef front in Tepungan Bay recently killed by heat stress in 2013 (left), and a primarily Pocillopora- and Goniastrea-dominated benthic community in the same vicinity in 2021 (right). The shallowest ~1–4 m of the reef front zone along the west coast often hosted significant numbers of Acropora spp. prior to the onset of severe bleaching in 2013. These areas were too shallow to safely survey around the island, so the observed catastrophic loss (>90%) of coral in this part of the reef could only be visually estimated.



FOUHA BAY

Key findings

- Coral cover remained moderate (18%) at the transects inside the bay between 2015 and 2019, but coral cover at the transects at the mouth of the bay decreased from the already very low 2% cover to ~1%
- *Porites rus* and mounding *Porites* spp. remained dominant at the inner transects, but displayed clear evidence of sediment stress; the decline detected at the outer transects was primarily among non-*Porites* taxa
- Data and staff capacity limitations currently prevent detailed analyses of reef fish survey data, but there are some indications that the generally very low food fish biomass may have increased between 2015 and 2019
- Total fish biomass in 2019 was very low (3–8 $g \cdot m^{-2}$), at 6–15% of the potential total reef fish biomass estimated for an unimpaired Guam reef by Williams et al. (2015)
- The biomass of surgeonfishes and “other families” may have increased between 2019 and 2021
- The density of small food fishes may have declined between 2015 and 2019, but likely increased between 2019 and 2021
- The density of edible shells (mainly giant clams) declined significantly between 2015 and 2019, possibly directly or indirectly related to heat stress events, but increased by 2021; no sea cucumbers were observed during any sampling year

***Above**: An algae-dominated benthic community with very low (<2%) coral cover along the submarine terrace on the north side of Fouha Bay in 2021.*

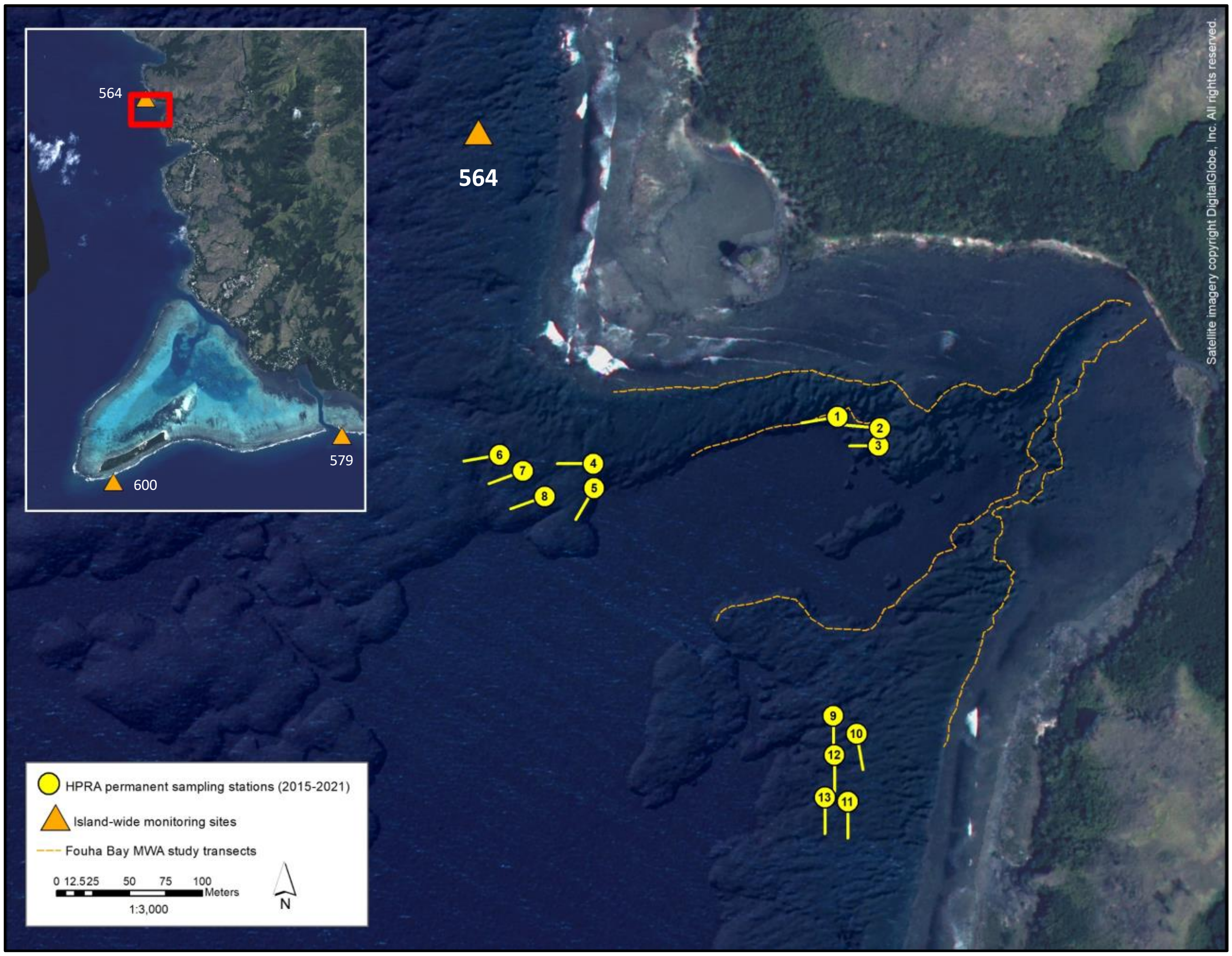


Figure 44. Map of the Fouha Bay HPRA sampling stations and one of the island-wide bleaching response and recovery sites in the vicinity of the HPRA.

Site overview

Fouha Bay was the most recent long-term monitoring site to be established, with baseline data collection occurring in 2015. Fouha Bay is a small, semi-enclosed embayment in southwestern Guam that comprises the nearshore extent of the La Sa Fua watershed. A small-scale watershed restoration effort was underway at the time the decision to monitor the Fouha Bay site was made, and although plans for an expansion of those efforts had not yet been formalized, there was an expectation that such an expansion would eventually occur. Fouha Bay was also a site for which some historical data existed, and in 2014 was the site of a NOAA PIRO-funded, GLTMP-supported study to estimate the potential movement of more diverse benthic communities towards the head of the bay with a decrease in sediment load (see Minton et al. 2022). Due to the extremely low coral cover, the generally poor condition of the benthic community, and the limited active management of the watershed it was determined that the Fouha Bay site was to be surveyed every three to five years. Since that determination was made, Dr. Bastian Bentlage of the UOGML has begun investigating the response of different aspects of the coral community to sediment stress in Fouha Bay, and has partnered with the GLTMP in the deployment and maintenance of GLTMP water quality monitoring instrumentation in the bay. This new focus on Fouha Bay may justify more frequent collection of biological data within the site.

Data collection within the Fouha Bay site occurs at a total of 13 permanent sampling stations, including a cluster of five along the relatively flat submarine terrace (5–7 m) in the outer portion of the northern side of the bay; a cluster of five along the flat submarine terrace in the outer portion of the southern side of the bay (5–7 m); and a cluster of three stations in a distinct, more structurally complex, habitat occurring closer to the head of the bay on the bay’s north side (~6 m depth; Fig. 44). The limited extent of reef habitat within the small bay constrained the number and length of transects that could be placed without resulting in overlapping survey areas. Low visibility and high reef complexity prevented the execution of comparable quantitative reef fish surveys at stations inside the bay, but qualitative species presence surveys have been conducted in the vicinity of these stations. A single permanently established island-wide bleaching response and recovery site (564) is located along the reef front approximately 200 m to the north of the bay. Fouha Bay does not host an RFMP site.

Baseline data collection occurred at the Fouha Bay HPRA site in 2015, with subsequent data collection efforts occurring in 2019 and 2021 (Appendix A). Benthic cover and macroinvertebrate abundance data was also collected at a shallow (5 m depth) site (564) along the reef front near Fouha Bay, including one close to the HPRA site boundaries, in 2013 as part of a UOGML-led bleaching response effort. The reef front site located close to the HPRA was re-visited in 2015 as part of an island-wide bleaching recovery assessment, in 2016 as part of an island-wide reef resilience assessment, in 2017 as part of the response to another major bleaching event, and was surveyed and marked with rebar in 2021.

The results of an analysis of available benthic cover, reef fish, and macroinvertebrate data obtained within the Fouha Bay HPRA in 2015 and 2019 are provided below; the analysis of benthic photo transect images obtained in 2021 were on-going at the time this report was being developed. The results of an analysis of data obtained at the nearby island-wide bleaching response and recovery site are presented beginning on p. 26 but are also referenced in the “Discussion” section below.

Benthic cover

Mean coral cover remained at 18% at the three transects located closer to the head of Fouha Bay (“Inner”) between 2015 and 2019, while it declined from the already extremely low 2% cover to 1.2% cover at the transects located at the mouth of the bay (“Outer”) during this period (Paired Wilcoxon Signed Ranks Test, p = 0.01; Fig. 45). The highly rugose reef within the bay is host to a *Porites*-dominated coral community, with *Porites rus* and mounding *Porites* spp. together comprising greater than 90% of total coral cover in both 2015 and 2019 (Figs. 46–47). As with the stability in total coral cover, the cover of individual coral groupings (e.g., mounding *Porites* spp., *Porites rus*, other *Porites* spp., and all other coral taxa) at the Inner transects did not change significantly between 2015 and 2019. *Porites* corals are generally considered stress tolerant, but the relatively low coral cover and observations of significant partial colony mortality and sediment accumulation suggests that the high sediment load frequently

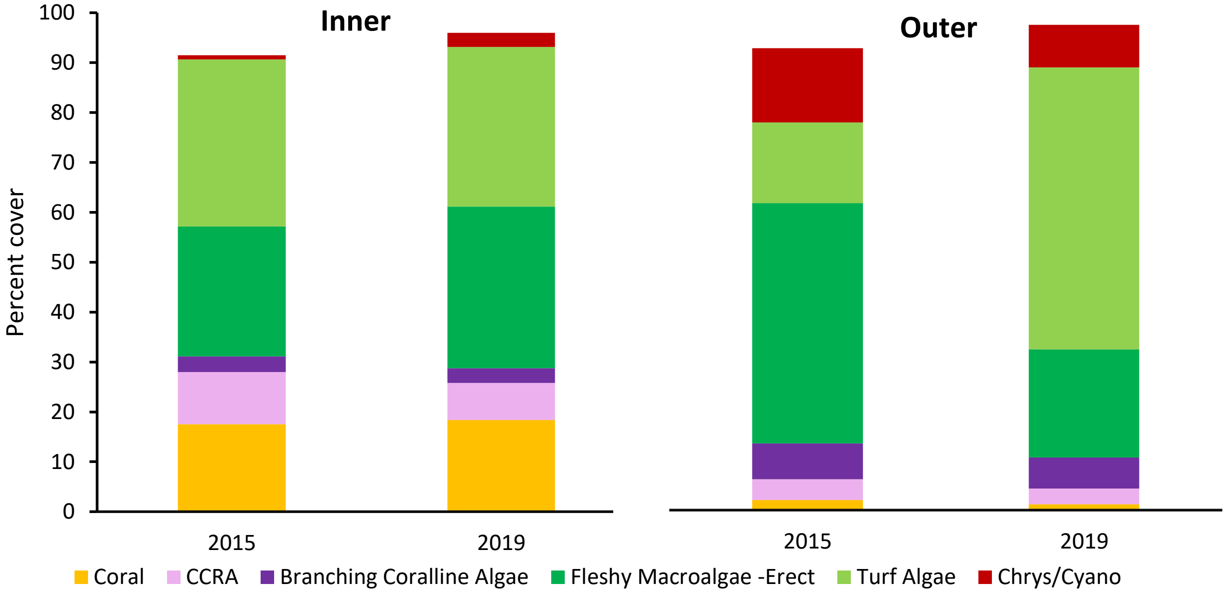


Figure 45. Percent cover of major benthic cover classes for the Fouha Bay HPRA in 2015 and 2019.

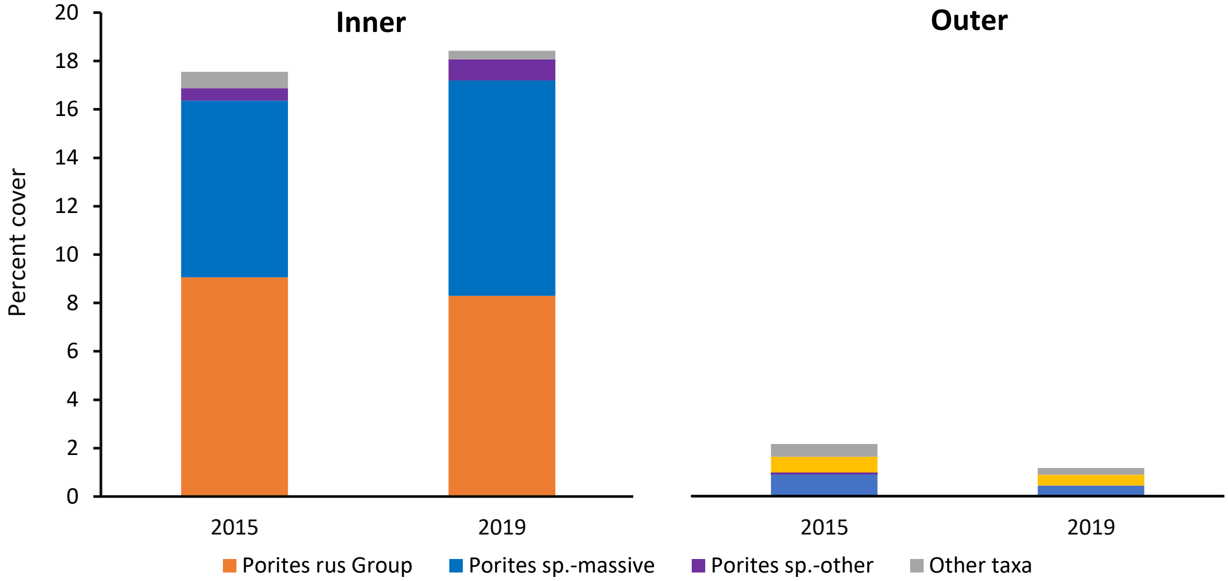


Figure 46. Percent cover of coral taxa for the Fouha Bay HPRA in 2015 and 2019.

delivered to the bay's waters by the La Sa Fua River is preventing net coral growth, even among the most stress tolerant taxa. In contrast to high degree of structural complexity of the reef within the bay, the reef at the mouth of the bay (where the "Outer" transects are located) is relatively flat on the upper surface, sloping steeply at the edge towards the sandy floor of the bay. More than half of community of scattered coral colonies in this area was comprised of coral taxa other than *Porites* (mainly *Astreopora*, *Stylophora*, and *Pocillopora*) in both 2015 and 2019, although mounding *Porites* still comprised 41% and 36% of total cover in 2015 and 2019, respectively. Even though the absolute cover of mounding *Porites* spp. was very low, a statistically significant decline from 0.9% to 0.4% was detected between 2015 and 2019 (Partover.test, 0.03). No change was detected for other coral taxa at the Outer transects between 2015 and 2019.

Statistical comparisons were not made for non-coral benthic features through time at the Fouha Bay HPRA, but some notable changes may have occurred at both the Inner and Outer transects between 2015 and 2019 (Fig. 45). The cover of crustose coralline red algae at the Inner transects was relatively low, and may have declined from 10% to 7% during this period. Most of the decrease in CCRA was comprised of taxa within the Order Corallinales (the pink and red crusts), which may have declined from 10% to 3%. The cover of turf algae at the Inner transects remained between 32% and 34% between 2015 and 2019, while the cover of fleshy macroalgae may have increased from 26% to 32% and the cover of cyanobacteria and chrysophytes may have increased from 1 to 3%. The cover of CCRA was even lower at the Outer transects, remaining between 3% and 4%. The cover of fleshy macroalgae was quite high at the Outer transects, but may have decreased significantly (from 48% to 22%) between 2015 and 2019, while the cover of turf algae may have increased from 16% to 56% during this period. It should be noted, however, that most of the multi-species turf algae matrix observed in the Fouha Bay HPRA was substantial, reaching up to the 1 cm canopy height threshold, and was often difficult to discern from macroalgae during the image analysis. As such, the changes in the percent cover of turf algae and fleshy macroalgae at the Fouha HPRA could be primarily an artifact of the image analysis method, and may not represent an actual change in these benthic cover classes. In any case, the often substantial turf algae matrix at the Fouha Bay HPRA was observed to retain a large amount of sediment. This is in contrast to the greater abundance of microturf at the Tumon and East Agana HPRAs, which is barely visible and is not believed to negatively impact corals or other calcifying organisms. The persistence of a multi-species mat of turfing algae and fleshy macroalgae at the site is likely inhibiting coral recovery at the Fouha Bay HPRA, especially at the Outer transects, which experience much lower sediment loads than the Inner transects.

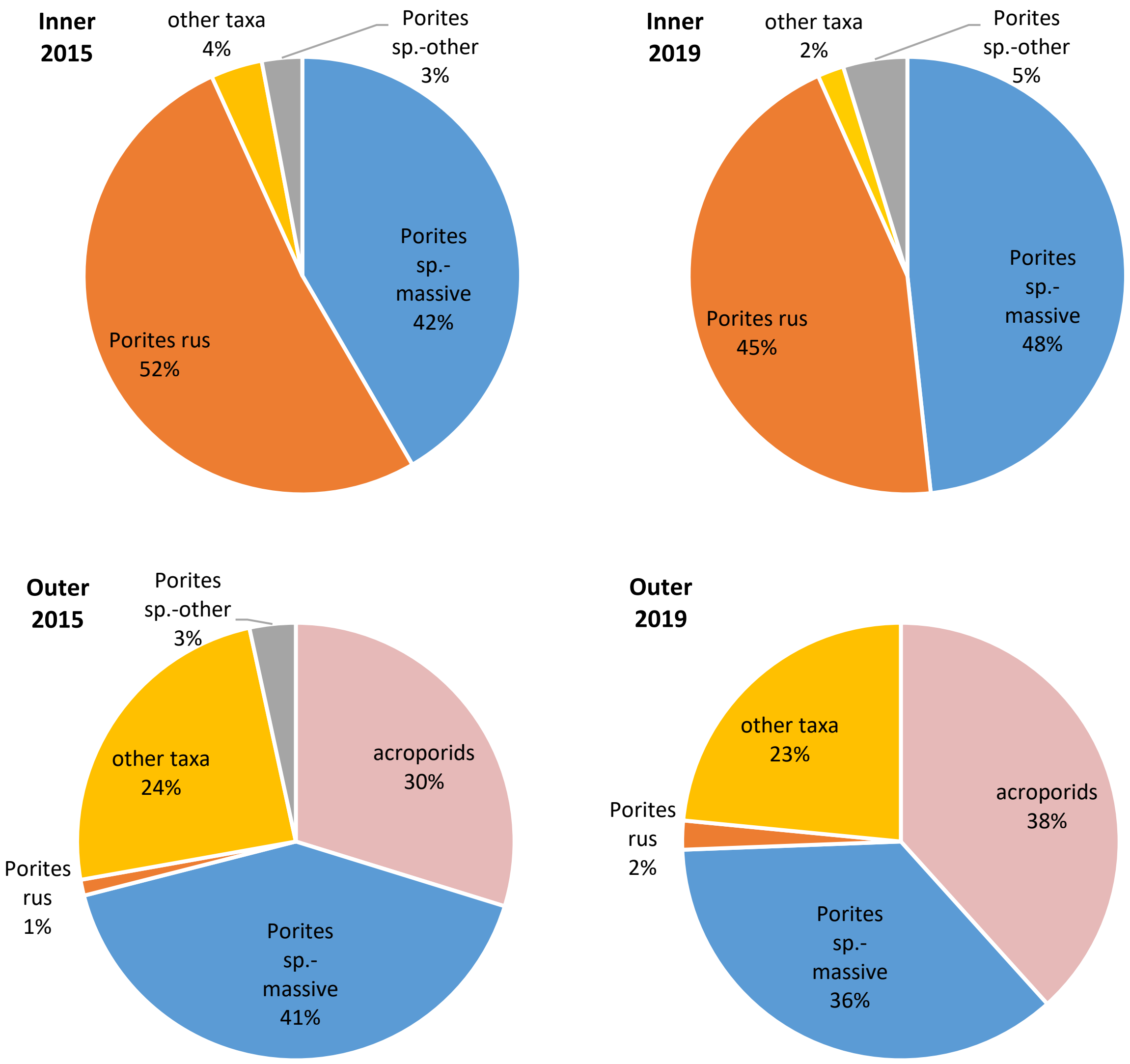


Figure 47. Percent contribution of individual coral taxa/taxa groupings to mean total coral cover for the Fouha Bay HPRA in 2015 and 2019.

Associated benthic communities

Reef fishes

Biomass

Mean food fish biomass within the Fouha Bay HPRA appeared to remain relatively stable, albeit very low, at around 3 to 4 g·m^{-2} between 2015 and 2021 (Fig. 48a). However, an examination of food fish biomass for the two individual observers suggests that this apparent stability is likely an artifact of interobserver bias, and that food fish biomass may have actually increased during this period. When Observer 1 and Observer 2 conducted surveys together in 2019 the differences in recorded food fish biomass was significant, with biomass reported by Observer 1 considerably higher than that reported by Observer 2 (Fig. 48b). Despite this significant interobserver bias, some value can still be gleaned from this preliminary analysis of food fish biomass data within the Fouha Bay HPRA. Mean food fish biomass as recorded by the more experienced observer (Observer 1) appeared to remain stable between 2015 and 2019 (Partover.test, p = 0.2). While the food fish biomass values recorded by the less experienced observer

(Observer 2) were significantly lower than those recorded by Observer 1 when they conducted surveys together in 2019, food fish biomass appeared to increase substantially (by 150%, from 1.7 ± 1 g·m^{-2} to 4.2 ± 1.2 g·m^{-2}) between 2019 and 2021 (Partover.test, p = <0.005).

Total fish biomass appeared to remain relatively stable between 2015 (19 ± 11 g·m^{-2}) and 2019 (20 ± 14 g·m^{-2}), but was very low (Fig. 48c,d). This stability in total fish biomass between 2015 and 2019 still holds when only observations obtained by Observer 1 are considered (Partover.test, p = 0.18). Non-food fish were not counted and sized beginning in 2020, so a total fish biomass value cannot be reported for the Fouha Bay HPRA for 2021. The substantial difference in the mean total fish biomass values recorded by Observer 1 and Observer 2 necessitates careful consideration of interobserver bias when data obtained by future surveys are compared to the older survey data. The total fish biomass of around 3 to 8 g·m^{-2} reported for the Fouha Bay HPRA between 2015 and 2019 is approximately 15% or less of the potential total reef fish biomass value (53 ± 7 g·m^{-2}) estimated for an unimpaired Guam reef community by Williams et al. (2015).

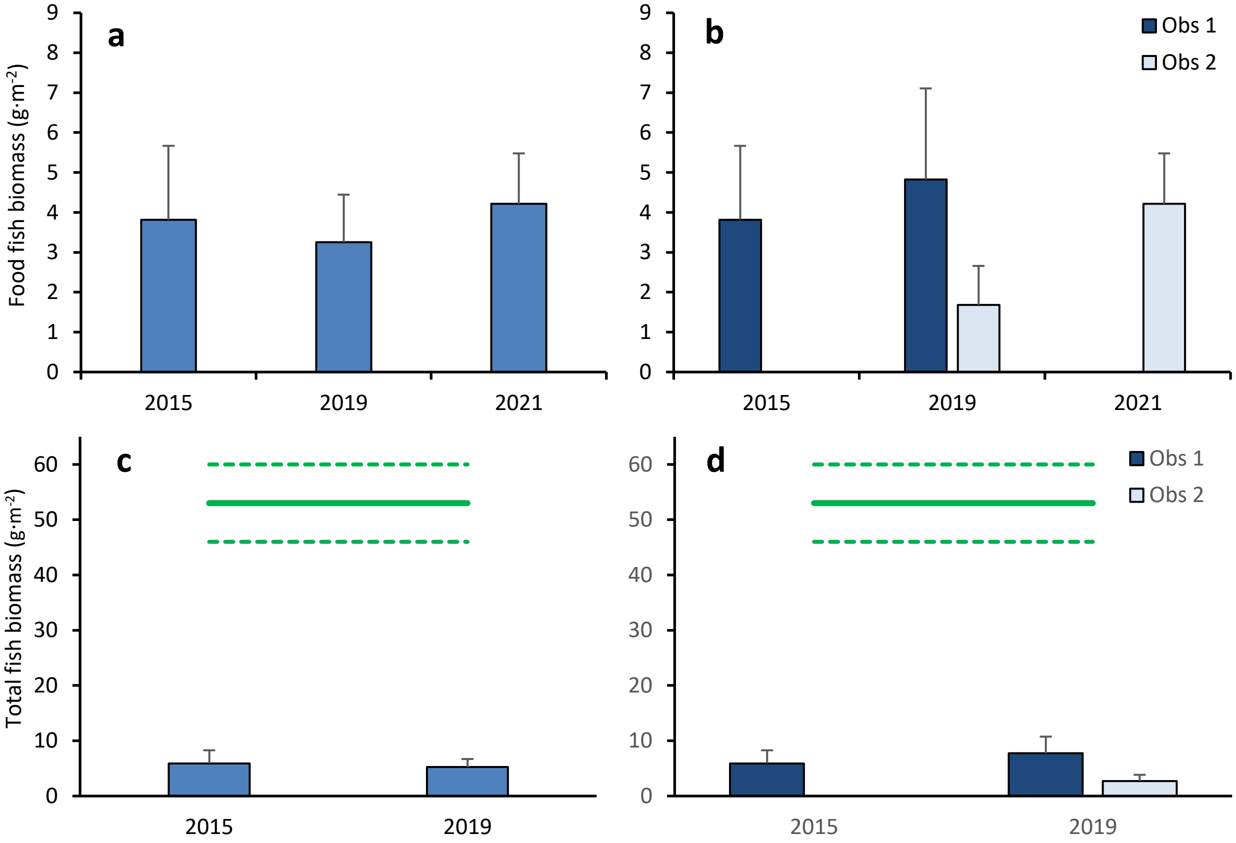


Figure 48. Food fish biomass (a, b) and total fish biomass (c, d) within the Fouha Bay HPRA between 2015 and 2021 and between 2015 and 2019, respectively, with data from both observers combined (a, c) and presented separately (b, d). The green lines indicate the potential total reef fish biomass for an unimpaired Guam reef community (53 ± 7 g·m^{-2}) estimated by Williams et al. (2015).

Due to the significant interobserver bias discussed above, biomass values for selected food fish families within the Fouha Bay HPRA were calculated using only those observations recorded by Observer 2 in 2019 and 2021 (Fig. 49). While statistical comparisons were not carried out for this limited preliminary analysis, the biomass of surgeonfishes and “other families,” which includes triggerfishes, squirrelfishes, and other food fish families, may have increased between 2019 and 2021. However, the biomass of these groups and others remained very low during this period. No emperors or snappers were detected by Observer 2 in either 2019 or 2021.

Density

The mean density of smaller (≤ 20 cm) food fishes within the Fouha Bay HPRA appeared to decrease from 17 ind·100 m^{-2} in 2015 to 14 ind·100 m^{-2} in 2019, but then increased to 21 ind·100 m^{-2} in 2021 (Fig. 50). The density of smaller food fishes does not appear to be affected substantially by interobserver bias, thus this apparent change likely reflects an actual increase in the number of smaller food fishes within the Fouha Bay HPRA between 2015 and 2021. The density of moderately sized (21 ≤ 40 cm) food fishes remained very low 2015 and 2019; no larger (> 40 cm) food fishes were detected during this period.

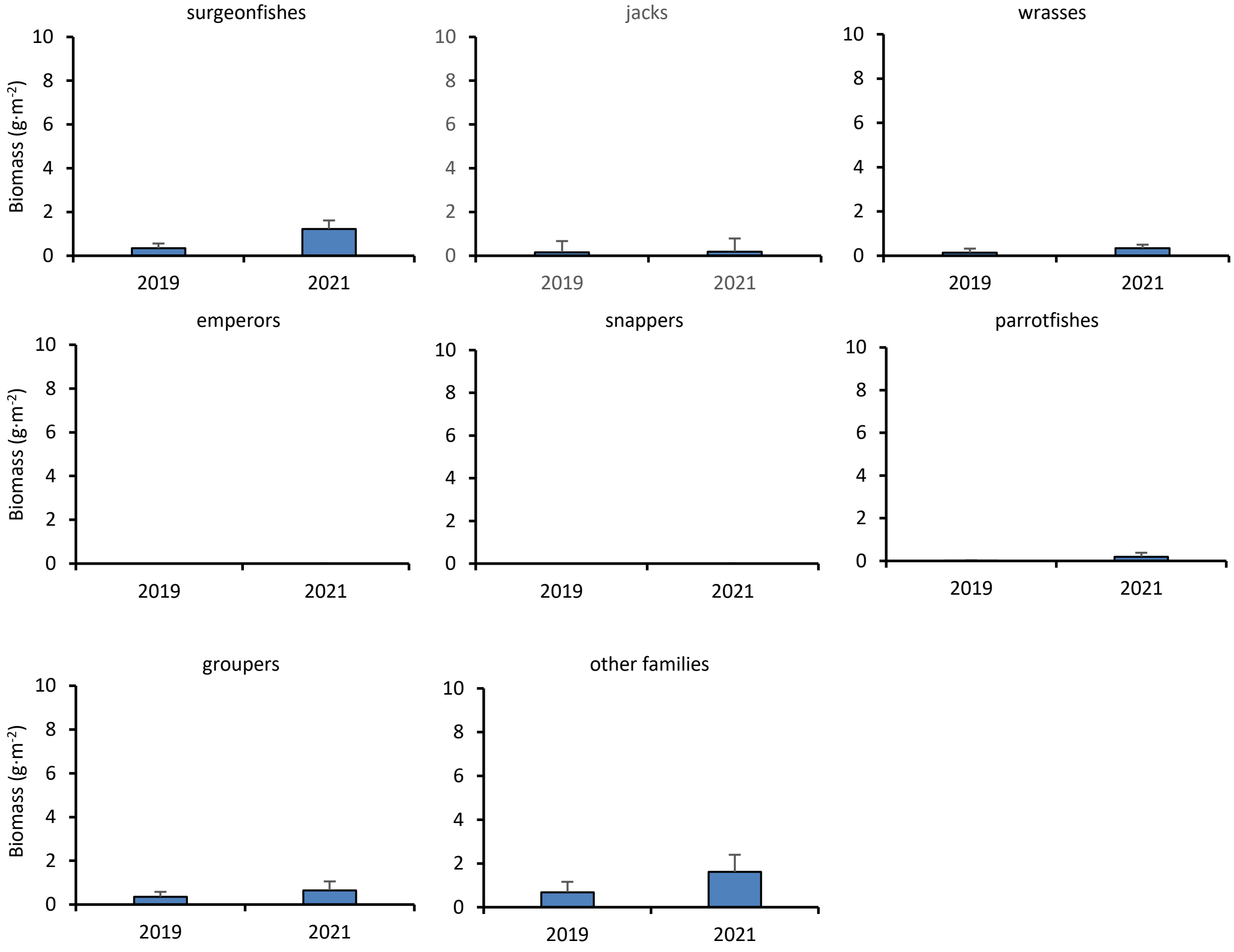


Figure 49. Biomass (g·m^{-2}) of selected food fish families within the Fouha Bay HPRA in 2019 and 2021. Due to the significant interobserver bias detected early in the analysis, these values were calculated using only those observations collected by Observer 2, who collected data for the two most recent survey years.

Species Richness

The total number of species recorded within the Fouha Bay HPRA across all sampling stations appeared to decline from 101 species in 2015 to 74 species in 2021 (Fig. 50). Mean species richness also appeared to have declined from 36 ± 9 species per sampling station in 2015 to 30 ± 4 species per sampling station in 2021. However, as with total fish biomass, the apparent declines in total and mean species richness are very likely artifacts of interobserver bias, as well as influenced by differing levels of sampling effort, rather than actual declines. For example, while the total number of species recorded by Observer 1 declined from 101 species in 2015 to 83 species in 2019, and mean species richness declined from 36 ± 9 in 2015 to 30 ± 5 in 2019, this observer conducted both survey replicates at all 13 sampling stations in 2015, but only one replicate for all stations in 2019 (i.e., half the survey effort in 2019 compared

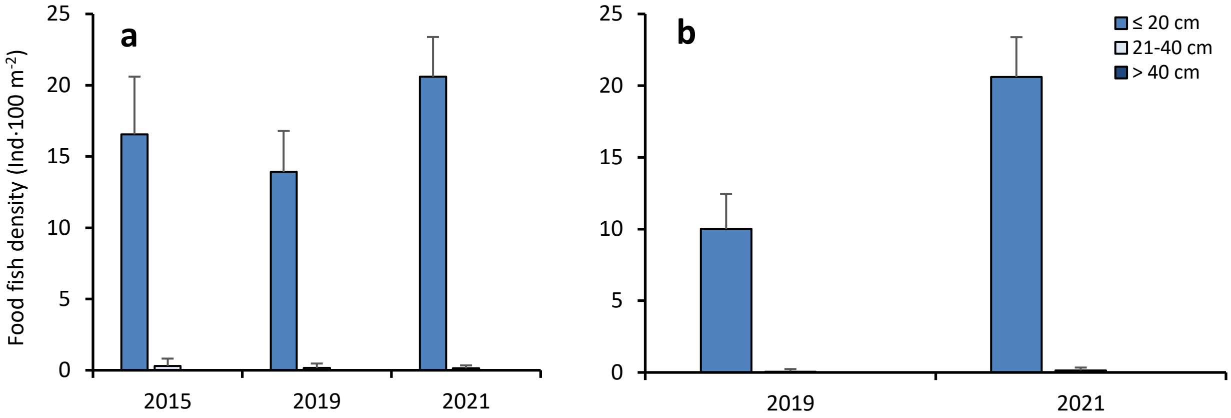


Figure 50. Mean food fish density (ind·100 m^{-2} ± SD) by size class within the Fouha Bay HPRA between 2015 and 2021, with data from two observers combined (a) and with only data from Observer 2 (b).

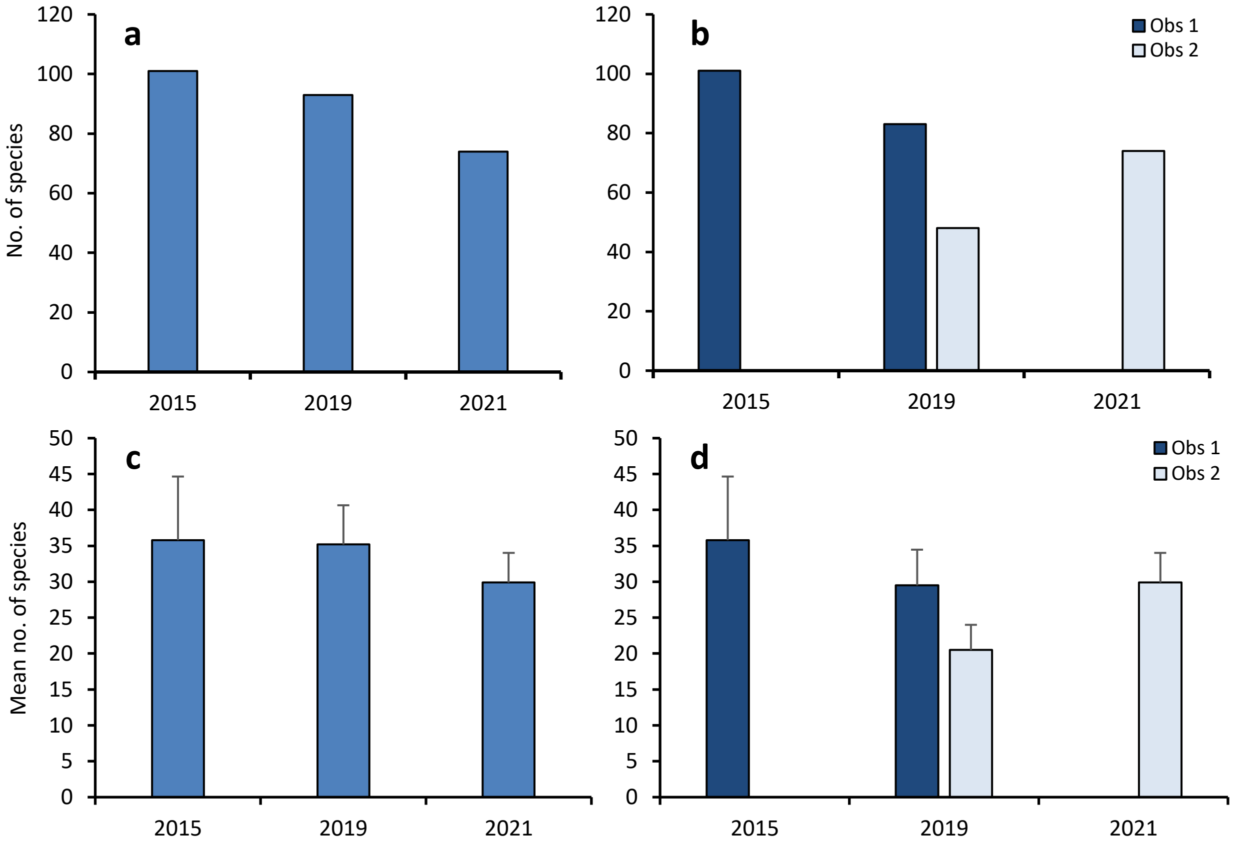


Figure 51. Total (a, b) and mean (c, d) species richness within the Fouha Bay HPRA between 2015 and 2021, with data from all observers combined (a, c) and presented separately (b, d). Note that the time interval between survey years varied.

to 2015). It is likey that Observer 1 would have detected additional species with additional survey effort in 2019. Similarly, the increase in the total number of species observed by Observer 2 between 2019 and 2021 (from 48 to 74 species), and the increase in mean species richness (from 21 ± 4 in 2019 to 30 ± 4 in 2021) are also likely related to sampling effort, as this observer conducted twice as many fish survey replicates in 2021 in comparison to 2019. A more detailed analysis of species counts obtained by the different observers is likely to reveal species that are consistently detected by one observer but not another, or other biases that could be minimized with targeted training and calibration.

Macroinvertebrates

The densities of all surveyed macroinvertebrate groups were quite low within the Fouha Bay HPRA across all survey periods, but some notable changes were still detected (Figs. 52–53). The mean density of edible shells, comprised almost entirely of giant clams (*Tridacna* spp.), decreased by 94% (from 1.2 to 0.1 ind·100 m^{-2}) between 2015 to 2019 (Partover.test, p = 0.02), before increasing to 0.7 ind·100 m^{-2} in 2021 (Partover.test, p = 0.02). It is not clear what

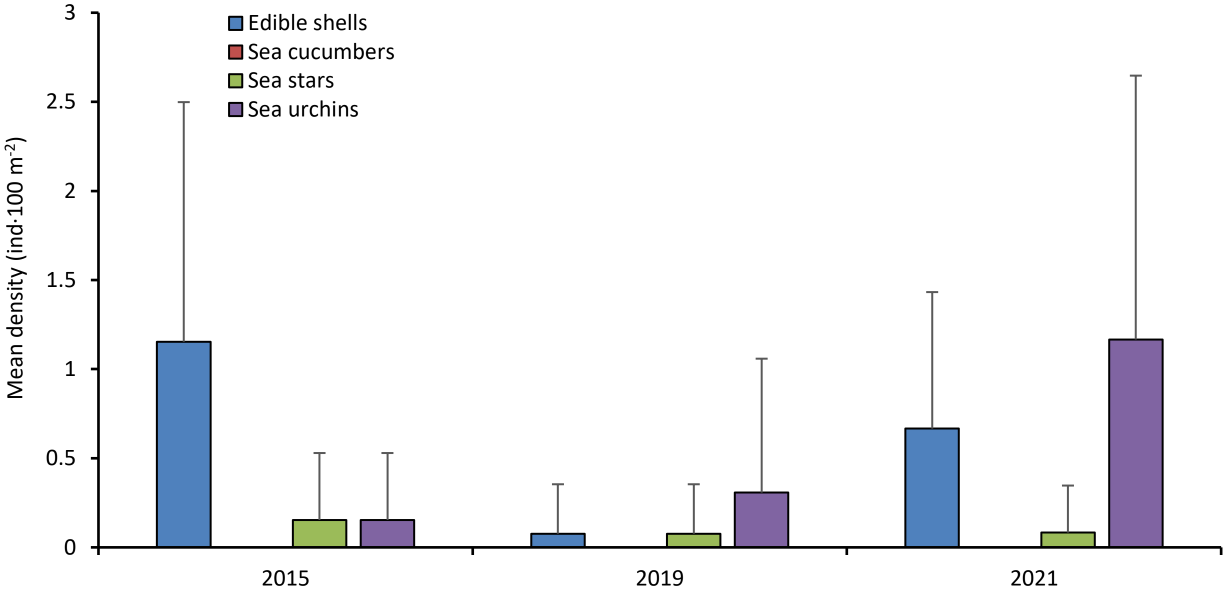


Figure 52. Mean density (ind·100 m^{-2}) of broad macroinvertebrate groups within the Fouha Bay HPRA between 2015 and 2021.

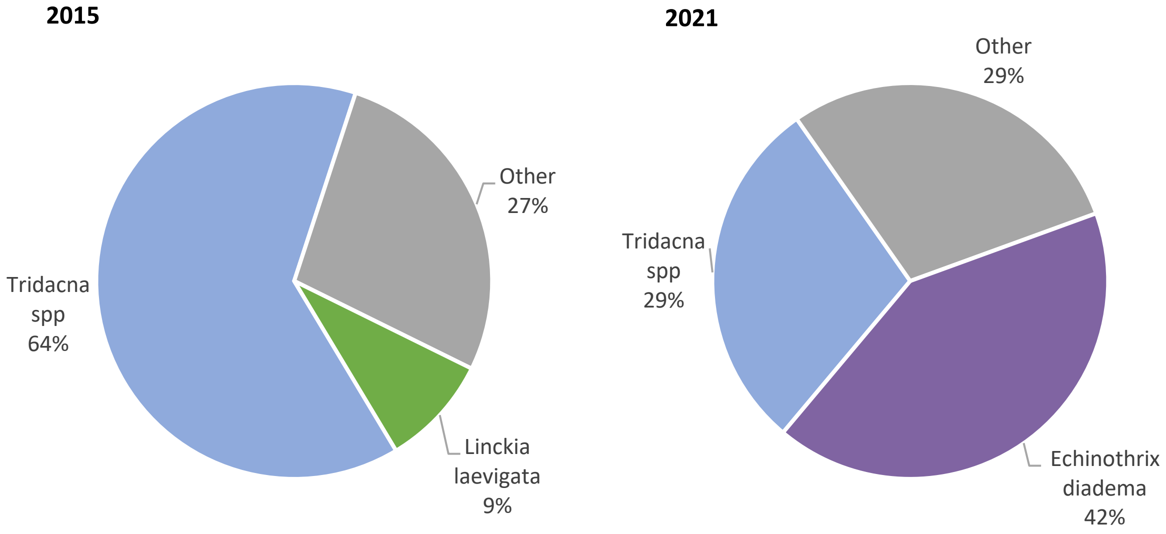


Figure 53. Relative abundance of individual macroinvertebrate taxa within the Fouha Bay HPRA in 2015 and 2021.

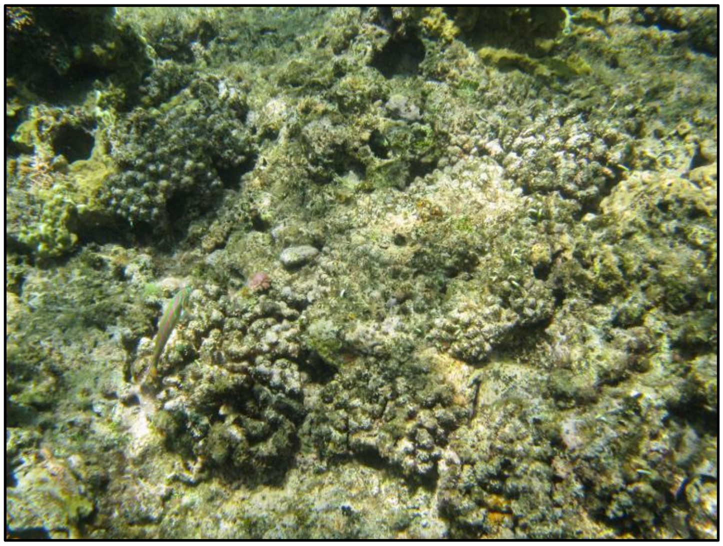
drove the decline in giant clam density between 2015 and 2019, although the historically severe 2017 thermal stress event may have played at least some role [*Tridacna* host symbiotic zooxanthellae and are known to bleach]. The density of large sea urchins appears to have increased significantly (from 0.2 to 1.2 ind·100 m^{-2}) between 2015 and 2021 (Partover.test, $p = 0.01$), while the density of large sea stars, although very low, remained stable during this period. No sea cucumbers were observed within the Fouha Bay HPRA during any survey year.

Discussion

Coral cover at the more structurally complex, primarily *Porites*-dominated transects inside Fouha Bay was low-to-moderate (18%) between 2015 and 2019, remaining relatively stable during this period despite the severe bleaching event in 2017 and the chronic levels of severe sediment stress in the bay. The impacts of sediment stress, such as sediment accumulation on coral colonies and extensive partial colony mortality, were evident, however, and only the most sediment-tolerant coral taxa can persist in this part of the bay. The relatively low diversity, *Porites*-dominated coral community within the bay, while currently fairly resilient to various stressors, will likely struggle as sea surface temperatures continue to increase, and the frequency and severity of bleaching events increases in response. The dominance of the coral community by a small number of *Porites* species may also make it more vulnerable to devastation by diseases.

Coral cover at the relatively flat submarine terrace at the mouth of the bay, which hosts a greater proportion of more stress-susceptible, non-*Porites* corals, was extraordinarily low (2%) in 2015, and declined to 1.2% by 2019, potentially due to bleaching, predation by COTS, sediment stress, or a combination of all of these stressors. Historical data for the reef community at the mouth of the bay is limited, but it is likely that acroporids and other less stress-tolerant coral taxa were more abundant in this area in the past, and their higher growth rates would have allowed them to outcompete the slower-growing mounding *Porites* species. The extremely low coral cover at the mouth of the bay is likely the result of past acute disturbances, such as COTS outbreaks and severe sedimentation associated with the construction of the coastal highway. The persistence of this low coral cover condition is likely enabled by chronically poor water quality, continued low-levels of predation by COTS, depauperate herbivore populations, bleaching-associated mortality, and possibly other sources of stress and impaired recovery processes. The benthic community in this area appears to have shifted to an alternate stable state, one dominated by a sediment-laden algal matrix rather than by corals and crustose coralline algae. Even if water quality conditions and herbivore populations increase, natural recovery may be slow or absent, and may require human intervention to prepare the benthic substrate for coral and crustose coralline algae recruitment and growth.

While an in-depth analysis of site-level changes in the coral community at individual island-wide bleaching response and recovery sites has not yet been conducted, coral cover at the site (564) located along the reef front (5 m depth) just to the north of the bay appears to have declined by about 70%, from 14% to 4%, between 2013 and 2017, very likely as a result of bleaching impacts during this period. However, coral cover at the site returned to 14% by 2021. Further analysis will show how the coral community may have shifted during this period. The apparent recovery of coral cover at the reef front near the bay, in contrast to the further decline of the already-low coral cover at the mouth of the bay, may have been facilitated, at least in part, by the effect of wave energy on the benthic community (e.g., more crustose coralline algae, less macroalgae, and fewer COTS) at the shallower depth, and by the distance from the sediment-laden water



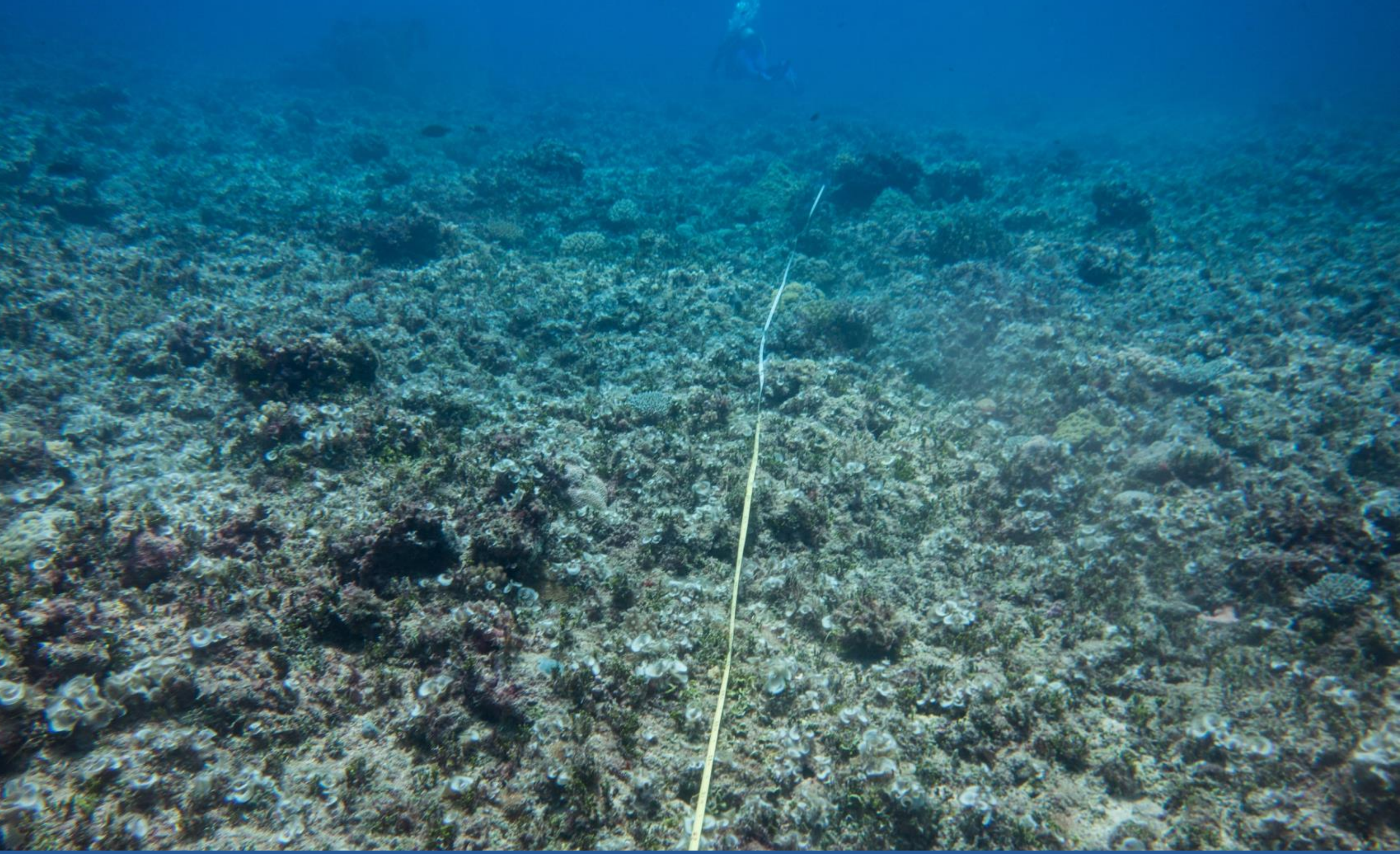
Skeletons of an undescribed Acropora species in the shallow (<3 m) reef front near the mouth of Fouha Bay. Numerous Acropora, Pocillopora, and likely other stress-susceptible coral taxa only found in this shallow, wave-exposed part of the reef, suffered significant mortality as a result of the bleaching events between 2013 and 2017.

discharged from the bay. Anecdotal observations made by the MPC along the shallow (< 3 m) reef front near the mouth of Fouha Bay in May 2014, about 6 months after the 2013 bleaching events, suggest a substantial loss of *Pocillopora* spp. and *Acropora* spp. once common in the area; however, he has not returned to that specific area to see if recovery had occurred.

Even in consideration of data limitations and interobserver bias, it is reasonable to conclude that food fish biomass was exceptionally low within the Fouha HPRA between 2015 and 2021. While a more detailed analysis is required to adequately address the data and staff capacity limitations, there are also indications that food fish biomass increased between 2015 and 2019, while the density of small food fishes may have declined during this period. The reef fish survey data obtained by a single observer since 2019 allows for more confidence in the assertion that total food fish biomass increased within the Fouha HPRA between 2019 and 2021, and that the density of small food fishes also increased during this period. While statistical comparisons have not yet been made for individual food fish families through time, the biomass of surgeonfishes and "other families" (which includes triggerfishes, squirrelfishes, and other food fish families), may have increased between 2019 and 2021, while the biomass of all other food fish families appears to have remained relatively stable. Total fish biomass in the Fouha HPRA in 2019 was very low, at 6–15% of the potential total reef biomass estimated for an impaired Guam reef by Williams et al. (2015). The very low fish biomass within the Fouha HPRA can be at least partially explained by the very low degree of reef complexity, and resulting limited fish habitat availability, at the "Outer" transects where reef fish count and size data were obtained. However, poor habitat condition and the harvesting pressure are likely also partly responsible for these low values. It is likely that fish biomass and density would be higher at the "Inner" transects, which occur in an area with considerably greater structural complexity. Count and size data have not yet been obtained at the "Inner" transects due to low visibility conditions, but an attempt will be made to carry out SPC surveys at these transects in the future. Further study may better resolve any trends in the reef fish community that may have occurred within the Fouha Bay HPRA over the last decade, and may provide an indication of the likely causes of any declines or improvements in reef fish community condition.

The decline in edible shells (mainly giant clams) within the Fouha Bay HPRA between 2015 and 2019 may be a result of impacts associated with the severe 2017 heat stress event, as giant clams (*Tridacna* spp.) host symbiotic dinoflagellates and are known to be susceptible to bleaching and bleaching-associated mortality. Edible shell density appeared to recover, at least partially, with respect to 2015 densities. However, edible shell densities may have been higher prior to the heat stress events in 2013 and 2014, and thus edible shell density in 2021 may represent a smaller proportion of pre-2013 densities than of the density recorded in 2015. The complete absence of sea cucumbers within the Fouha Bay HPRA transects in any survey year is notable, but it's not clear if this represents a significant change compared to historic sea cucumber densities, and thus may be related to an increase in the deposition and retention of terrestrially derived sediments or some other stressors, or if there are other explanations for their absence.

Next page: *A GLTMP biologist investigates a high-relief, Porites-dominated reef near the center of Fouha Bay, in the vicinity of the HPRA's "Inner" transects, in 2021 (top), and a low-rugosity, very low coral cover, algae-dominated benthic community at one of the permanent transects on the submarine terrace at the outer portion of the bay in 2019 (bottom).*



ACHANG REEF FLAT MARINE PRESERVE

Key findings

- Coral cover was very low (~5%) in 2014 and 2018, but may have increased slightly during this period
- Despite the severe coral bleaching event in 2017, the cover of the generally stress tolerant mounding *Porites* corals, which comprised about half of total coral cover, remained stable between 2014 and 2018; the cover of the faster growing, but generally stress susceptible acroporids corals, increased slightly
- Data and staff capacity limitations currently prevent detailed analyses of reef fish survey data obtained between 2014 and 2021, but there are some indications that the generally low food fish biomass persisted between 2014 and 2018 and increased between 2018 and 2021
- The biomass of surgeonfishes, wrasses, emperors, parrotfishes, and groupers may have increased between 2014 and 2021
- Total fish biomass in 2018 was relatively low (9–23 g·m^{-2}), at 17–43% of the potential total reef fish biomass estimated for an unimpaired Guam reef by Williams et al. (2015)
- The density of small food fishes may have increased between 2014 and 2021, but the density of moderate and large food fishes remained very low
- The density of sea cucumbers may have declined between 2014 and 2018, while the density of edible shells (mainly top shells and giant clams) increased significantly during this period

Above: *An algae-dominated benthic community along the submarine terrace of the Achang Reef Flat Marine Preserve in 2021. Coral cover across the Achang HPRA was very low (~5%) and the cover of cyanobacteria and sediment-laden algae were high, indicating that the benthic community at the site has been highly impacted by various stressors.*

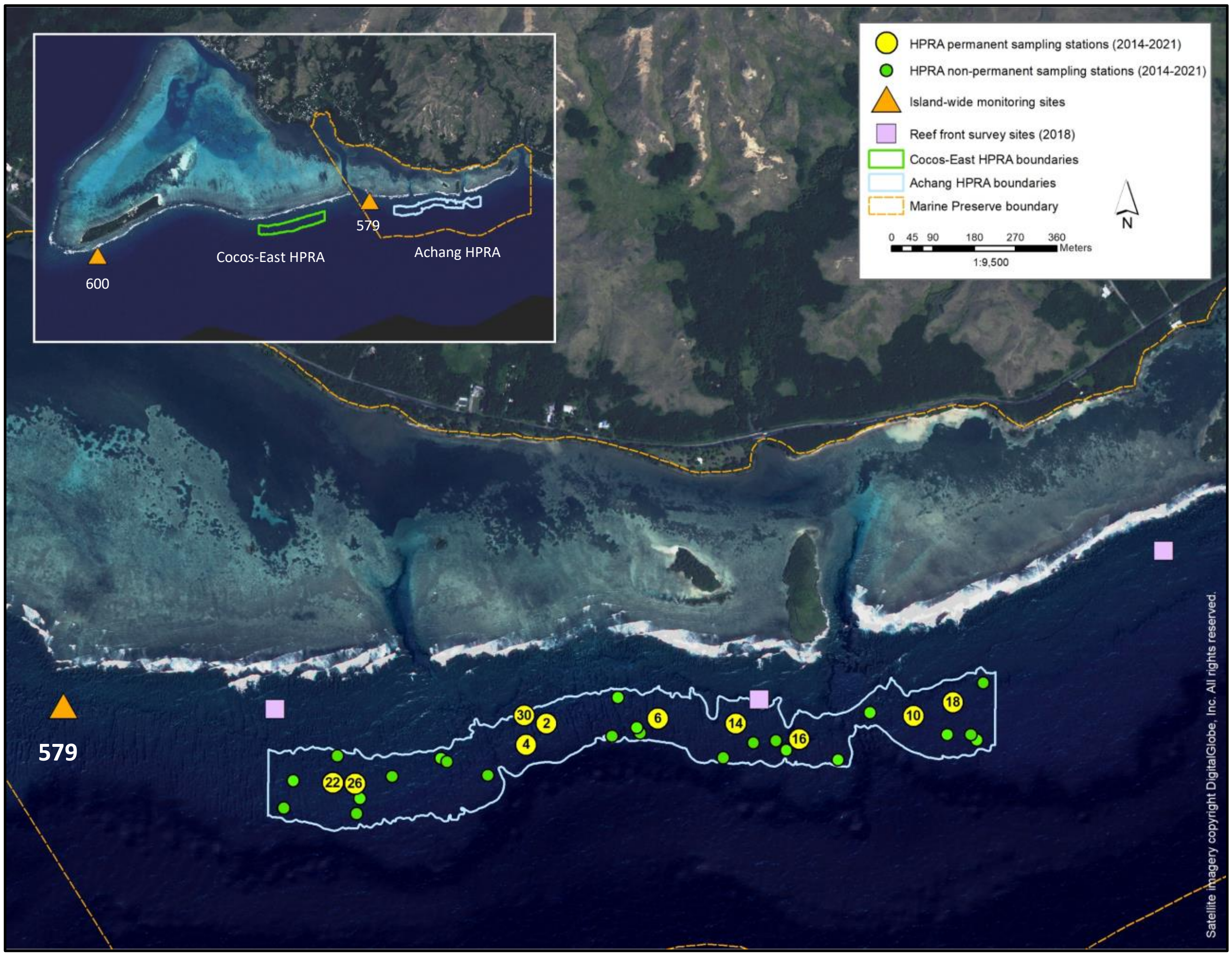


***Figure 54**.* Map of the Achang HPRA site boundaries and sampling stations and the island-wide bleaching response and recovery site in the vicinity of the HPRA.

Site overview

The Achang Reef Flat Marine Preserve, and an intended comparison non-preserve site along the seaward slope to the southwest of Manell Channel (Cocos-East), were established in 2014. These sites were selected primarily to serve as another potential preserve/non-preserve pair, but also because these two sites are influenced by poor water quality associated with the Manell River and smaller rivers in the Manell-Geus watershed unit. The Manell-Geus watershed unit was designated as a high priority watershed management area just prior to selection as monitoring sites. It has since been designated as a NOAA Habitat Blueprint site, and has been the target of several pilot watershed restoration and community outreach projects.

Data collection within the Achang HPRA occurs at a total of 10 permanent sampling stations and up to 10 non-permanent stations that are re-randomized each data collection year (Fig. 54). Eleven (11) permanent stations were initially targeted, but one of the stations was abandoned due to repeated encounters with dangerously strong currents. A single permanently established island-wide bleaching response and recovery site (579) is located along the reef front approximately 200 m to the west of the HPRA, next to the mouth of the Mannel Channel. No RFMP sites are located near the Achang HPRA.

Baseline data collection occurred at the Achang HPRA in 2014, with subsequent data collection efforts occurring in 2018 and 2021 (Appendix A). The original intent was to visit the Achang and Cocos-East HPRAs on a biennial basis, rather than annually, due to the challenge in regularly accessing these sites. Staffing limitations and the re-prioritization of resources required to carry out an island-wide reef resilience survey in 2016 prevented the collection of data that year, and the COVID pandemic prevented data collection at the site in 2020. Benthic cover and macroinvertebrate abundance data was also collected at three shallow (5 m depth) sites along the reef front in the vicinity of the Achang HPRA in 2013 as part of the UOGML-led bleaching response effort. One of the reef front sites (579) was re-visited in 2015 as part of an island-wide bleaching recovery assessment, in 2016 as part of an island-wide reef resilience assessment, in 2017 as part of the response to another major bleaching event, and was surveyed and marked with rebar in 2021. In 2018, benthic photo transects were opportunistically carried out at three sites along the reef front (5 m depth) along the Achang Marine Preserve, including two immediately upslope from the HPRA, with three 50 m transects surveyed at each site.

The results of an analysis of available benthic cover obtained within the Achang HPRA in 2014 and 2018, and reef fish and macroinvertebrate data obtained between 2014 and 2021 are provided below; the analysis of benthic photo transect images obtained in 2021 were on-going at the time this report was being developed. Data obtained at the nearby island-wide bleaching response and recovery site is presented on p. 26, but is also referenced in the "Discussion" section below.

Benthic cover

Mean coral cover remained very low within the Achang HPRA between 2014 and 2018, but appeared to increase slightly, from 5% to 6%, during this period (Partover.test, p = 0.004; Fig. 55). Nearly half of the coral community within the Achang HPRA was comprised of mounding *Porites* spp., with acroporids, merulinids, pocilloporids and other taxa comprising the other half (Figs. 56–57). The cover of acroporids increased slightly, from 0.4% to 0.9%, between 2014 and 2018 (Partover.test, p =< 0.01), but the cover of other taxa did not change during this period. The slight increase in coral cover in the Achang HPRA between 2014 and 2018, with a substantial portion of that increase comprised by an increase in acroporids despite the impacts of the bleaching events occurring between survey efforts, is encouraging. However, coral cover remained very low across the site and the limited amount of historical data and anecdotal observations suggest the cover of non-*Porites* coral taxa, especially *Montipora* spp. and *Acropora* spp., were likely greatly impacted by COTS well before baseline monitoring began at the site, and before the first severe bleaching event in 2013. It's also possible that the slight apparent increase in coral cover is an effect of the much lower cyanobacteria cover in 2018, which would result in fewer instances of cyanobacteria covering live coral.

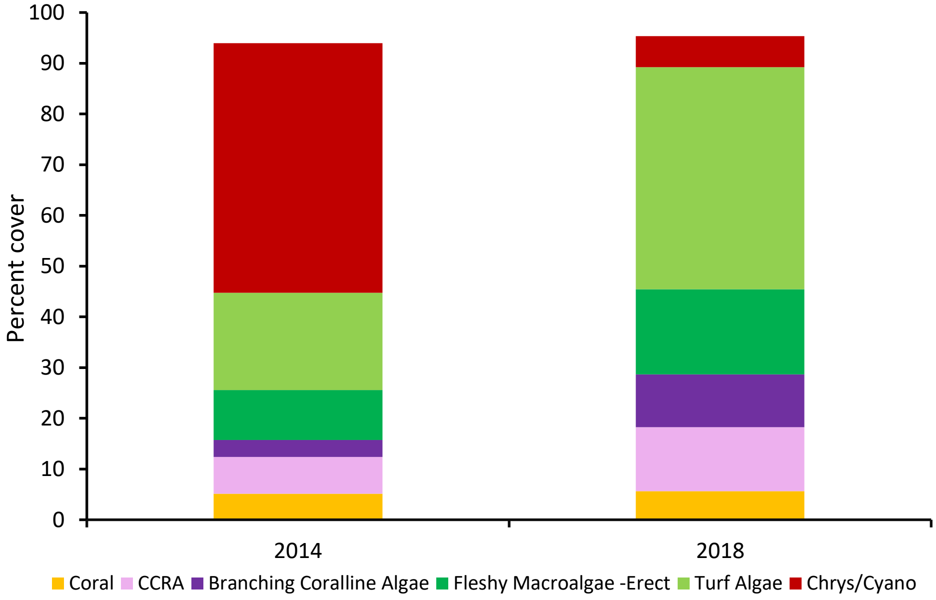


Figure 55. Percent cover of major benthic cover classes for the Achang HPRA in 2014 and 2018.

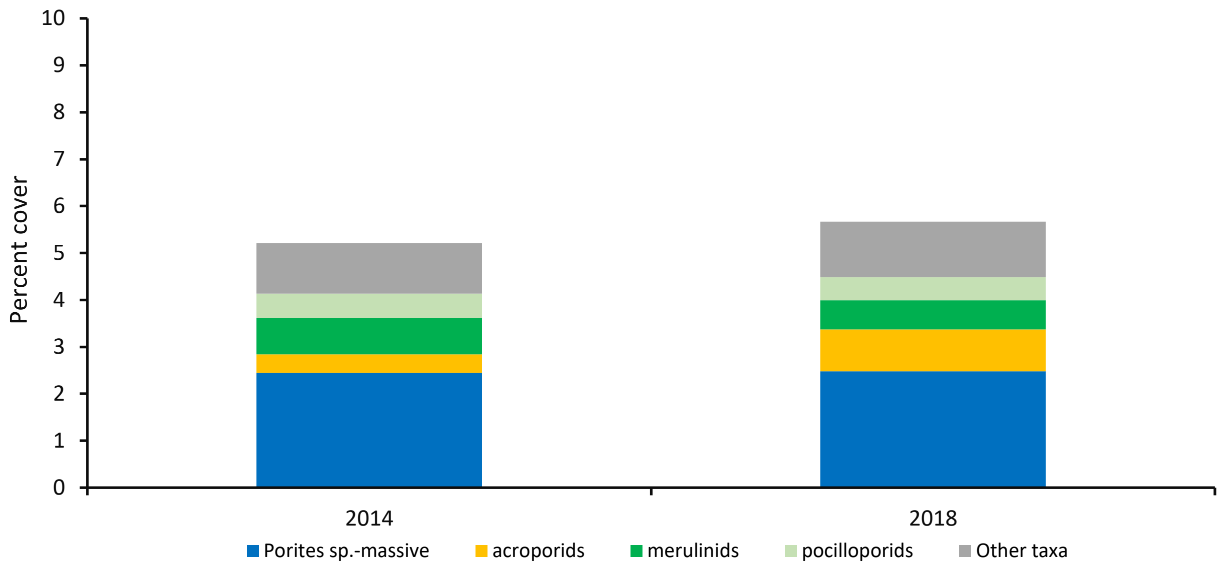


Figure 56. Percent cover of coral taxa for the Achang Reef Flat Marine Preserve monitoring site in 2014 and 2018.

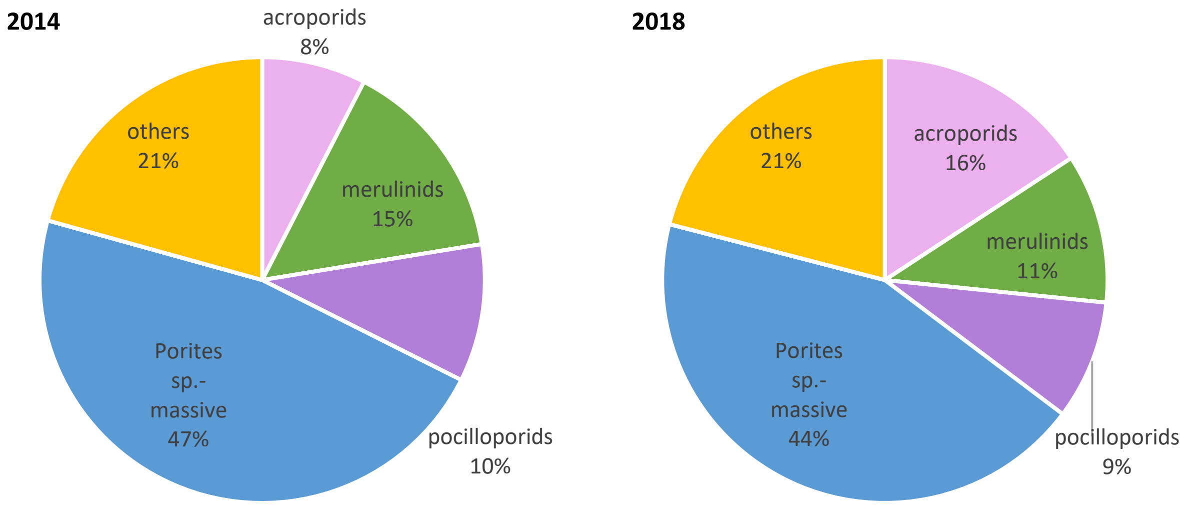


Figure 57. Percent contribution of individual coral taxa/taxa groupings to mean total coral cover for the Achang HPRA in 2014 and 2018.

Statistical comparisons were not made for non-coral benthic features through time at the Achang HPRA, but the cover of crustose coralline red algae, while low, may have increased from 7% to 13% during this period (Fig. 55). Most of the increase in CCRA was comprised of taxa within the Order Corallinales (the pink and red crusts), which may have increased from 5% to 9%. The cover of branching coralline algae, fleshy macroalgae, and turf algae appeared to all increase between 2014 and 2018, but this could be a result of a large decline in cyanobacteria and chrysophytes. Cyanobacteria and chrysophytes are generally more ephemeral than many benthic classes, and can quickly cover other benthic features before then dying back and revealing the benthic features living below.

Associated biological communities

Reef fishes

Biomass

Food fish biomass within the Achang HPRA appeared to decline from 16 ± 11 g·m^{-2} in 2014 to 11 ± 7 g·m^{-2} in 2021 (Fig. 58a). However, an examination of food fish biomass for the two individual observers during this period suggests that the apparent decline is likely an artifact of interobserver bias, rather than an actual decline in food fish biomass within the Achang HPRA. When Observer 1 and Observer 2 conducted surveys together (2014 and 2018) the differences in recorded food fish biomass were significant, with biomass reported by Observer 1 about three times, on average, higher than that reported by Observer 2 (Fig. 58b). Despite this significant interobserver bias, some value can still be gleaned from this preliminary analysis of food fish biomass data within the Achang HPRA. Mean food fish biomass as recorded by the more experienced observer (Observer 1) appeared to remain stable between 2014 (23 ± 19 g·m^{-2}) and 2018 (24 ± 24 g·m^{-2}). While the food fish biomass values recorded by the less experienced observer (Observer 2) were significantly lower than those recorded by Observer 1 during a given sampling period, these values were relatively consistent across years (~8–11 g·m^{-2}). While this limited analysis did not involve statistical comparisons, it is interesting to note that when using only those observations collected by Observer 2 food fish biomass may have actually increased from 8 ± 5 g·m^{-2} in 2014 to 11 ± 6 g·m^{-2} in 2021, a 38% increase.

Total fish biomass, which includes the biomass of both food and non-food fishes, appeared to remain stable between 2014 (19 ± 11 g·m^{-2}) and 2018 (20 ± 14 g·m^{-2}; Fig. 58c,d). Non-food fish were not counted and sized beginning in 2020, so a total fish biomass value cannot be reported for the Achang HPRA for 2021. The total fish biomass of 9–23 g·m^{-2} reported for the Achang HPRA by the two observers in 2018 is 17–43% of the potential total

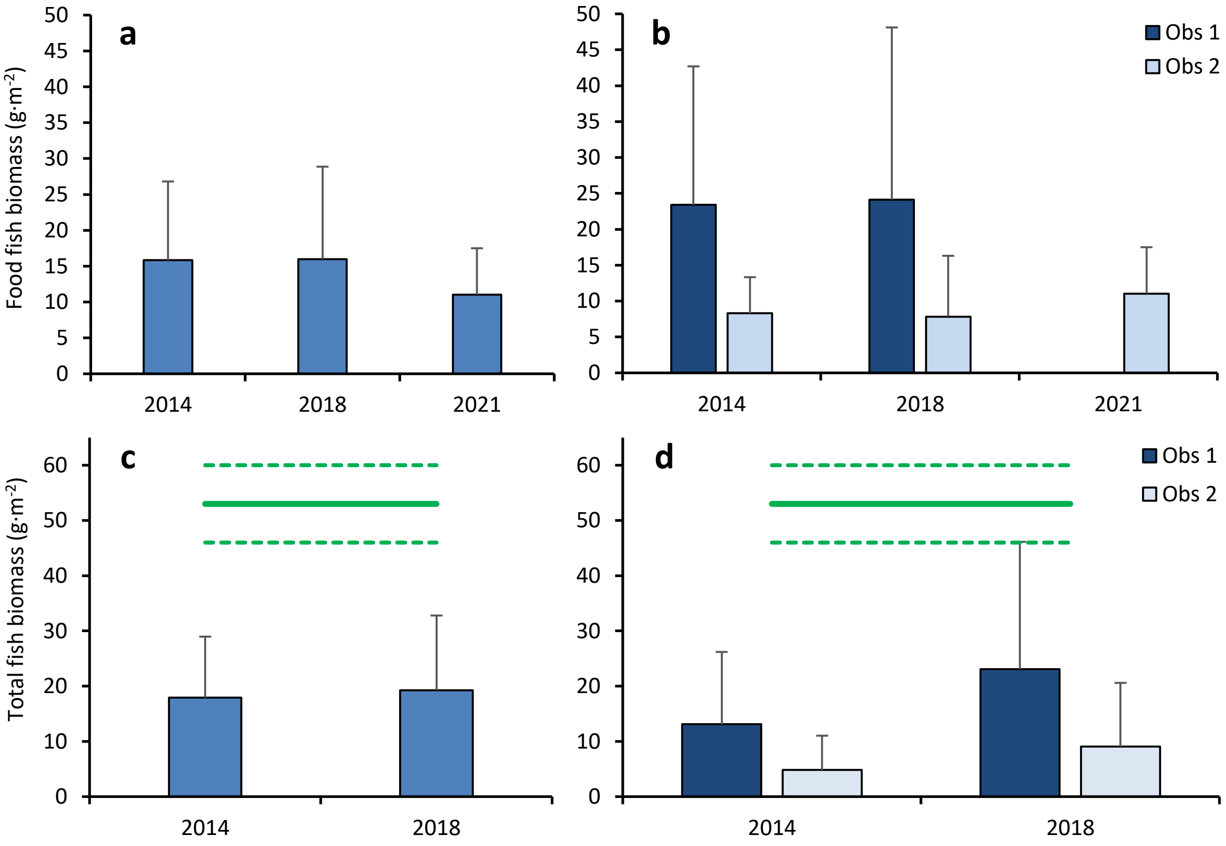


Figure 58. Food fish biomass (a, b) and total fish biomass (c, d) within the Achang HPRA between 2014 and 2021 and between 2014 and 2018, respectively, with data from both observers combined (a, c) and presented separately (b, d). The green lines indicate the potential total reef fish biomass for an unimpaired Guam reef community (53 ± 7 g·m^{-2}) estimated by Williams et al. (2015).

reef fish biomass value (53 ± 7 g·m^{-2}) estimated for an unimpaired Guam reef community by Williams et al. (2015).

Due to the significant interobserver bias discussed above, biomass values for selected food fish families within the Achang HPRA between 2014 and 2021 were calculated using only those observations recorded by Observer 2, who participated in surveys across all three sampling periods (Fig. 59). While statistical comparisons were not carried out for this limited analysis, the biomass of surgeonfishes, wrasses (food fish species only), emperors, parrotfishes, and groupers appear to have increased between 2014 and 2021, although apparent increases in emperor and parrotfish biomass may not be statistically significant due to the high degree of variability across sampling stations in 2021. The biomass of jacks was very low in 2014 and 2018, and was zero in 2021. The biomass of snappers was also relatively low, but appeared to remain relatively stable between 2014 and 2021. The combined biomass of all other food fish families also appeared to remain relatively stable during this period.

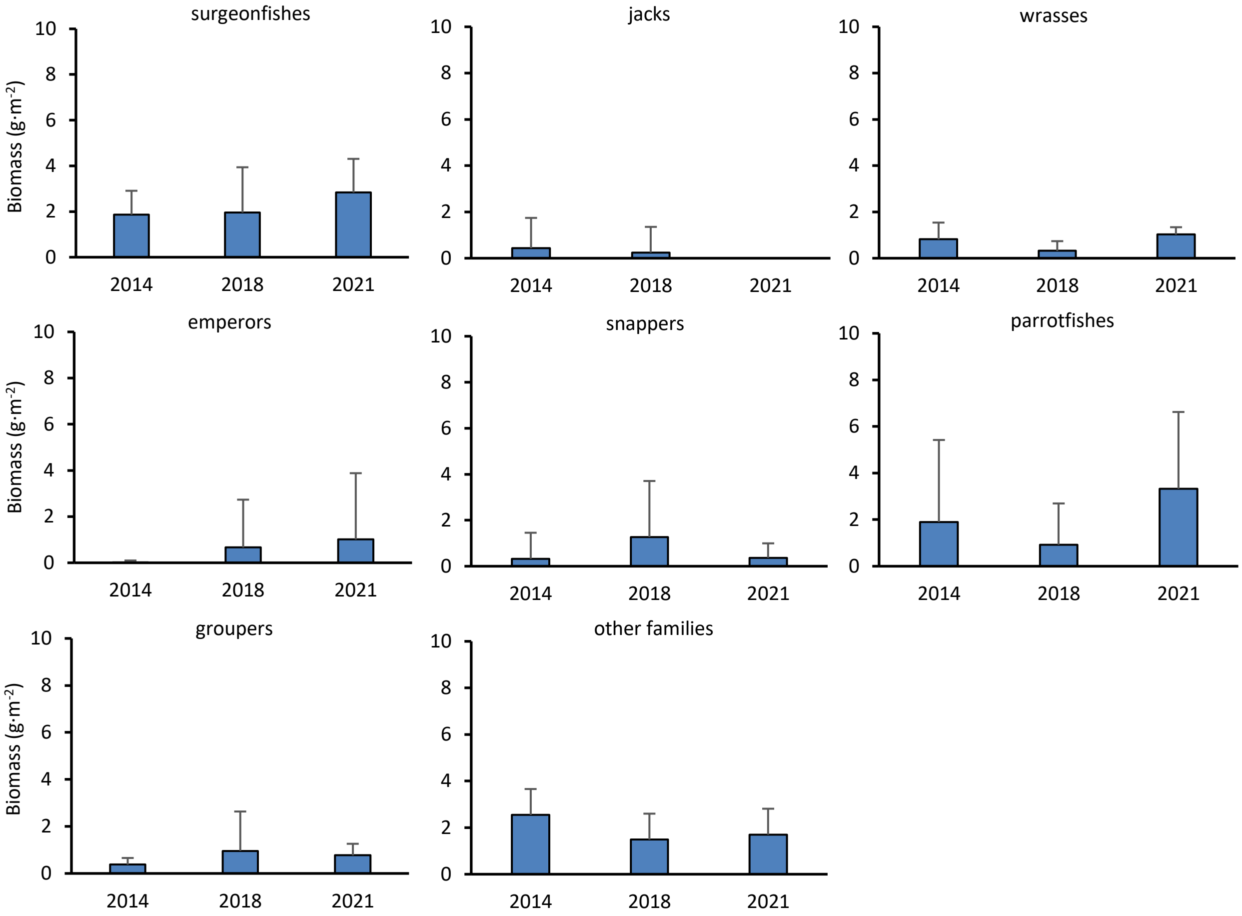


Figure 59. Biomass (g·m^{-2}) of selected food fish families within the Achang HPRA between 2014 and 2021. Due to the significant interobserver bias detected early in the analysis, these values were calculated using only those observations collected by Observer 2, as this observer carried out surveys all years.

.

Density

The density (ind·100 m^{-2}) of smaller (≤ 20 cm) food fishes appeared to have increased between 2014 and 2021, but the density of moderately sized (21 ≤ 40 cm) food fishes was very low and appeared to remain stable during this period (Fig. 60).

Species Richness

The total number of species recorded within the Achang HPRA across all sampling stations appeared to decline from

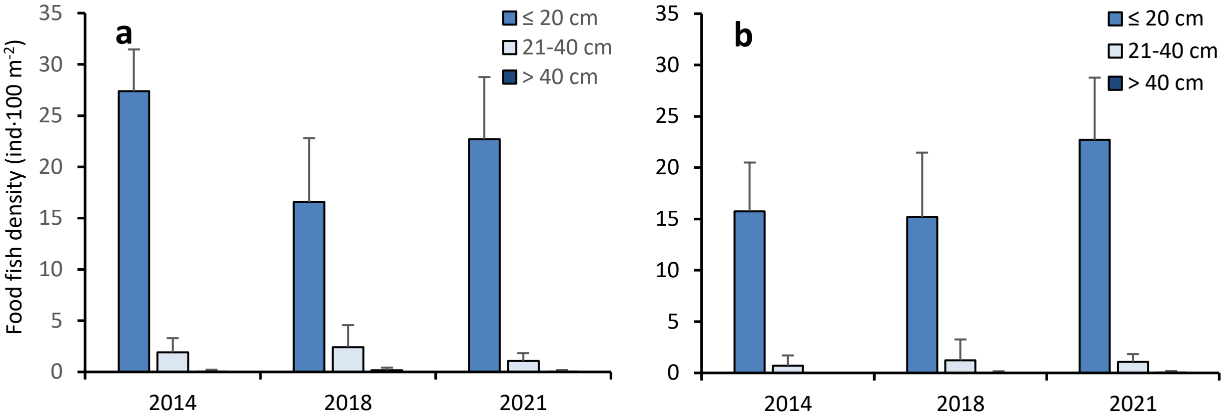


Figure 60*.* Food fish density (ind·100 m^{-2} ± SD) by size class within the Achang HPRA between 2014 and 2021, with data from two observers combined (a) and with only data from Observer 2 (b).

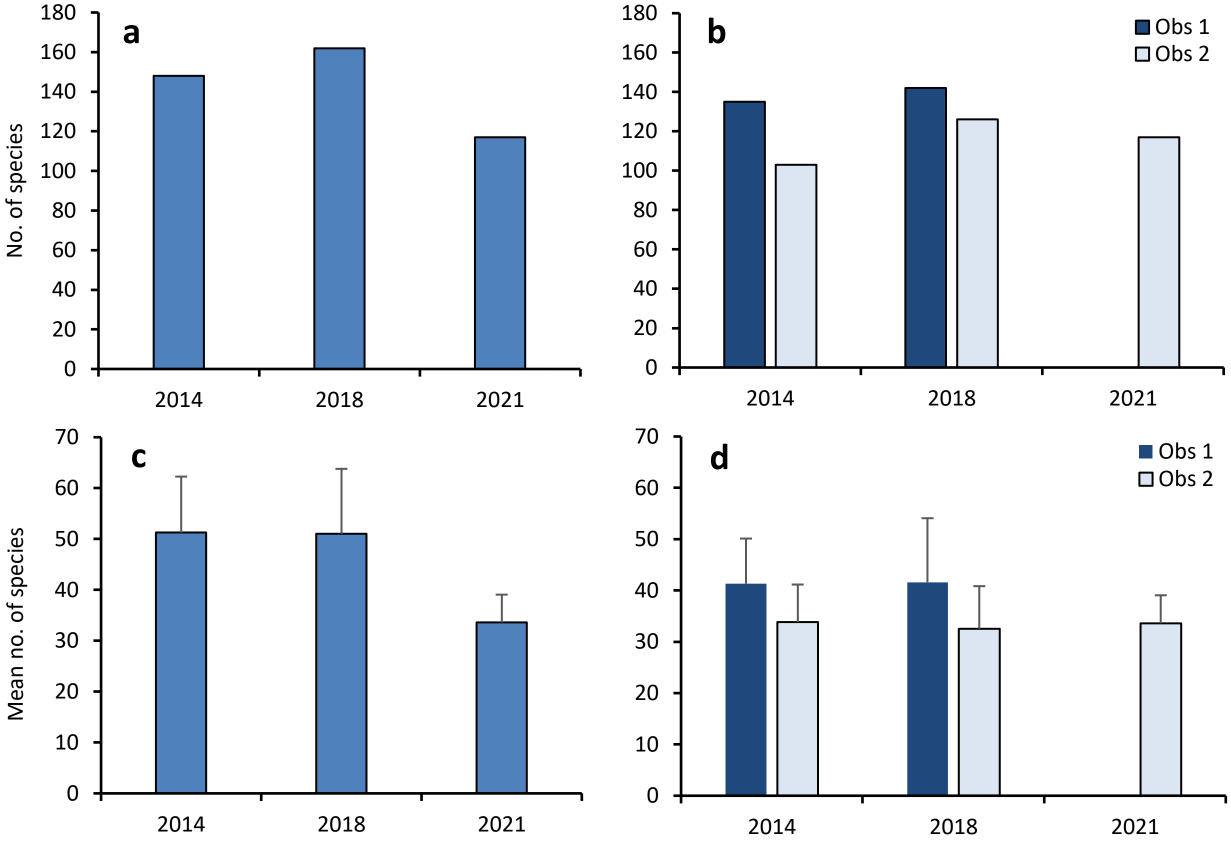


Figure 61. Total (a, b) and mean (c, d) species richness within the Achang HPRA between 2014 and 2021, with data from all observers combined (a, c) and presented separately (b, d). Note that the time interval between survey years varied.

148 species in 2014 to 117 species in 2021, after reaching a peak in 2018 at 162 species (Fig. 61a). Mean species richness also appeared to have declined from (51 ± 11) in 2014 to (34 ± 5) in 2021 within the Achang HPRA (Fig. 61c). However, as with total fish biomass, the apparent declines in total and mean species richness are very likely artifacts of interobserver bias rather than actual declines. When observations for the two observers are presented separately, the total number of species recorded by Observer 2 actually increased from 103 to 117 between 2014 and 2021 (Fig.

61b). This increase was relatively small, however, and the effect of Observer's 2 increasing experience since 2014 should be considered in a more detailed analysis.

Macroinvertebrates

Mean sea cucumber density within the Achang HPRA remained low throughout the study period, but may have declined from 1 to 0.2 ind·100 m^{-2} between 2014 and 2018 (Partover.test, p = 0.07); however, this change was only significant at the 0.1 level (Figs. 62–63). Two thermal stress events, one in 2016 and one in 2017, occurred between the 2014 and 2018 surveys, suggesting that, as was speculated regarding the decline in sea cucumber densities detected within other HPRAs, thermal stress may have had a direct or indirect role in the possible decline in sea cucumbers (mainly *Stichopus chloronotus*) detected within the Achang HPRA. The absence of severe thermal stress following the historically severe 2017 event may have allowed sea cucumber densities to partially recover by 2021. If the earlier (2012–2014) declines in sea cucumber densities detected at other HPRAs were indeed driven in part by a large-scale thermal stress event, it would not be unreasonable to speculate that sea cucumber densities within the Achang HPRA were higher prior to 2014.

The mean density of edible shells increased by more than 130% (from 1.7 to 3.9 ind·100 m^{-2}) between 2014 and 2018 (Partover.test, p = 0.02; Figs. 62–63). This increase in edible shells was comprised primarily by an increase in the density of top shells (*Tectus* spp.) and giant clams (*Tridacna* spp.). The difference in edible shell densities recorded in 2018 and 2021 was not statistically significant, nor was the difference between densities in 2014 and 2021. While statistical comparisons were not made for individual taxa, a large increase in the density of spider conch (*Lambis* spp.) between 2018 and 2021 is likely significant. No sea stars were recorded within the Achang HPRA in 2014 or 2021, while only a few (two *Culcita novaeguineae* and one *Linckia laevigata*) were observed in 2018. The density of large sea urchins (*Diadema* spp. and *Echinothrix* spp.) was relatively low through the study period, but may have increased by more than 200% between 2014 and 2018 (Partover.test, p = 0.05).

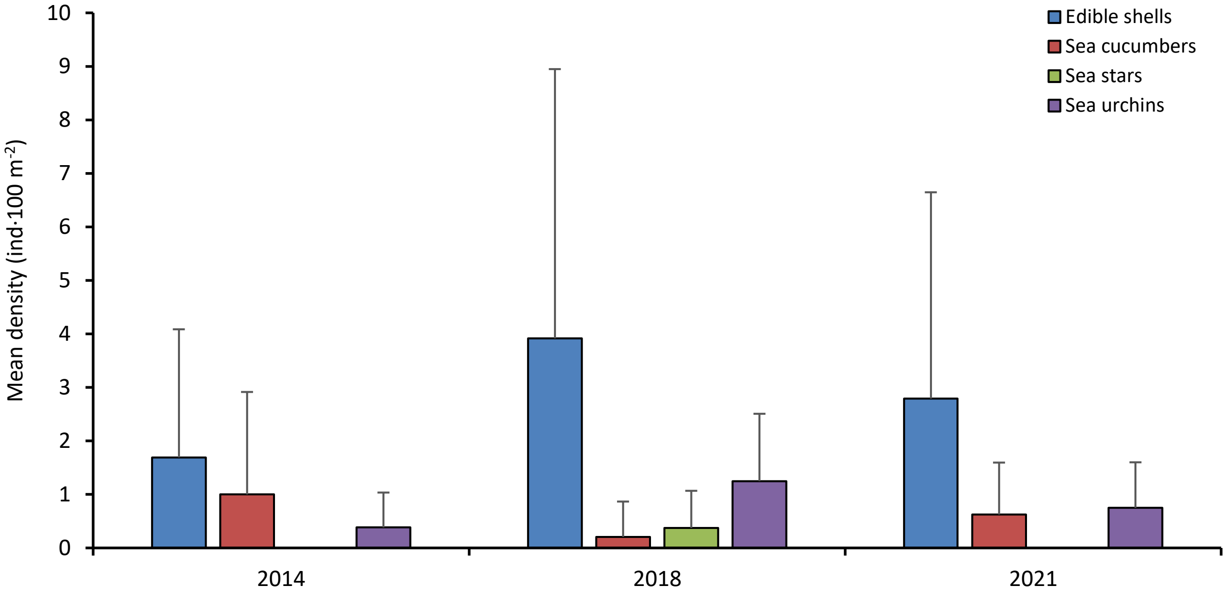


Figure 62. Mean density (ind·100 m^{-2} ± SD) of broad macroinvertebrate groups within the Achang HPRA between 2014 and 2021.

Discussion

While a small increase in coral cover (from 5% to 6%) was detected within the Achang HPRA between 2014 and 2018, this amount of coral cover is still quite low, and an increase of 1% absolute coral cover in a four-year period is not particularly notable. As mentioned above, the increase may actually be a result of the significantly lower cyanobacteria cover in 2018 compared to 2014, rather than represent an actual increase in the amount of coral

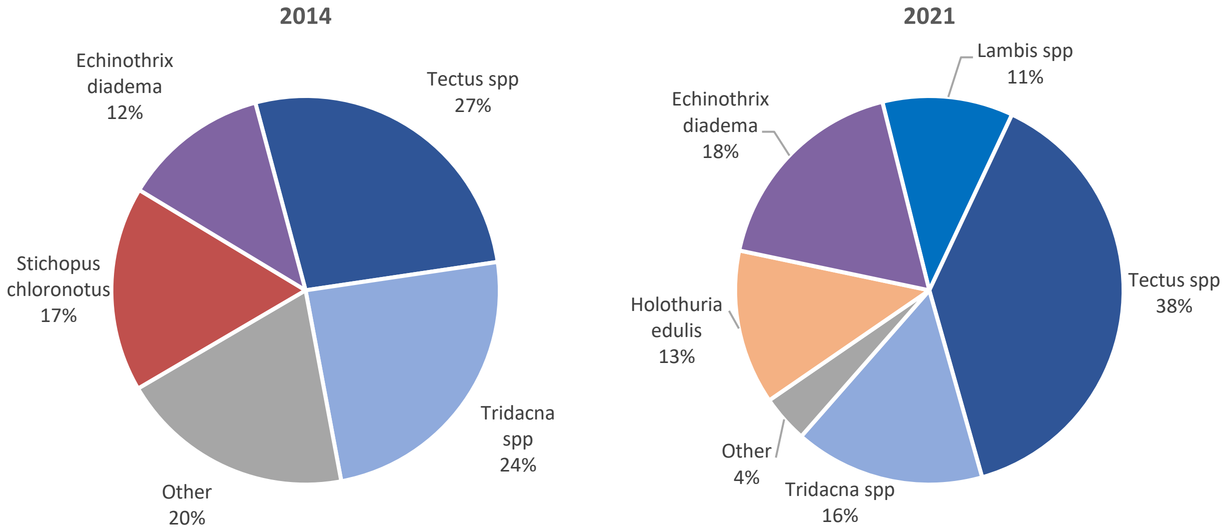


Figure 63. Relative abundance of individual macroinvertebrate taxa within the Achang HPRA in 2014 and 2021.

within the HPRA. However, if the slight increase in coral cover during this period, which included the severe 2017 bleaching event, was real, it would suggest that the coral assemblage within the HPRA at that time was at least somewhat resilient to heat stress events and other stressors that may have impacted the area.

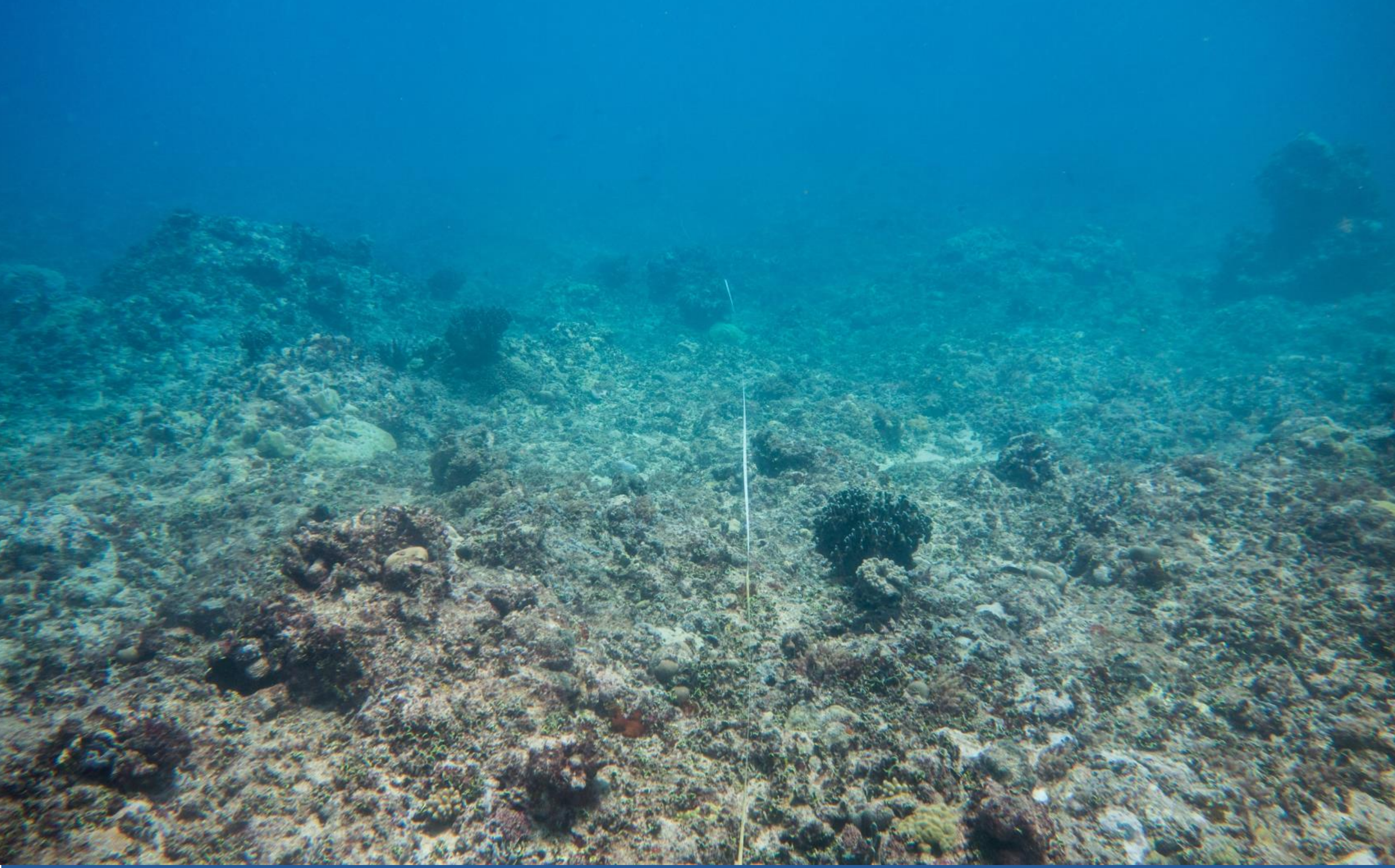
As with other reef areas along the southeast coast, historical data is limited for the submarine terrace in the vicinity of the Achang HPRA. The MPC visited the submarine terrace in Asgadao Bay, which falls within the HPRA boundaries, in the mid-2000s, when observations of outbreak densities of COTS were reported in the area, and noted a considerably higher cover of acroporids (especially encrusting *Montipora*) and other stress-susceptible coral taxa compared to visits made years later. While a detailed analysis of benthic cover data obtained during towed-diver benthic photo surveys carried out by NOAA PIFSC along the 12 m depth contour at seaward slope locations around the island since 2003 has not been carried out, a preliminary analysis by the MPC using benthic cover estimates provided by NOAA PIFSC showed a 60% decline in total coral cover along the east coast of Guam, from 18 ± 7% to 7 ± 4%, between 2003 and 2014. A separate analysis to the coral genus level carried out by the MPC using NOAA PIFSC towed diver images for a subset of tows obtained in 2003 and 2011 found that encrusting *Montipora* comprised >70% of total coral cover along the east coast in 2003, but dropped to <5% of total coral cover by 2011. The towed-diver surveys recorded outbreak densities of COTS at multiple locations around the island between this period, including along the southeast coast, with a peak of COTS activity in 2007 (Vargas-Ángel 2012). It is highly likely that the coral community along the submarine terrace zone within the Achang HPRA had been impacted by previous COTS outbreaks, but it's not clear how different the benthic communities surveyed by NOAA PIFSC in 2003 and visited by the MPC in the mid-2000s were to the communities of previous decades. It's possible that they represent recovered communities similar to those prior to the major COTS outbreaks in the late 1960s/early 1970s and the 1980s, but it's more likely that they represent an already-compromised benthic community that suffered further losses with additional COTS activity in the mid-2000s. The persistence of a sediment-laden algal matrix and very low coral cover along the submarine terrace in the Achang HPRA between 2014 and 2018 suggests that, like the benthic community at the mouth of Fouha Bay, recovery processes, such as coral recruitment and the maintenance of suitable substrate, in this area are compromised, and that coral mortality (e.g., by continued COTS activity, heat stress, disease, etc.) is exceeding the rate of recovery possible in this system.

Total coral cover at the island-wide bleaching response and recovery site (579) that occurs within the reef front zone adjacent to the Achang HPRA appeared to decrease by 50%, from 26% to 13%, between 2013 and 2017. The general absence of COTS in the higher wave energy reef front zone suggests that this decline is very likely due to the heat and light stress associated with the 2013–2017 bleaching events. However, unlike the recovery in coral cover observed at the reef front site near the Fouha Bay HPRA, and at some of the other reef front sites around the island (see p. 26), coral cover at the Achang reef front site remained at 14% in 2021, identical to coral cover in 2017, despite several years without significant heat stress. A more in-depth analysis will reveal how the coral community may have shifted since the onset of repeated bleaching events in 2013.

While the previously discussed interobserver bias issues must still be considered when comparisons are made using combined data from both fish observers, the availability of fish survey data obtained by Observer 1 in 2014 and 2018 allows for a relatively high degree of confidence in the assertion that generally low food fish biomass persisted within the Achang HPRA between 2014 and 2018. The availability of data obtained by Observer 2 for 2014, 2018 and 2021 indicates that food fish biomass and the density of small food fishes likely increased between 2018 and 2021. The biomass of surgeonfishes, wrasses, emperors, parrotfishes, and groupers may have increased between 2014 and 2021, while the biomass of all other food fish families appears to have remained relatively stable. Total fish biomass in the Achang HPRA in 2018 was relatively low, at 17-43% of the potential total reef biomass estimated for an impaired Guam reef by Williams et al. (2015). The low fish biomass and density within the Achang HPRA can be at least partially explained by the very low reef complexity, and resulting limited fish habitat availability, across much of the submarine terrace occurring within the HPRA boundaries. GLTMP biologist have remarked on the “patchiness” of reef fish populations within the Achang HPRA, with rather stark differences in the size and number of reef fish at a small number of transects in reef areas with a high degree of structural complexity, in comparison to the majority of transects, which occur in relatively flat, featureless reef areas with limited habitat availability. Further study, particularly involving the integration of data obtained using various fish survey methods, may better resolve any trends in the reef fish community that may have occurred within the Fouha Bay HPRA over the last decade, and may provide an indication of the likely causes of any declines or improvements in reef fish community condition.

The decline in sea cucumber density, mainly spiky sea cucumber density, within the Achang HPRA between 2014 and 2018 may be a result of impacts associated with the severe 2017 heat stress event. Interestingly, the density of edible shells, which was comprised primarily of top shells and bleaching-susceptible giant clams, increased significantly during this period. A more detailed examination of taxon-specific changes during this period may shed more light on the potential impacts of the heat stress events to the macroinvertebrate community within the Achang HPRA.

Bleached and recently killed corals, mainly *<u>Acropora</u>* *spp. and* *<u>Montipora</u>* *spp., along the reef front near Asgadao Bay, just upslope from the Achang HPRA, at the end of September 2013. While COTS outbreaks in the mid-2000s had significantly reduced the abundance of stress-susceptible corals on the seaward slope, these corals still thrived in the reef front zone, where the sea stars typically couldn't reach due to the high wave energy. The severe coral bleaching event in 2013 was the first of several large scale heat stress event to impact these corals through 2017, with the most severe impacts recorded along the southeast coast, where these corals were in very high abundance prior to bleaching.*



COCOS-EAST

Key findings

- Coral cover remained very low (5%) between 2014 and 2018
- The cover of mounding *Porites* spp., which comprised more than half of the total coral cover, did not change between 2014 and 2018, while a decline in the cover of other taxa, such as *Astreopora* and *Pocillopora*, during this period may have been caused by heat stress or predation by COTS
- Data and staff capacity limitations currently prevent detailed analyses of reef fish survey data obtained between 2014 and 2021, but there are some indications that food fish biomass remained low during this period
- The biomass of parrotfishes may have increased between 2014 and 2021, while few or no emperors, groupers, or jacks were observed during any sampling year
- Total fish biomass in 2014 was relatively low (10–20 $g \cdot m^{-2}$), at 19–38% of the potential total reef fish biomass estimated for an unimpaired Guam reef by Williams et al. (2015)
- The density of small food fishes may have increased between 2014 and 2021 and the density of moderately-sized food fishes may have increased; few or no large food fishes were observed during surveys
- While macroinvertebrate survey data were limited, the density of sea cucumbers and edible shells (mainly top shells and giant clams) may have declined by > 50% between 2014 and 2021

***Above**: An algae-dominated benthic community along the submarine terrace within the Cocos-East site in 2021. Similar to the Achang HPRA, coral cover within the Cocos-East site remained very low (~5%) between 2014 and 2021.*

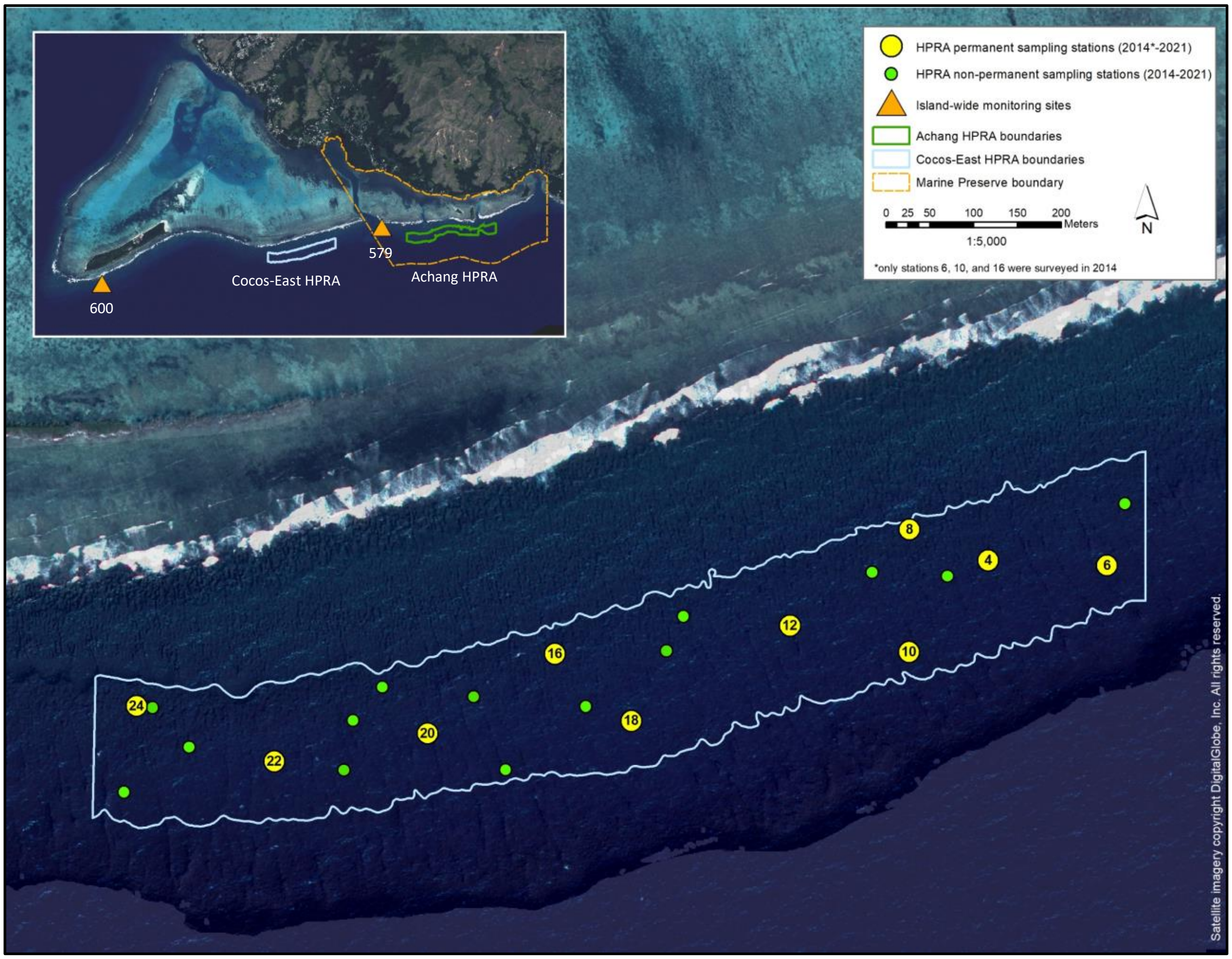


Figure 64. Map of the Cocos-East HPRA site boundaries and sampling stations.

Site overview

The Cocos-East HPRA was established in 2014 as a non-preserve comparison site for the nearby Achang HPRA, which occurs within the Achang Reef Flat Marine Preserve. As mentioned above, these sites were selected primarily to serve as another potential preserve/non-preserve pair, but also because these two sites are influenced by poor water quality associated with the Manell River and smaller rivers in the Manell-Geus watershed unit. The Manell-Geus watershed unit was designated as a high priority watershed management area just prior to selection as monitoring sites. It has since been designated as a NOAA Habitat Blueprint site, and is currently the location of several pilot watershed restoration and community outreach projects.

Data collection within the Cocos-East HPRA occurs along the seaward slope (7–15 m depth) at a total of 10 permanent sampling stations and up to 10 non-permanent stations that are re-randomized each data collection year (Fig. 63). As with the Achang HPRA, the original intent was to visit the Cocos-East HPRA on a biennial basis, rather than annually, due to the challenge in regularly accessing these sites. The closest permanently established island-wide bleaching response and recovery site (579) is located along the reef front approximately 800 m to the east of the HPRA, on the opposite side of the Mannel Channel. Another reef front site (600) is located 3 km to the southeast of the HPRA, near the south tip of Cocos Island. No RFMP sites are located near the Cocos-East HPRA.

A limited amount of baseline data collection occurred at the Cocos-East HPRA in 2014, with more complete data collection efforts occurring in 2018 and 2021 (Appendix A). Staffing limitations and the re-prioritization of resources required to carry out an island-wide reef resilience survey in 2016 prevented the collection of data that year, the limited availability of the lead fish observer during 2018 prevented the collection of reef fish data that year, and the COVID pandemic prevented data collection at the site in 2020. Benthic cover and macroinvertebrate abundance data was collected at four shallow (5 m depth) sites along the reef front near the Cocos-East HPRA in 2013 as part of the UOGML-led bleaching response effort. Two of these reef front sites (579 and 600) were re-visited in 2015 as part of an island-wide bleaching recovery assessment, in 2016 as part of an island-wide reef resilience assessment, in 2017 as part of the response to another major bleaching event, and were surveyed and marked with rebar in 2021.

The results of an analysis of benthic cover obtained within the Cocos-East HPRA in 2014 and 2018, and reef fish and macroinvertebrate data between 2014 and 2021 are provided below; the analysis of benthic photo transect images obtained in 2021 were on-going at the time this report was being developed. The results of an analysis of data obtained at the nearby island-wide bleaching response and recovery sites are presented beginning on p. 26 but are also referenced in the "Discussion" section below.

Benthic cover

There was no detectable change in mean total coral cover across the Cocos-East HPRA between 2014 and 2018, with cover remaining at 5% for both years (Fig. 65). More than half of the coral community within the HPRA was comprised of mounding *Porites* spp. in both 2014 (54% of total coral cover) and 2018 (71% of total coral cover), with merulinids and other taxa comprising the remaining proportion of total coral cover (Figs. 66–67). The absolute cover of mounding *Porites* spp. appeared to increase slightly between 2014 and 2018, but this change was not statistically significant (Wilcox Signed Rank Test, paired stations only, p = 0.13). The cover of merulinids remained <1% during this period, but the combined cover of all other coral taxa declined from 1.6% to 0.8% (Wilcox Signed Rank Test, paired stations only, p = 0.01).

Statistical comparisons were not made for non-coral benthic features through time at the Cocos-East HPRA, but the cover of crustose coralline red algae may have increased from 13% to 21% during this period (Fig. 65). Most of the increase in CCRA cover was among the Order Corallinales (the pink and red crusts), which may have increased

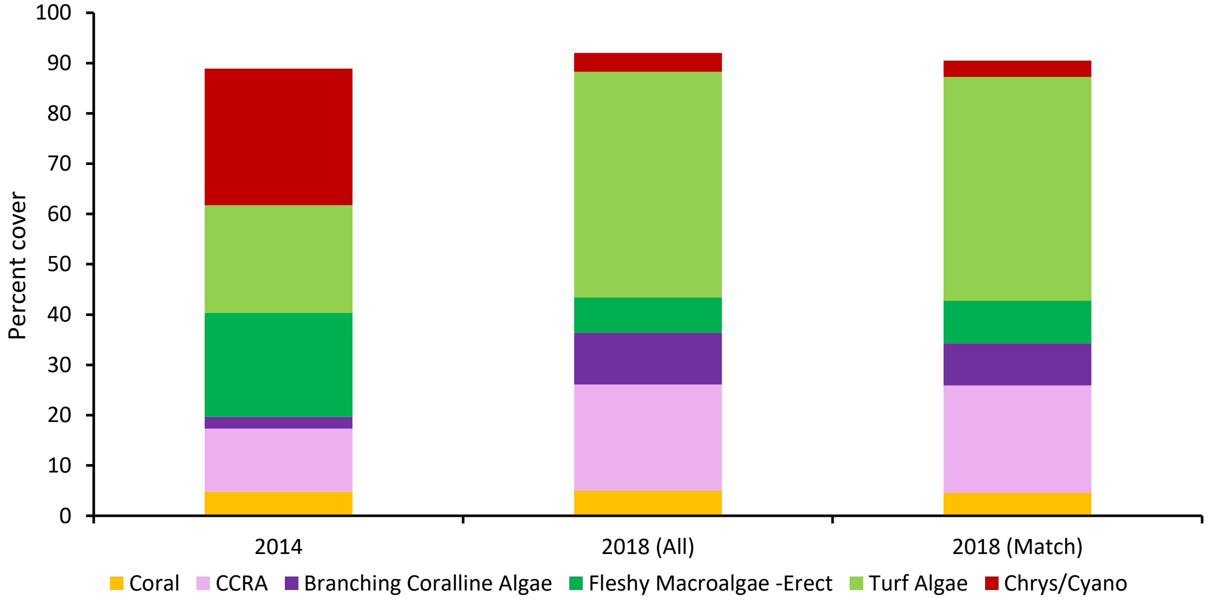


***Figure 65**.* Percent cover of major benthic classes for the Cocos-East HPRA in 2014 and 2018. Because data were only collected at three permanent transects in 2014, data from 2018 are presented for all 10 permanent transects ("All"), as well as for only the three permanent transects surveyed in both 2014 and 2018 ("Match"). The data indicate that percent cover for major benthic cover categories was highly similar between the selection of three transects and all transects surveyed, suggesting that the benthic community is relatively homogeneous across the site, and that it is reasonable to at least tentatively consider the benthic community at the three transects surveyed in both 2014 and 2018 as representative of the broader site.

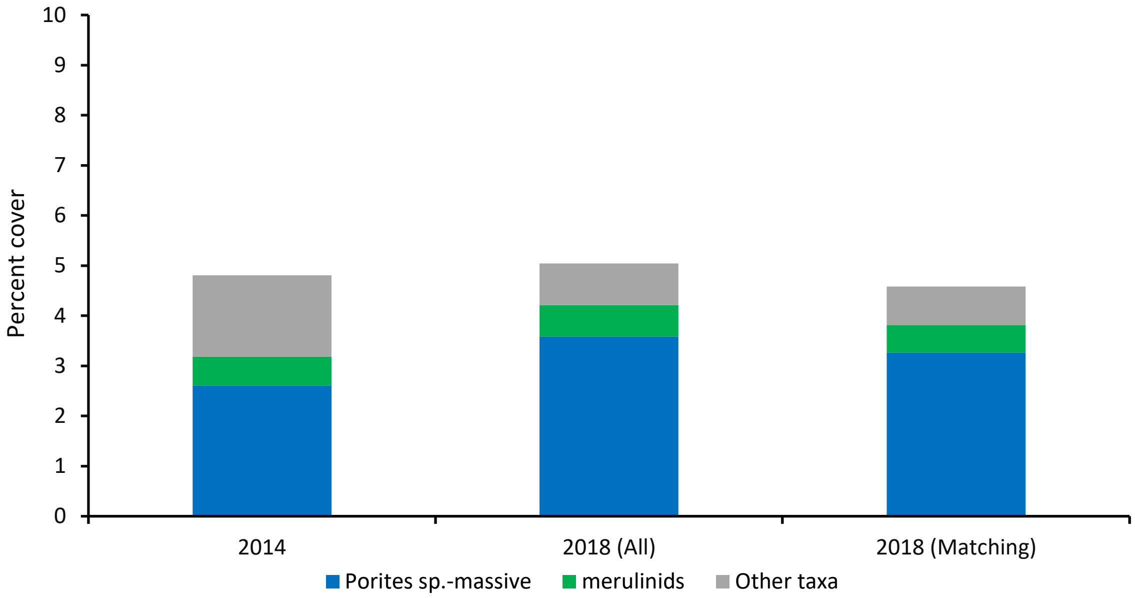


Figure 66. Percent cover of coral taxa for the Cocos-East HPRA in 2014 and 2018. As with the previous figure, data from 2018 are presented for all 10 permanent transects ("All"), as well as for only the three permanent transects surveyed in both 2014 and 2018 ("Matching").

from 8% to 16%. The cover of branching coralline algae, fleshy macroalgae, and turf algae appeared to all increase between 2014 and 2018, but this could be a result of a large decline in cyanobacteria and chrysophytes. Cyanobacteria and chrysophytes are generally more ephemeral than other benthic cover classes, and can quickly cover other benthic features before then dying back and revealing the benthic features living below.

Associated biological communities

Reef fishes

Biomass

Mean food fish biomass within the Cocos-East HPRA appeared to remain stable (~15–16 g·m^{-2}) between 2014 and 2021, although it's possible it could have fluctuated in the intervening years (Fig. 67a). The significant interobserver bias evident when comparing mean food fish biomass values reported by Observer 1 and Observer 2 in 2014 makes it problematic to compare combined data from 2014 with data collected solely by Observer 2 in 2021 (Fig. 67b). However, the large variability in the mean food fish biomass across sampling stations, and the relatively limited number of survey replicates carried out by Observer 2 in 2014 (7 reps at 7 stations, compared to 40 reps at 20 stations in 2021), also limits comparisons using only those observations obtained by Observer 2. A statistical comparison of mean food fish biomass reported by Observer 2 in 2014 and 2021 indicates that there was no significant change during this period (Partover.test, p = 0.17).

Non-food fish were not counted and sized beginning in 2020, so a total fish biomass value within the Cocos-East HPRA can only be reported for 2014 (Fig. 68). The total fish biomass of around 10–20 g·m^{-2} reported for the Cocos-East HPRA in 2014 is approximately 19–38% of the potential total reef fish biomass value (53 ± 7 g·m^{-2}) estimated for an unimpaired Guam reef community by Williams et al. (2015).

Due to the significant interobserver bias discussed above, biomass values for selected food fish families within the Cocos-East HPRA between 2014 and 2021 were calculated using only those observations recorded by Observer 2, who participated in surveys across both sampling periods (Fig. 68). While statistical comparisons were not carried out for most families for this limited preliminary analysis, the biomass of parrotfishes appears to have increased during this period (Partover.test, p = 0.043). The biomass of wrasses may have also increased slightly, while the biomass of surgeonfishes, snappers, groupers, and "other families" appears to have remained relatively stable. Few emperors or groupers were observed, and no jacks were observed either year.

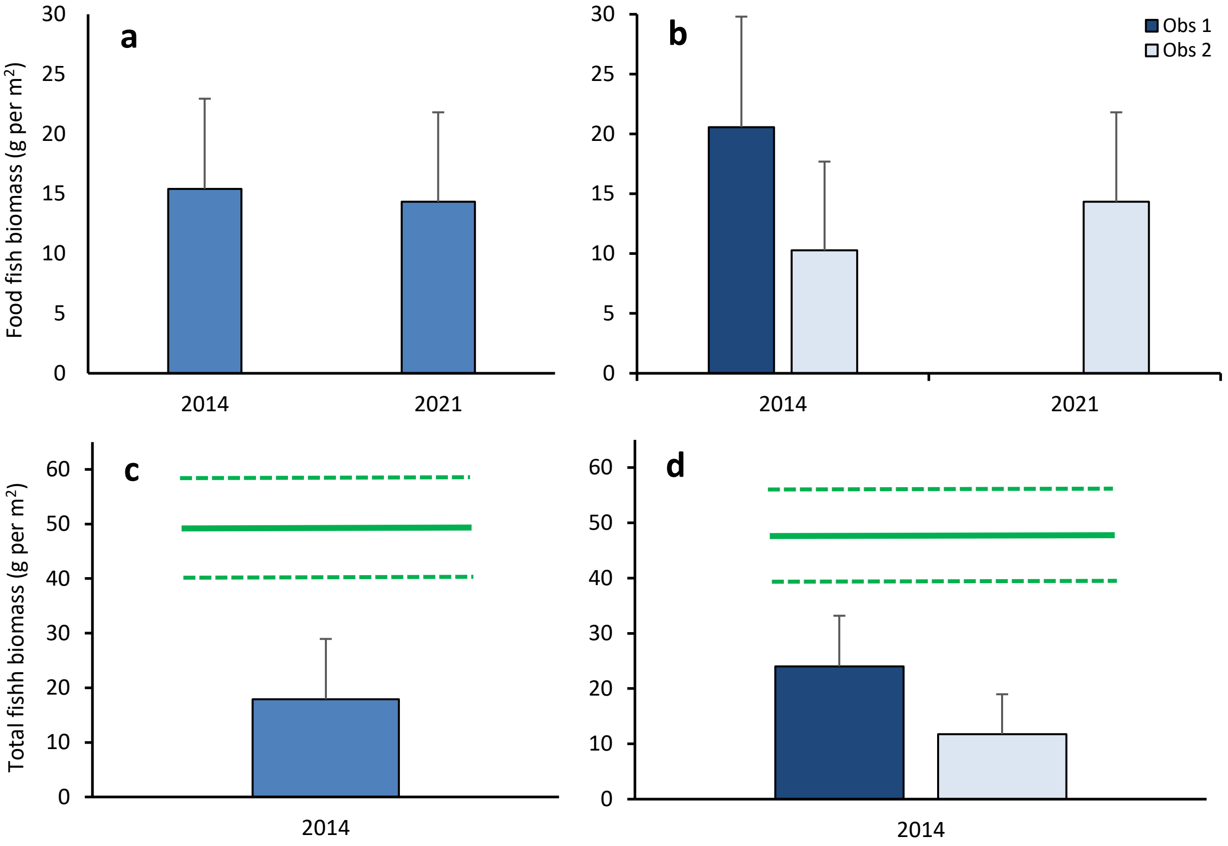


Figure 67. Food fish biomass (a, b) and total fish biomass (c, d) within the Cocos-East HPRA in 2014 and 2021 and in 2014, respectively, with data from both observers combined (a, c) and presented separately (b, d). The green lines indicate the potential total reef fish biomass for an unimpaired Guam reef community (53 ± 7 $g{\cdot}m^{-2}$) estimated by Williams et al. (2015).

Density

The mean density of smaller (≤ 20 cm) and moderately sized (21 ≤ 40 cm) food fishes within the Cocos-East HPRA appeared to remain relatively stable between 2014 and 2021 (Fig. 69a), but this apparent stability is likely a result of interobserver bias. When only those observations obtained by Observer 2 are examined, a clear increase in the density of small food fishes is detected between 2014 and 2021 (Partover.test, $p < 0.001$), while the density of moderately sized fishes may have increased slightly during this period (Partover.test, $p = 0.08$; Fig. 69b). No large (> 40 cm) food fish were observed in 2014, while only a very small number of individuals ≥ 40 cm was detected in 2021. As discussed above, the low densities of moderately sized and large food fishes make the detection of trends in these groups challenging; however, a more detailed analysis may be able to detect trends in the density of these larger size classes.

Species Richness

The slight increase (from 112 to 121) in the total number of species recorded within the Cocos-East HPRA across all sampling stations is likely an artifact of differing levels of survey effort, as the number of replicate surveys conducted in 2014 (14) was substantially lower than in 2021 (40; Fig. 70). One would expect additional species to be detected with an increased number of surveys. The increase from 80 to 121 species detected by Observer 2 is also likely due to an even greater disparity in survey effort, as this observer carried out only 7 replicate surveys in 2014 compared to 40 in 2021. Still, the generally narrow range of the total number of recorded species (~110–120) between 2014 and 2021 (using data from both observers combined) provides a good indication of the level of species richness within the Cocos-East HPRA during this period. A more detailed analysis of species counts obtained by the different observers is likely to reveal species that are consistently detected by one observer but not another, or other biases

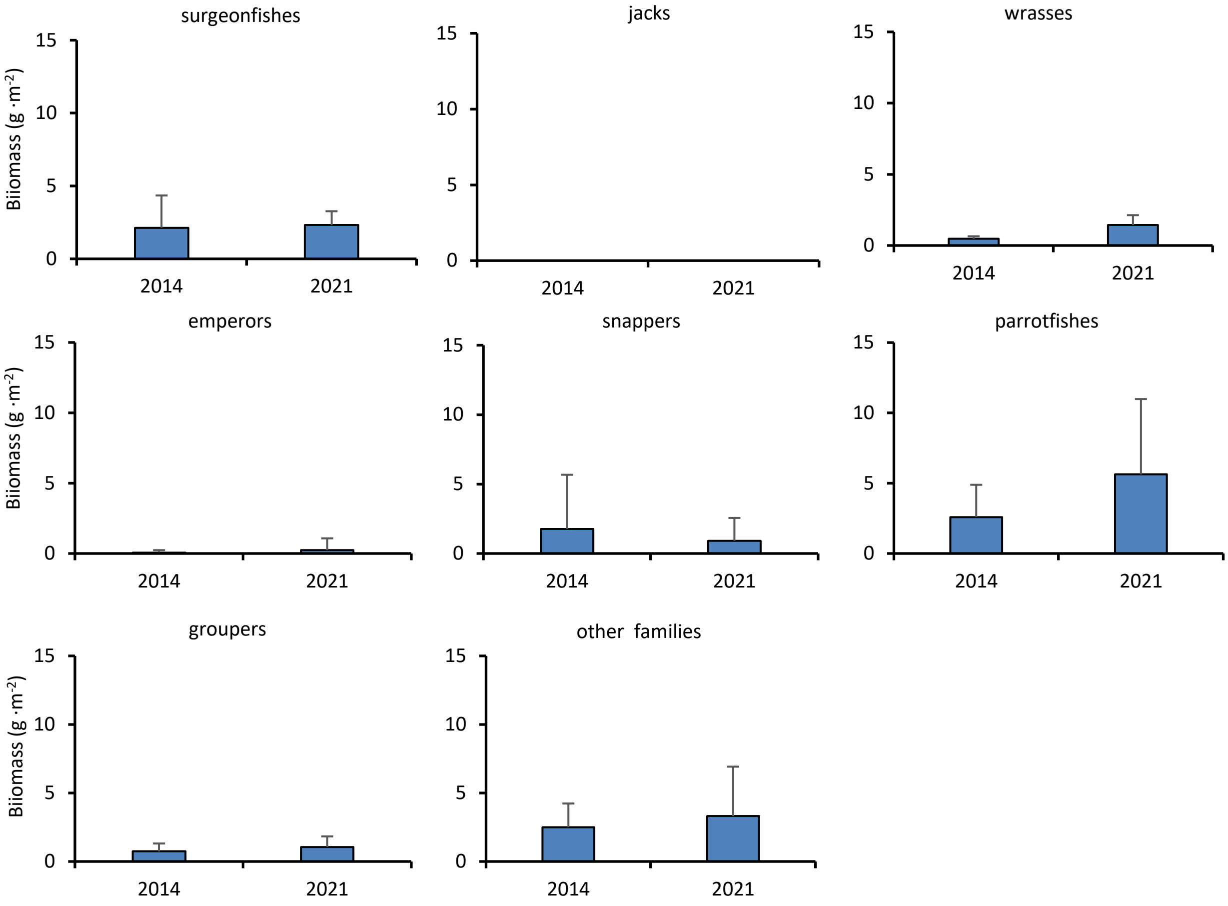


Figure 68. Biomass (g·m^{-2}) of selected food fish families within the Cocos-East HPRA in 2014 and 2021. Due to the significant interobserver bias detected early in the analysis, these values were calculated using only those observations collected by Observer 2, as this observer carried out surveys both years.

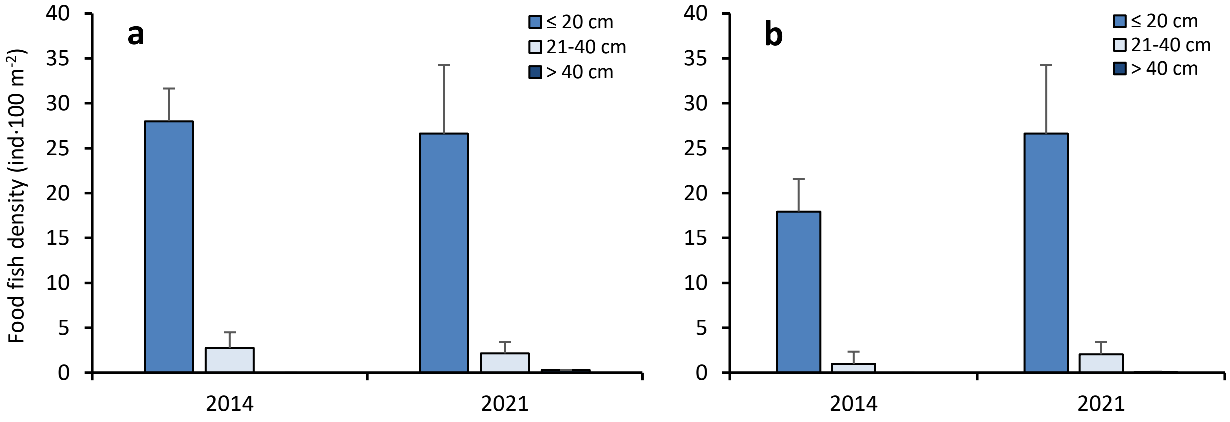


Figure 69*.* Food fish density (ind·100 m^{-2}± SD) by size class within the Cocos-East HPRA in 2014 and 2021, with data from two observers combined (a) and with only data from Observer 2 (b).

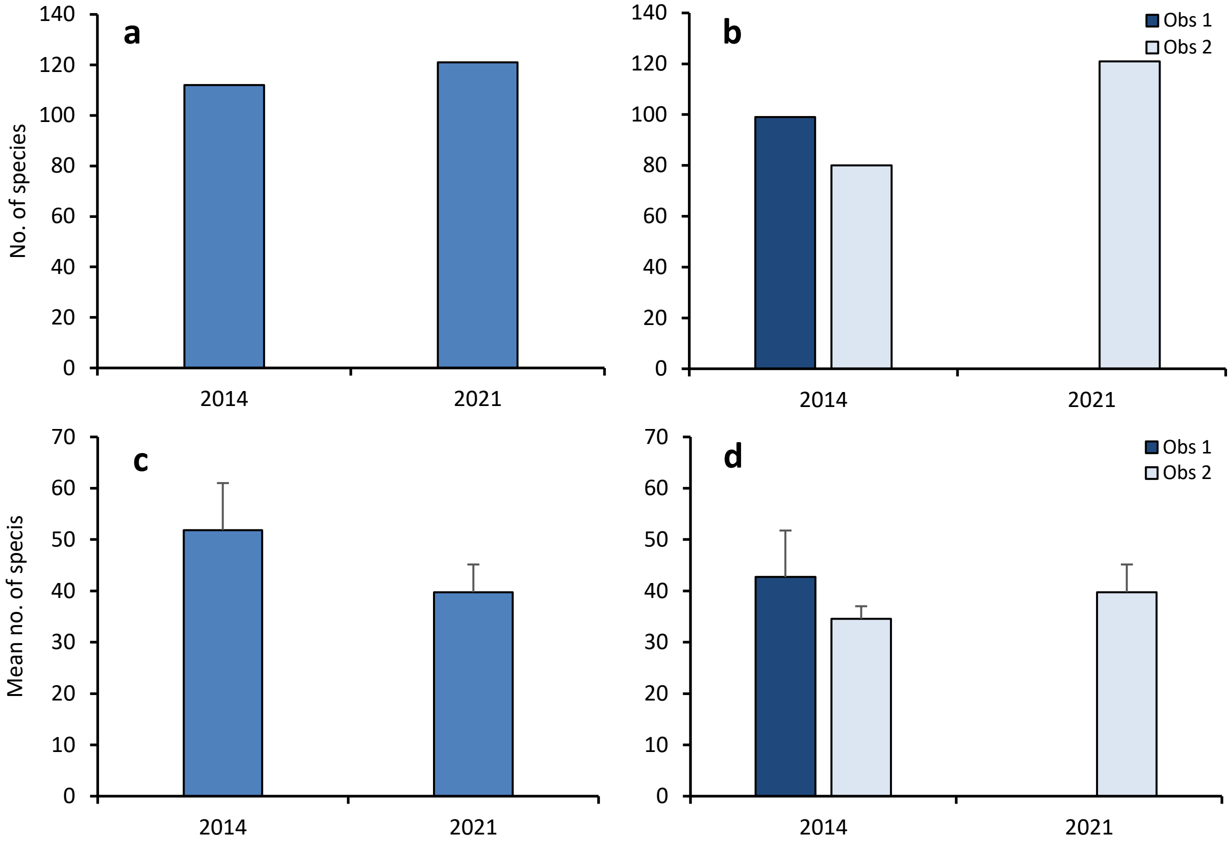


Figure 70. Total (a, b) and mean (c, d) species richness within the Cocos-East HPRA in 2014 and 2021, with data from all observers combined (a, c) and presented separately (b, d).

that could be minimized with targeted training and calibration. Additionally, a deeper analysis that accounts for rugosity and benthic community structure may also reveal that the fluctuations in the survey data may reflect real-world change in the species diversity in the East Agana Bay HPRA.

Macroinvertebrates

While some statistically significant changes in macroinvertebrate densities were detected during the study period, it's important to note that very few Cocos-East sampling stations were surveyed for macroinvertebrates in 2014 and 2018 (three and five stations, respectively); as such, any comparisons between those years and between those years and 2021 should be made cautiously. With the data limitations in mind, mean sea cucumber density within the Cocos-East HPRA may have decreased by around 50% between 2014 and 2021 (Partover.test, $p = 0.1$), but this change was only significant at the 0.1 level (Figs. 71–72).

The mean density of edible shells within the Cocos-East HPRA appeared to have decreased by nearly 60% (from 11.7 to 4.9 ind·100 m^{-2}) between 2014 to 2021 (Partover.test, $p < 0.01$; Figs. 71–72).While statistical comparisons were not made for individual taxa across survey years, this decrease in edible shell density appears to have been driven primarily by a decrease in the density of top shells (*Tectus* spp.) and giant clams (*Tridacna* spp.). No significant changes in the density of large sea stars or sea urchins were detected within the Cocos-East HPRA between 2014 and 2021, but their generally low densities at the site and the limited amount of data collected in 2014 and 2018 limits the ability to detect changes in the densities of these taxa.

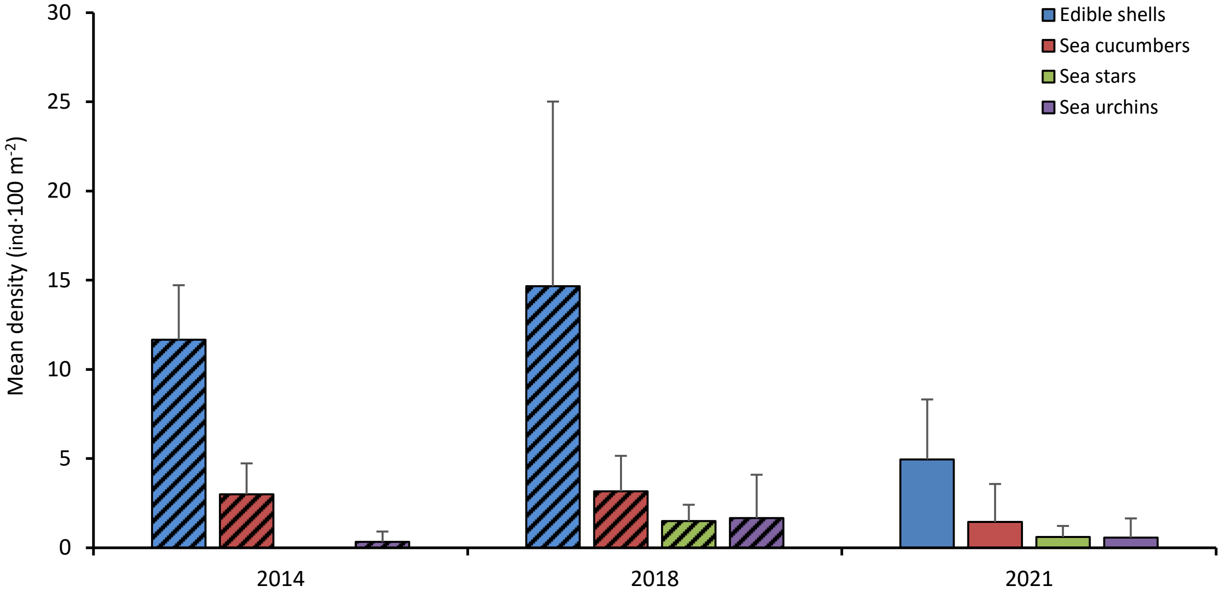


Figure 71. Mean density (ind·100 m^{-2} ± SD) for broad macroinvertebrate taxonomic groups within the Cocos-East HPRA between 2014 and 2021. Mean density values in 2014 and 2018 (represented by bars with the diagonal lines) should be considered cautiously, as these data include observations obtained at 3 and 5 out of the usual 20 sampling station, respectively.

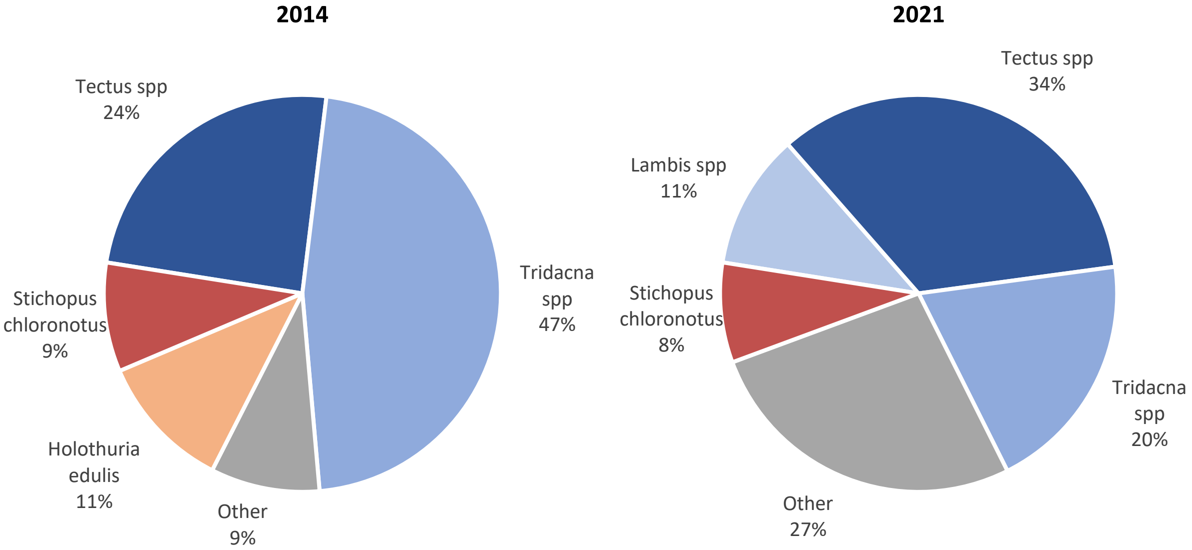


Figure 72. Total relative abundance for individual macroinvertebrate taxa within the Cocos-East HPRA in 2014 and 2021.

Discussion

Very low (5%) coral cover persisted within the Cocos-East HPRA between 2014 and 2018. As observed in all of the other HPRAs, the coral community within the Cocos-East HPRA was primarily comprised of stress-tolerant taxa such as mounding *Porites* spp. The slight, but still detectable, decrease in the cover of corals other than mounding *Porites* and merulinids suggests that taxa generally considered more susceptible to stress continue to face challenges at the

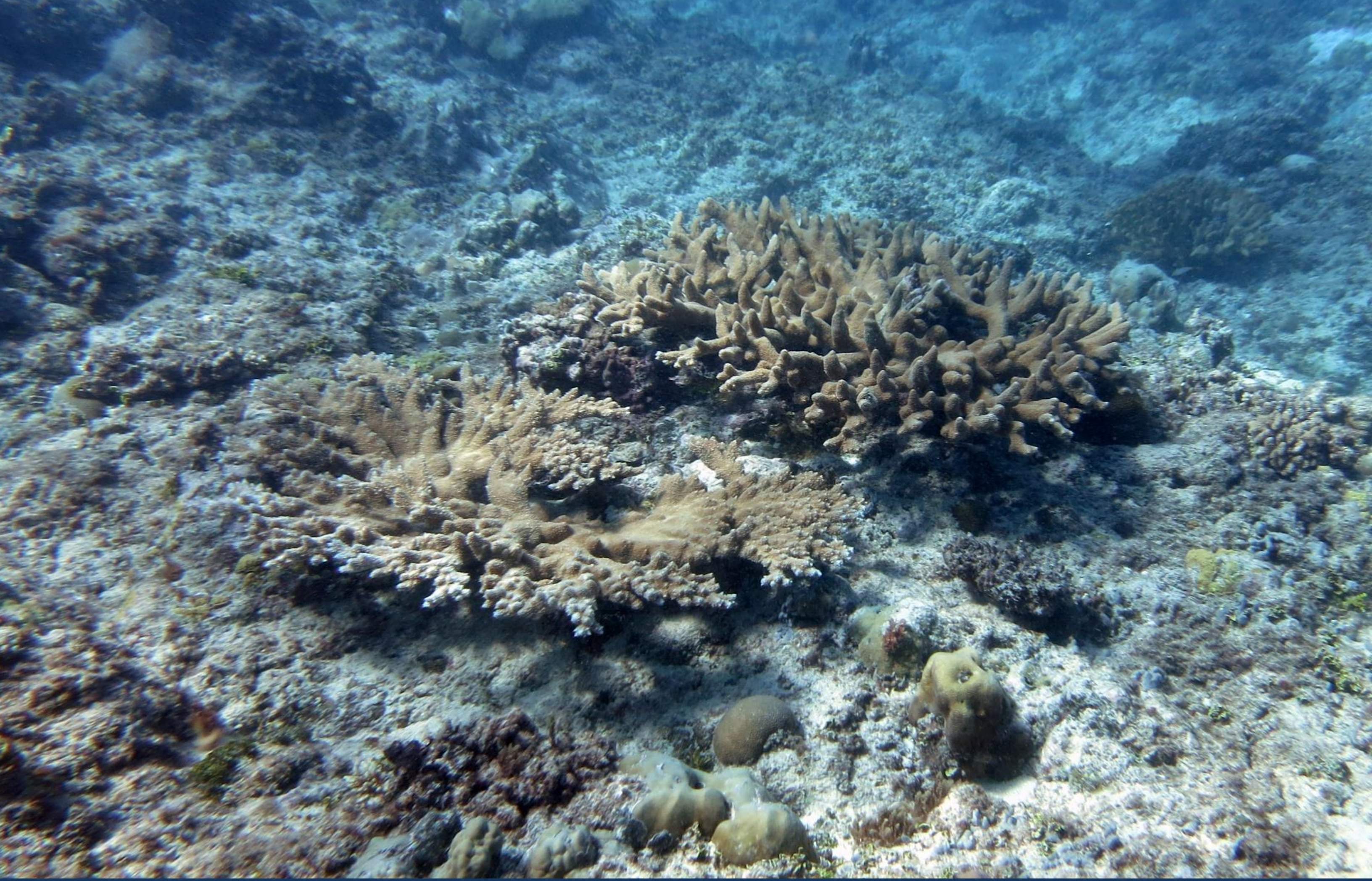
Cocos-East site. While it's not clear exactly what may have caused this decline, it is likely that thermal stress and predation by COTS both played a role. Broadly speaking, coral taxa more susceptible to stress, such as *Acropora*, *Montipora*, and *Pocillopora*, tend to grow more rapidly in favorable conditions and also tend to exhibit more complex colony structures that can provide habitat for a variety of reef organisms. Their continued decline not only means a further decline in diversity within the coral community, but will also result in a further "flattening" of the reef and the associated loss in habitat for fish and invertebrates.

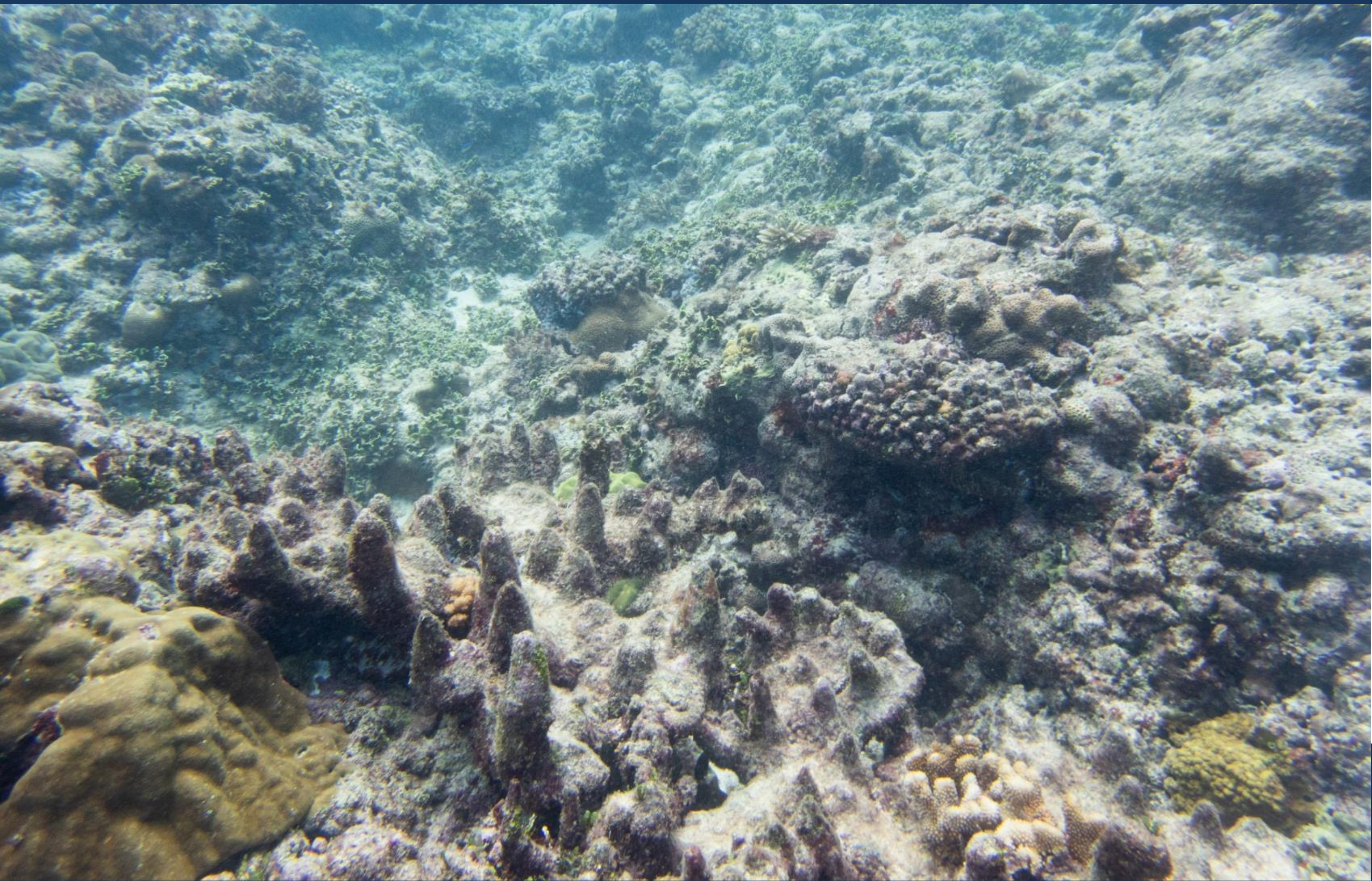
As discussed in regard to the Achang HPRA, it is likely that coral cover, particularly the cover of *Montipora* spp. and other generally stress-susceptible coral taxa, may have been higher within the Cocos-East HPRA prior to the onset of severe bleaching in 2013, but the multi-year COTS outbreak in the mid-2000s likely caused a significant reduction in coral cover in the area even prior to the 2013–2017 bleaching events. However, further study using historical data available for the area is required to better understand the changes in benthic community structure and condition that may have occurred withing the vicinity of the Cocos-East HPRA in previous decades.

Significant declines in total coral cover were detected at the two island-wide bleaching response and recovery sites that occur within the reef front zone in the general vicinity of the Cocos-East HPRA as a result of the multiple bleaching events between 2013 and 2017. A catastrophic 77% decline, from 50% to 12%, in coral cover was reported for the site behind Cocos Island (600), while a still notable 50% decline, from 26% to 13%, was reported for the site to the east of Mannel Channel (579). Unlike the recovery in coral cover observed at the reef front site near the Fouha Bay HPRA, and at some of the other reef front sites around the island (see p. 26), coral cover at site 600 did not recover, at 12% coral cover in both 2017 and 2021. A slight (8%) increase, from 13% to 14%, in coral cover may have occurred at site 579 during this period, but a statistical analysis must be carried out to determine if this change was statistically significant. Further analysis will also reveal how the coral community at these reef front sites shifted since the onset of repeated bleaching events in 2013.

Comparisons of reef fish community parameters within the Cocos-East HPRA between 2014 and 2021 is unfortunately limited due to the relatively small number of reef fish surveys carried out within the HPRA in 2014, the lack of any fish survey data for 2018, and the significant interobserver bias issues detected during the analysis. However, the available data clearly indicate that food fish biomass was relatively low in both 2014 and 2021. Using only data collected by Observer 2, who carried out surveys in both 2014 and 2021, food fish biomass within the HPRA did not appear to change between 2014 and 2021, in contrast to the increases in food fish biomass detected at the other HPRAs between 2019 and 2021. The density of small and moderately sized food fishes appears to have increased between 2014 and 2021, but few or no large food fishes were observed either survey year. The biomass of parrotfishes may have increased between 2014 and 2021; few or no emperors, groupers, or jacks were observed during either sampling year. Total fish biomass in the Cocos-East HPRA in 2014 was relatively low, at 19–38% of the potential total reef biomass estimated for an impaired Guam reef by Williams et al. (2015). Further study, including the integration of data obtained using various fish survey methods as well as benthic habitat data, may better resolve any trends in the reef fish community that may have occurred within the Cocos-East HPRA over the last decade, and may provide an indication of the likely causes of any declines or improvements in reef fish community condition.

The possible decline in the density of sea cucumbers (mainly spiky sea cucumbers) and edible shells (mainly top shells and giant clams) within the Cocos-East HPRA between 2014 and 2021 was likely a direct or indirect result of impacts associated with one or more of the heat stress events between 2014 and 2017, as was hypothesized for declines detected in all of the other HPRAs. However, the very limited amount of data available for 2014 and 2018, and the lack of data for other intervening years limit the degree of confidence in this assertion. The density of the spiky sea cucumber (*Stichopus chloronotus*) was already low in 2014, but without earlier survey data it's not clear if the 2014 densities represent a decrease relative to densities prior to the 2013 and 2014 heat stress events, or if they were normal for that species. In contrast to the sea cucumber community observed in other HPRAs even before the onset of multiple heat stress events in 2013, other sea cucumber taxa, such as *Holothuria edulis* and *Actinopyga varians* (formerly *A. mauritiana*) were commonly observed within the Cocos-East HPRA throughout the study period; these species may not have been impacted, or not impacted as severely, by the thermal stress, pathogen, and/or other driver(s) of the declines that appear to have greatly impacted *S. chloronotus*. An analysis of macroinvertebrate survey data collected along shallower (5 m) transects at the island bleaching response and recovery sites in 2013 and 2021 may shed some light on changes in *A. varians* and other macroinvertebrate taxa that are more common at shallow, wave exposed areas than in the mid-depth HPRA sites.





REFERENCES

Ault, J.S., S. G. Smith, J. A. Bohnsack, J. Luo, D. E. Harper, and D. B. McClellan, 2006. "Building sustainable fisheries in Florida's coral reef ecosystem: positive signs in the Dry Tortugas," *Bulletin of Marine Science*, vol. 78, no. 3, pp. 633–654, 2006.

Burdick, D. and L. Raymundo. 2018. An interim report of the Comprehensive Long-term Coral Reef Monitoring at Permanent Sites on Guam project. Report submitted to the Guam Bureau of Statistics and Plans and the NOAA Coral Reef Conservation Program.

Burdick, D., V. Brown, and R. Miller. 2019. A report of the Comprehensive Long-term Coral Reef Monitoring at Permanent Sites on Guam project. Report submitted to the NOAA Coral Reef Conservation Program and Government of Guam Bureau of Statistics and Plans.

Burdick, D., V. Brown, J. Asher, C. Caballes, M. Gawel, L. Goldman, A. Hall, J. Kenyon, T. Leberer, E. Lundblad, J. McIlwain, J. Miller, D. Minton, M. Nadon, N. Pioppi, L. Raymundo, B. Richards, R. Schroeder, P. Schupp, E. Smith, and B. Zgliczynski. 2008. Status of the Coral Reef Ecosystems of Guam. Bureau of Statistics and Plans, Guam Coastal Management Program. iv + 76 pp.

Green, A., M. Donahue, J. Caldwell, S. Heron, E. Geiger, and L. Raymundo. 2020. Coral disease time series highlight size-dependent risk and other drivers of white syndrome in a multi-species model. Frontiers and Marine Science 7: 601469.

Green, A. L. and D. R. Bellwood. 2009. Monitoring functional groups of herbivorous reef fishes as indicators of coral reef resilience — A practical guide for coral reef managers in the Asian Pacific region. IUCN working group on Climate Change and Coral Reefs. IUCN, Gland, Switzerland. 70 pp.

Heenan, A., I. D. Williams, T. Acoba, A. DesRochers, R. K. Kosaki, T. Kanemura, M. O. Nadon, and R. E. Brainard. 2017. Long-term monitoring of coral reef fish assemblages in the Western central pacific. Scientific Data 4: Article number: 17–176 (2010).

Johnson, M. S. Holbrook, R. Schmitt, and A. Brooks. 2011. Fish communities on staghorn coral: effects of habitat characteristics and resident farmerfishes. Environmental Biology of Fishes 91: 429–448.

Kuffner, I. and L. Toth. 2016. A geological perspective on the degradation and conservation of western Atlantic coral reefs. Conservation Biology 30(4): 706–715.

Maynard, J. S., Johnson, D. Burdick, A. Jarrett, J. Gault, J. Idechong, R. Miller, G. Williams, S. Heron, and L. Raymundo. 2017. Coral resilience to climate change in Guam in 2016. NOAA Coral Reef Conservation Program. NOAA Technical Memorandum CRCP29, 51 pp.

Minton, D., D. Burdick, and V. Brown. 2022. Changes in coral community structure across a sediment gradient in Fouha Bay, Guam. Marine Pollution Bulletin 181, August 2022, 113816. Myers R., Raymundo L. 2009. Coral disease in Micronesian reefs: a link between disease prevalence and host abundance. Diseases of. Aquatic Organisms 87:97–104.

Paulay, G. and Y. Benayahu. 1999. Patterns and consequences of coral bleaching in Micronesia (Majuro and Guam) in 1992–1994. Micronesica 31(2): 109–124.

***Previous page**: While much of the finer-branched Acropora species and encrusting Montipora at the reef front monitoring site (600) near Cocos Island (about 3 km to the east of the Cocos-East HPRA) were killed in the 2013 bleaching event, numerous large Acropora abrotanoides colonies such those in the top image were still present during surveys in 2015. However, many of the shallower colonies of this species were killed during the 2017 event, as well as by predation by the corallivorous snail, Drupella. Coral cover at this site declined from an exceptional 50% in 2013 to 12% in 2017 (a 76% decline), and had not shown signs of recovery between 2017 and 2021; this is in contrast to the increases observed at several of the other reef front monitoring sites. During a survey of the site in 2021, most of the benthic community at the site was comprised of turf algae that colonized the skeletons of the dead coral colonies (bottom).*

Randall, R. 1971. Tanguisson-Tumon, Guam, reef corals before, during and after the crown-of-thorns starfish (*Acanthaster planci*) predation. M.Sc. thesis, University of Guam, Mangilao.

Raymundo, L., D. Burdick, V. Lapacek, R. Miller, and V. Brown. 2017. Anomalous temperatures and extreme tides: Guam staghorn *Acropora* succumb to a double threat. Marine Ecology Progress Report 564: 47–55.

Raymundo L., D. Burdick, W. Hoot, R. Miller, V. Brown, T. Reynolds, J. Gault, J. Idechong , J. Fifer, A. Williams (in review). Successive bleaching events cause mass coral mortality in Guam, Micronesia.

Raymundo, L., M. Andersen, C. Moreland-Ochoa, A. Castro, C. Lock, N. Burns, F. Taijeron, D. Combosch, D. Burdick. 2022. Conservation and active restoration of Guam's staghorn *Acropora* corals. University of Guam Marine Laboratory Technical Report 168.

Redding J., R. Myers-Miller, D. D. Baker, M. Fogel, L. Raymundo, K. Kim. 2013. Link between sewage-derived nitrogen pollution and coral disease severity in Guam. Marine Pollution Bulletin 73:57–63.

Reynolds, T. 2016. Environmental regimes predict the spatial distribution of coral assemblages and climate-induced bleaching patterns around Guam. M.Sc. thesis, University of Guam, Mangilao.

Vargas-Ángel, B. 2012. "New data on benthic cover at Guam now available." NOAA Pacific Islands Fisheries Science Center Blog. https://pifscblog.wordpress.com/2012/10/25/data-benthic-cover-guam/. Accessed on 23 June 2023.

Williams, I. D., B.M. Richards, S. A. Sandin, J.K. Baum, R.E. Schroeder, M.O. Nadon, B. Zgliczynski, P. Craig, J.L. McIlwain, R.E. Brainard. 2011. Differences in reef fish assemblages between populated and remote reef spanning multiple archipelagos across the Central and Western Pacific. Journal of Marine Biology 2011: 1-14.

William, I. D., J. K. Baum, A. Heenan, K. M. Hanson, M. O. Nadon, and R. E. Brainard. 2015. Human, oceanographic and habitat drivers of central and western Pacific coral reef fish assemblages. PLoS ONE 10(4): e0120516. doi:10.1371/journal.pone.0120516.

Williams, I., C. Couch, O. Beijbom, T. Oliver, B. Vargas-Angel, B. Schumacher, and R. Brainard. 2019. Leveraging automated image analysis tools to transform our capacity to assess status and trends of coral reefs. Frontiers in Marine Science 6: 222.

APPENDIX A.

Guam Long-term Coral Reef Monitoring Program data collection activities at High Priority Reef Areas between 2009 and 2022.

P = permanent sampling stations, NP = Non-permanent sampling stations

Site	Survey year	Perm. (P) stations surveyed	Non-perm. (NP) stations surveyed	Fish SPC		Benthic cover		Corals		Macro-inverts		Rugosity	
				P	NP	P	NP	P	NP	P	NP	P	NP
Achang	2014	11	3	8	3	11	3	8	3	11	3	8	3
	2018	10	10	10	10	10	10	9	0	10	0	9	0
	2021	10	10	10	10	10	10	4	0	10	10	10	10
Cocos-East	2014	3	4	3	4	3	2	0	0	3	2	2	0
	2018	10	0	0	0	10	0	3	0	7	0	3	0
	2021	10	10	10	10	10	10	0	0	10	10	10	10
East Agana Bay	2010	10	10	10	10	10	10	10	10	5	10	0	0
	2012	10	0	0	0	10	0	10	0	10	0	10	0
	2014	10	3	0	0	10	0	10	0	10	0	10	0
	2015	10	0	10	10	10	0	10	0	10	0	10	0
	2017	10	12	10	12	10	12	10	0	10	0	9	0
	2018	10	0	0	0	10	0	0	0	10	0	10	0
	2019	10	6	10	6	10	6	9	0	10	6	9	1
	2020	10	10	10	10	10	10	10	0	10	10	10	10
	2021	10	10	10	10	10	10	0	0	10	10	10	10
	2022	10	10	10	10	10	10	0	0	10	10	10	10
Fouha Bay	2015	3, 5, 5	-	5, 5	-	13	-	13	-	13	-	13	-
	2019	3, 5, 5	-	5, 5	-	13	-	5,	-	13	-	13	-
	2021	3, 5, 5	-	5,5	-	13	-	5	--	13	-	13	-
Piti Bay	2012	4, 6	4, 6	6	8	10	10	10	10	10	10	8	10
	2014	4, 6	1, 1	10	2	10	2	10	2	10	2	4	1
	2018	10	10	10	10	10	10	3	0	3	0	3	0
	2020	10	10	10	10	10	10	0	0	10	10	10	10
	2022	10	10	10	10	10	10	1	0	10	10	10	10
Tumon Bay-2009	2009	2	5	1	4	2	5	2	5	0	0	0	0
Tumon Bay (original)	2010	10	10	10	10	10	10	10	10	8	7	0	0
Tumon Bay (modified)	2012	12	9	2	2	12	9	12	9	12	9	12	9
	2014	12	5	0	5	12	4	12	4	12	4	12	4
	2015	12	10	12	10	12	10	12	10	12	10	12	10
	2017	12	10	12	10	12	10	12	2	12	2	12	2
	2018	12	0	0	0	12	0	4	0	12	0	12	0
	2019	12	10	12	10	12	10	12	0	11	8	11	8
	2020	12	10	12	10	12	10	0*	0	12	10	3	9
	2021	12	10	12	10	12	10	0	0	12	10	12	10
	2022	12	10	12	10	12	10	0	0	12	10	12	10

*Coral colony size and health data were not recorded at the 12 permanent sampling stations in the Tumon (modified) HPRA in 2020, but nails were maintained, and images were obtained of all quadrats at all of these stations

APPENDIX B.

Comprehensive list of publications produced by the GLTMP or to which GLTMP staff contributed (listed by publication date from most recent).

Minton, D., D. Burdick, and V. Brown. 2022. Changes in coral community structure across a sediment gradient in Fouha Bay, Guam. Marine Pollution Bulletin 181, August 2022, 113816. https://www.sciencedirect.com/science/article/abs/pii/S0025326X22004982

Raymundo, L., M. Andersen, C. Moreland-Ochoa, A. Castro, C. Lock, N. Burns, F. Taijeron, D. Combosch, D. Burdick. 2022. Conservation and active restoration of Guam's staghorn *Acropora* corals. University of Guam Marine Laboratory Technical Report 168. https://www.uog.edu/_resources/files/ml/technical_reports/UOGML_TechRep168_Raymundo_2022.pdf

Heron, S., L. Raymundo, W. Sweet, A. Papa, C. Moreland-Ochoa, and D. Burdick. 2020. Predicting extreme tide events to inform shallow reef community restoration and management in Guam. NOAA Technical Memorandum CRCP 38 and UOG Marine Laboratory Technical Report 167. https://repository.library.noaa.gov/view/noaa/27292

Raymundo, L., D. Burdick, W. Hoot, R. Miller, V. Brown, T. Reynolds, and J. Gault. 2019. Successive bleaching events cause mass coral mortality in Guam, Micronesia. Coral Reefs 38: 677–700. https://link.springer.com/article/10.1007/s00338-019-01836-2

Burdick, D., V. Brown, and R. Miller. 2019. A report of the Comprehensive Long-term Coral Reef Monitoring at Permanent Sites on Guam project. Report submitted to the NOAA Coral Reef Conservation Program and Government of Guam Bureau of Statistics and Plans. https://www.guamcoralreefs.info/resources/#management-plans-and-reports

Burdick, D. and L. Raymundo. 2018. An interim report of the *Comprehensive Long-term Coral Reef Monitoring at Permanent Sites on Guam* project. Report submitted to the NOAA Coral Reef Conservation Program and Government of Guam Bureau of Statistics and Plans.

NOAA Pacific Islands Fisheries Services Center and NOAA Coral Reef Conservation Program. 2018. Coral reef condition: A status report for Guam. https://repository.library.noaa.gov/view/noaa/19538

Maynard, J., S. Johnson, D. Burdick, A. Jarrett, J. Gault, J. Idechong, R. Miller, G. Williams, S. Heron, and L. Raymundo. 2017. Coral reef resilience to climate change in Guam in 2016. NOAA Coral Reef Conservation Program. NOAA Technical Memorandum CRCP29. https://www.coris.noaa.gov/activities/guam_coral_resilience/

Raymundo, L., D. Burdick, V. Lapacek, R. Miller, V. Brown. 2017. Anomalous temperatures and extreme tides: Guam staghorn *Acropora* succumb to a double threat. Marine Ecology Progress Series 564:47-55. https://www.int-res.com/abstracts/meps/v564/p47-55/

Fenner, D. and D. Burdick. 2016. Field identification guide to the threatened corals of the U.S. Pacific Islands. NOAA National Marine Fisheries Service, Pacific Islands Regional Office, Protected Resources Division. https://www.ncei.noaa.gov/data/oceans/coris/library/NOAA/CRCP/NMFS/PIFSC/Projects/31066/Fenner2016_CoralsID.pdf

Hoot, W. and D. Burdick. 2016. Guam Coral Bleaching Response Plan. Bureau of Statistics and Plans. https://www.guamcoralreefs.info/resources/#management-plans-and-reports

Reynolds, T., D. Burdick, P. Houk, L. Raymundo, S. Johnson. 2014. Unprecedented coral bleaching across the Mariana Archipelago. Coral Reefs 33: 499. https://link.springer.com/article/10.1007/s00338-014-1139-0

Boulay, J., D. Burdick, A. Halford, J. McIlwain, and I. Baums. 2014. Genetic connectivity and spatial clonal structure in a dominant staghorn coral on reefs around Guam. Chapter 5, Dissertation in Biology, The Pennsylvania State University. https://etda.libraries.psu.edu/files/final_submissions/10034

Burdick, D., R. Miller. 2012. Comprehensive long-term monitoring at permanent sites on Guam: 2012 status report. Report submitted to the NOAA Coral Reef Conservation Program. https://www.guamcoralreefs.info/resources/#management-plans-and-reports

Burdick, D., V. Brown. 2011. Comprehensive long-term monitoring at permanent sites in Guam: Report of program status and presentation of preliminary baseline data and power analyses results for Tumon Bay, East Agana Bay, and Western Shoals sites. Report submitted to the NOAA Coral Reef Conservation Program. Available at https://www.coris.noaa.gov/search/catalog/search/resource/details.page?uuid=%7B9AF65D1F-668E-4D2B-99A0-4B0D2907F1A8%7D

Minton, D., D. Burdick, J. den Haan, S. Kolinksi, and T. Schils. 2009. Comparison of a photographic and in situ method to assess the coral reef benthic community in Apra Harbor, Guam. Prepared for Naval Facilities Engineering Command Pacific, Pearl Harbor, Hawai'i. https://www.uog.edu/_resources/files/ml/technical_reports/135Minton_et_al_2009_UOGMLTechReport135.pdf

Goldberg, J., K. Adams, J. Albert, J. Asher, P. Brown, V. Brown, D. Burdick, B. Carroll, P. Craig, D. Fenner, C. Fillmed, V. Fread, M. Gawel, A. George, Y. Golbuu, L. Goldman, C. Graham, A. Hall, M. Hasurmai, L. Jacob, D. Jacobson, E. Joseph, J. Kenyon, W. Kostka, T. Leberer, M. Luckymis, E. Lundblad, S. Malakai, J. Maragos, A. Marcus, S. Marino, D. Mathias, J. McIlwain, J. Miller, D. Minton, M. Nadon, S. Palik, N. Pioppi, L. Raymundo, B. Richards, M. Sebater, R. Schroeder, P. Schuup, E. Smith, A. Takesy, and B. Zgliczynski. 2008. Status of Coral Reef Resources in Micronesia and American Samoa: 2008. pp. 199-212. In: Wilkinson (ed.), Status of Coral Reefs of the World: 2008. https://researchonline.jcu.edu.au/24211/1/Status_of_Micronesia_and_AS.pdf

Burdick, D., V. Brown, J. Asher, C. Caballes, M. Gawel, L. Goldman, A. Hall, J. Kenyon, T. Leberer, E. Lundblad, J. McIlwain, J. Miller, D. Minton, M. Nadon, N. Pioppi, L. Raymundo, B. Richards, R. Schroeder, P. Schupp, E. Smith, and B. Zgliczynski. 2008. Status of the Coral Reef Ecosystems of Guam. Bureau of Statistics and Plans, Guam Coastal Management Program. iv + 76 pp. Available at https://www.researchgate.net/publication/236590561_Status_of_the_Coral_Reef_Ecosystems_of_Guam

Burdick, D., V. Brown, J. Asher, M. Gawel, L. Goldman, A. Hall, J. Kenyon, T. Leberer, E. Lundblad, J. McIlwain, J. Miller, D. Minton, M. Nadon, N. Pioppi, L. Raymundo, B. Richards, R. Schroeder, P. Schupp, E. Smith, and B. Zgliczynski. 2008. The State of the Coral Reef Ecosystems of Guam. pp. 465-509. In: J.E. Waddell and A.M. Clarke (eds.). The State of the Coral Reef Ecosystems of the United States and Pacific Freely Associated States: 2008. NOAA Technical Memorandum NOS NCCOS 73. NOAA/NCCOS Center for Coastal Monitoring and Assessment's Biogeography Team. Silver Spring, MD. 569 pp. https://coastalscience.noaa.gov/data_reports/the-state-of-coral-reef-ecosystems-of-the-united-states-and-pacific-freely-associated-states-2008/